Black-throated Finch Research Program Final Report

1 December 2025

Bravus Mining and Resources

Carmichael Coal Mine



Document Management

Rev.	Issue Date	Description	Author (s)	Approved	Signature
A	18/11/2024	Working draft for initial consultation	John van Osta Brad Dreis Jemma McCrossin Breanna Williams Additional authors acknowledged in-text	Brad Dreis	
B	13/03/2025	Issued for independent peer review	John van Osta Brad Dreis Jemma McCrossin	Brad Dreis	
0	15/08/2025	Issued for review	John van Osta Brad Dreis Jemma McCrossin	Brad Dreis	
1	1/09/2025	Issued for use	John van Osta	Brad Dreis	
2	1/12/2025	Issued for use. Revised in response to DETSI comments.	John van Osta Jemma McCrossin	Brad Dreis	
3	1/12/2025	Issued for use. Revised in response to DETSI comments.	John van Osta Jemma McCrossin	Brad Dreis	

Document Reference: X:\JOBS\~2019\QEJ19152\DELIVERABLES\FINAL RESEARCH REPORT\REV 2\BTFRP_Final_Report_Rev_2.docx

DISCLAIMER

1. **Scope, Use and Purpose**
   a. This document has been prepared by E2M solely for Bravus Mining and Resources and may only be used and relied upon by Bravus Mining and Resources for the specific purpose agreed between E2M and Bravus Mining and Resources (**Agreed Purpose**).
   b. This document may not contain sufficient information for purposes extraneous to the Agreed Purpose and E2M will not be liable for any loss, damage, liability or claim if this document or its contents is used or relied upon for any purpose extraneous to the Agreed Purpose.
2. **Limitations of this document**
   a. The opinions, conclusions, recommendations and information included in this document are:
      i. limited to the scope of the relevant engagement agreed between E2M and Bravus Mining and Resources;
      ii. limited by the limitations indicated in this document;
      iii. based on E2M's knowledge and approach, and the conditions encountered and information reviewed by E2M, as at the date of the preparation of this document (**Prevailing Knowledge**);
      iv. based on E2M's assumptions described or indicated in this document (**Assumptions**); and
      v. based on information provided to E2M by Bravus Mining and Resources and others including government authorities (**Supplied Information**).
   b. Bravus Mining and Resources acknowledges that any Prevailing Knowledge may have ceased or may in the future cease to be correct, accurate or appropriate in light of subsequent knowledge, conditions, information or events. E2M has no obligation to update Bravus Mining and Resources with respect to changes in the Prevailing Information occurring after the date this document was prepared.
   c. While E2M does not have any reason to believe any Assumptions are incorrect, E2M has not made any independent investigations with respect to the Assumptions and shall have no liability arising from any incorrect Assumptions.
   d. Supplied Information has not been independently verified by E2M. E2M shall have no liability in connection with Supplied Information, including errors and omissions in this document which were caused by errors or omissions in the Supplied Information.
3. **Warranties, Liabilities and Consequential Loss**
   a. A reference to 'liability' or 'liable' in this disclaimer refers to any liability for any direct or indirect loss, damage, liability, cost, expense or claim.
   b. E2M excludes implied warranties to the extent legally permissible and shall have no liability arising out of the reliance on such implied warranties.
   c. E2M shall have no liability for any interpretation, opinion or conclusion that Bravus Mining and Resources may form as a result of examining this document.
   d. Bravus Mining and Resources acknowledges and agrees that the maximum aggregate liability of E2M in connection with the preparation and provision of this document is limited to the value of the consideration paid or payable by Bravus Mining and Resources to E2M for it.
   e. E2M will not be liable to Bravus Mining and Resources or any other person for any special, indirect, consequential, economic loss, or loss of profit, revenue, business, contracts or anticipated savings suffered or incurred by Bravus Mining and Resources or any other person arising out of or in connection with the provision of this document.
4. **Third Parties**
   a. This document may not, without E2M's prior written consent, be disclosed to any person other than Bravus Mining and Resources (**Third Party**).
   b. This document may not contain sufficient information for the purposes of a Third Party and is prepared and provided without E2M assuming or owing a duty of care to any Third Party.
   c. E2M will not be liable to a Third Party for any liability arising out of or incidental to this document or any publication of, use of or reliance on it (Third Party Liability). Bravus Mining and Resources and any Third Party assumes all risk, and releases, indemnifies and will keep indemnified E2M from any Third Party Liability.

Contact Information	
Ph: 07 3062 6960	Website: e2mconsulting.com.au
Level 3, 19 Lang Parade, Milton, Queensland, 4064	PO Box 1444, Milton, Queensland, 4064

Contents

Definitions xv

Abbreviations xv

Summary xvi

1 Study background 19
- 1.1 Project background 19
- 1.2 Research objectives 21
- 1.3 Structure of this report 21
- 1.4 Southern black-throated finch species profile 23

2 Research project design 25
- 2.1 Project Area description 25
- 2.2 Permits and approvals 31
- 2.3 Surveys and timing 31
- 2.4 Research methods overview 34

3 Aim 1: Estimate the home range and movements of SBTF in the Project Area 36
- 3.1 Home range, landscape scale habitat selection and daily activity patterns 36
- 3.2 Landscape genetics 57
- 3.3 Dispersal movements of southern black-throated finch over three years 75
- 3.4 Sex-related variation in the morphometrics, movement ecology and diet of the southern black-throated finch 85
- 3.5 Southern black-throated finch population estimate 100

4 Aim 2: Determine the foraging and breeding habitat requirements and the relationship with vegetation structure and composition and water 111
- 4.1 Nest site and habitat selection 111
- 4.2 Foraging habitat selection 131

5 Aim 3: Determine the dietary requirements for SBTF in the Project Area 152
- 5.1 Diet of Black-throated finch 152
- 5.2 Grass seed phenology 178

6 Aim 4: Identify management strategies for SBTF habitat in the Project Area regarding fire, grazing and water 188
- 6.1 Relative impacts of grazing and fire on habitat quality for black-throated finch 188
- 6.2 Response of black-throated finch to the provision of artificial water sources 203
- 6.3 Design guidelines for black-throated finch friendly water sources 218

7 Additional research 221
- 7.1 A review of southern black-throated finch survey and monitoring methods 221
- 7.2 A survey of southern black-throated finch presence and habitat characteristics within the Desert Uplands Bioregion 240
- 7.3 Environmental DNA detection trial 254
- 7.4 Development of an automated SBTF call detection model 261

7.5 Development of a data pipeline to localise automated radio telemetry tower data 277

8 A review of land management strategies to support habitat quality for the southern black-throated finch 298

8.1 Abstract 298

8.2 Introduction 298

8.3 Important habitat requirements for the southern black-throated finch 299

8.4 Habitat management 301

8.5 Conclusion 307

9 Recommendations 308

9.1 General habitat management recommendations for the southern black-throated finch 308

9.2 Recommended updates to the Project's Black-throated Finch Management Plan 312

9.3 Opportunities for additional research 325

10 References 328

List of Tables

Table 2.3-1: Survey events, timing and methods employed 32

Table 2.4-1: Summary of methods employed for the BTFRP 34

Table 3.1-1: Vegetation communities within the study area 40

Table 3.1-2: Estimated regression parameters, standard errors, z-values and p values for a generalised linear mixed effects model relating home range size to season. 49

Table 3.1-3: Estimated regression parameters, standard errors, t-values and P-values for the linear model assessing the relationship among activity, season and time of day. 52

Table 3.2-1: 1-dimensional demographic models used in the δaδi analysis to infer demographic history of black-throated finch in their southernmost range of the Desert Uplands Bioregion. $\boldsymbol{T}$ is in time of $\boldsymbol{2Nref}$ generations ago, and $\boldsymbol{nu}$ are ratios of effective population size relative to $\boldsymbol{Nref}$. 64

Table 3.2-2: Measures of regional neutral autosomal genetic diversity for black-throated finches in the Desert Uplands Bioregion. $\boldsymbol{Ar}$ = allelic richness (accounting for differences in sample sizes, n), $\boldsymbol{Ho}$ = observed heterozygosity, $\boldsymbol{He}$ = unbiased expected heterozygosity, and $\boldsymbol{FIS}$ = inbreeding coefficient. Bootstrapped 95% C.I.'s are shown in brackets (999 bootstraps). Declines in $\boldsymbol{Ar}$ (diversity at the allelic level) may impair a populations capacity to adapt to environmental changes, while declines in $\boldsymbol{He}$ (diversity at the genotypic level) have been linked with decreases in the average fitness of individuals (Fisher, 1930; Reed and Frankham, 2003). Management plans aimed at conserving genetic diversity should consider both types for long-term population persistence (Greenbaum *et al.*, 2014). 64

Table 3.2-3: Percentage of samples (including related individuals) either assigning correctly to the region they were sampled in (i.e., no mixed ancestry), assigning to a different region (i.e., potential 1st generation migrant to the region they were sampled in), or having a parent, grandparent or great grandparent that migrated from a different region (i.e., recent ancestry in a different region, back to three generations ago). Only results with posterior probabilities of assignment P > 0.5 or 0.9 are shown. 66

Table 3.2-4: Estimates of the effective number of breeders ($\boldsymbol{Nb}$) for all SBTF samples in the Desert Uplands population, accounting for overlapping generations, and effective population size ($\boldsymbol{Ne}$), under different estimates of age at sexual maturity (α) and adult lifespan ($\boldsymbol{AL}$) in years, rounded to the nearest 10. Lower and upper give the 95% jack-knifed confidence intervals. 70

Table 3.2-5: Mean estimates of the effective number of breeders ($\boldsymbol{Nb}$) for the North-Central and South subpopulations of SBTF in the Desert Uplands Bioregion, accounting for overlapping generations, and effective population size ($\boldsymbol{Ne}$), under different estimates of age at sexual maturity (α) and adult lifespan ($\boldsymbol{AL}$) in years, rounded to the nearest 10. Estimates for the North-Central subpopulation are based on a $\boldsymbol{Pcrit}$ of 0.02 and for the South 0.03; based on the $\boldsymbol{Pcrit}$ rule of thumb proposed by Waples and Do (2010) for n ≤ 25 ($\boldsymbol{1/2n < Pcrit < 1/n}$). 71

Table 3.4-1: Assessment of morphological variation between male (n = 58) and female (n = 53) southern black-throated finch 93

Table 3.4-2: Estimated regression parameters, standard errors, z-values and p values for a generalised linear model relating home range size to sex and season. 93

Table 3.4-3: Relationships between proportion of taxa in diet, season and sex for a generalised linear model with an ordered beta distribution. 95

Table 3.5-1: Number of marked SBTF captured and resighted during survey periods 101

Table 3.5-2: Population size estimates for the northern subpopulation of SBTF within the study area calculated using the mark-resight method 102

Table 3.5-3: Number of 1 km transects undertaken and southern black-throated finch recorded during the study period 103

Table 3.5-4: Population size estimates derived from distance sampling 104

Table 3.5-5: Survey effort during water source census per methodology and subpopulation 105

Table 3.5-6: Number of SBTF recorded during water source census per methodology and subpopulation 107

Table 3.5-7: Comparison of southern-black throated finch population estimates 110

Table 4.1-1 Climatic and vegetation characteristics of the Townsville Plains and Desert Uplands which host the remaining black-throated finch 'stronghold' populations 114

Table 4.1-2: Summary of environmental variables included in candidate models assessing habitat characteristics influencing nest-site selection by black-throated finch 119

Table 4.1-3: Nest and host tree characteristics of black-throated finch nests recorded in the Desert Uplands 122

Table 4.1-4: Host species containing black-throated finch nests in the Desert Uplands 123

Table 4.1-5: Vegetation characteristics of black-throated finch nesting habitat in the Desert Uplands 125

Table 4.1-6: Results of AIC-based model selection for black-throated finch nest site selection. Number of model parameters (k), maximised log-likelihood values (logLik), AIC values (AIC), AIC differences (delta) and Akaike weights (weight) are shown for each of the models. 126

Table 4.2-1: Climatic and vegetation characteristics of the Townsville Plains and Desert Uplands which host the remaining black-throated finch 'stronghold' populations 134

Table 4.2-2: Number of 1 km transects completed within each RE 137

Table 4.2-3: Summary of landscape and site-scale environmental variables used to assess the habitat characteristics influencing forage-site selection of black-throated finch 139

Table 4.2-4: Comparison of ground and shrub cover characteristics for three vegetation communities in the Desert Uplands where black-throated finch were observed foraging 142

Table 4.2-5:Comparison of ground and shrub cover characteristics for black-throated finch foraging sites and pseudo-absence sites in the Desert Uplands 142

Table 4.2-6: Results of AIC-based model selection for black-throated finch foraging habitat selection using a mixed-effect with the presence or absence of foraging as the response variable. Number of model parameters (k), maximised log-likelihood values (logLik), AIC values (AIC), AIC differences (delta) and Akaike weights (weight) are shown for each of the models. 143

Table 5.1-1: Number of crop and faecal samples taken during mist-netting 158

Table 5.1-2 Summary statistics (number of sites, mean percentage cover and standard deviation) and comparison (Mann-Whitney U) of the cover of important grass species in foraging and systematically collected vegetation survey plots 160
Table 5.1-3: Visual foraging observations of southern black-throated finch within the study area between 2021 and 2023 161
Table 5.1-4: DNA metabarcoding results from crop and faecal samples taken between 2022 and 2023 showing the number of samples in which each taxon was recorded, as well as the mean and maximum proportion contributed to each sample. 167
Table 5.2-1: Number of vegetation monitoring sites at which phenology data were collected 179
Table 5.2-2: Summary of seeding phenology of grass genera within the study area 181
Table 6.1-1: Number of vegetation monitoring sites surveyed per year and season 194
Table 6.1-2: Summary of the conceptual effects of fire and grazing regimes on ground cover composition. 200
Table 6.2-1: Number of 1km transects carried out per season 205
Table 6.2-2: Number of camera days of survey effort per water source type and season 210
Table 6.3-1: Design guidelines for southern-black throated finch friendly water sources 219
Table 7.1-1: Duration and timing of water source watches during case study 223
Table 7.1-2: Sample sizes of survey events and survey periods used to calculate SBTF detection rates 226
Table 7.1-3: A review of survey and monitoring methods for the southern black-throated finch (SBTF) 229
Table 7.1-4: Selection table of SBTF detection and monitoring methods 237
Table 7.2-1: Records of southern black-throated finch collated from field surveys on Moray Downs between 2011 and 2022 as well as from the Atlas of Living Australia, BirdLife Australia Atlas and Wildnet databases 243
Table 7.2-2: Environmental variables used in MaxEnt modelling 244
Table 7.2-3: Properties accessed 246
Table 7.2-4: Percentage decrease in SBTF habitat within the Desert Uplands Bioregion 249
Table 7.3-1: Number of tater sources sampled for environmental DNA across two water source types (surface water and troughs) and differing frequency of use by southern black-throated finch. 256
Table 7.3-2: Number of tater sources sampled for environmental DNA across two water source types (surface water and troughs) and differing frequency of use by southern black-throated finch. 257
Table 7.4-1:The number of target and non-target audio frames used within each model iteration 268
Table 7.4-2: Audio annotation rules provided to annotators 269
Table 7.5-1: Candidate models evaluating the effects of localisation method, mean distance to the receiver, number of receivers, pulse interval and mean relative signal strength (RSS) on the positional error of location estimates. 289
Table 7.5-2: Relationships between positional error and fixed factors for the top performing model identified in Table 7.5-1. Methods were compared using a generalised linear mixed effects model with a log-link gamma distribution. Model estimates (β and 95% confidence intervals [CIs]) have been scaled. 289
Table 8.4-1: Summary of key forage species for SBTF found within the Project Area and their response to grazing 304
Table 8.4-2: Summary of key forage species for SBTF found within the Project Area and their response to fire 305
Table 9.1-1: Recommended detection and monitoring methods for the southern black-throated finch (SBTF) 311
Table 9.2-1: Recommended updates to the Black-throated Finch Management Plan (BTFMP; EcoLogical Australia, 2023) management requirements (Tables 8 and 9 of the BTFMP) 314
Table 9.2-2: Recommendations for SBTF monitoring methods 319

Table 9.3-1: Responses to relevant conditions, reporting requirements and research objectives identified within the EA (EPML01470513), BTFRP and BTFP Project Plan Section 3.5 377

List of Figures

Figure 1.1-1: Project Area 20
Figure 1.3-1: Conceptual diagram of the structure of this report 22
Figure 1.4-1: Regional context 24
Figure 2.1-1: Southern black-throated finch records and vegetation communities within the Project Area 27
Figure 2.1-2: Time since last burn 28
Figure 2.1-3: Fire frequency 29
Figure 2.1-4: MAXENT habitat mapping 30
Figure 3.1-1: Study area for southern black-throated finch research in savanna vegetation within the Desert Uplands region of Queensland, Australia. The map depicts the locations of automated radio towers, mist-netting sites, vegetation communities, major roads, major watercourses and mining infrastructure. Abbreviations include A = *Acacia*, E = *Eucalyptus*, M = *Melaleuca*, O.F. = open forest and W = woodland. 39
Figure 3.1-2: Map of the study area showing the locations of presence and pseudoabsence sites as well as mean wet and dry season SBTF home ranges. 44
Figure 3.1-3: Distribution of presence sites and pseudoabsence sites relative to their distance to the nearest vehicle track. 45
Figure 3.1-4: Histogram showing the distribution of radio tracked southern black-throated finch in the Desert Uplands region of Queensland, Australia. Resightings are relative to the distance from their point of capture and include resightings from automated radio telemetry (ART, red) and manual radio tracking (blue). 47
Figure 3.1-5: Odds ratios of the predictor variables of a logistic regression resource use model in the Desert Uplands region of Queensland, Australia. Odds ratios above 1 represent a use preference by southern black-throated finch (blue), whereas odds ratios lower than 1 represent avoidance (red). The error bars show 95% confidence intervals. ***p < 0.001, **p < 0.01, *p < 0.05. Abbreviations include A = *Acacia*, E = *Eucalyptus*, M = *Melaleuca*, O.F. = open forest and W = woodland. 48
Figure 3.1-6: 95% autocorrelated kernel density estimate (AKDE) home ranges and 50% AKDE core ranges compared to the effective sample sizes that are calculated within the ctmm package. 49
Figure 3.1-7: Daily activity records of southern black-throated finch by time of day and season. Numbers show a count of the number of observations for each bar. 51
Figure 3.1-8: Diel activity patterns of southern black-throated finch. The bars show confidence intervals established using a linear model. Dusk and night periods were excluded from the linear model due to sample size constraints. 52
Figure 3.2-1: Genetic sample locations 61
Figure 3.2-2: Sex inference results for black-throated finches. A-C) correlations between pairs of sex-associated features used in the analysis, coloured by sex inferred with *mclust* clustering method (green = males, yellow = females). D) Assignment probability of samples to each cluster (cluster 1 = female, cluster 2 = male). Males are ZZ in birds, and thus are expected to have higher Z heterozygosity, a higher W missing rate and a higher Z/W ratio. 62
Figure 3.2-3: A) Map of southern black-throated finch samples collected around the Carmichael Coal Mine and coloured by region (blue = North, red = Central, green = South). B) Discriminant analysis of principal components (DAPC) showing the first two discriminant functions, with individuals coloured by region of origin. C) Spatial autocorrelation analysis (SAA) showing mean within distance-class genotypic correlation (related to kinship). Irregular distance classes were chosen to maximise

sample size per distance class, solid error bars denote 95% bootstrapped C.I. about the mean r within each distance class (999 bootstraps), dashed line denotes the 95% C.I. for the hypothesis of 'no spatial structure' (999 permutations). D) STRUCTURE plot of admixture coefficients for each individual, assuming K = 3 ancestral populations (highest delta K). 65

Figure 3.2-4: Mean kinship among black-throated finch compared A) between sexes, B) within versus between localities, C) within versus between regions, and D) between each region. Kinship was estimated with KING accounting for genetic structure with PC-AiR and PC-Relate, using neutral, autosomal loci and the 'unrelated' dataset (no full- or half-sibs). Note that samples size is larger in the Central region. 67

Figure 3.2-5: Spatial principal coordinates analysis (sPCA) for black-throated finches across their distribution in the lower Desert Uplands Bioregion. A) Geographic location of samples coloured by region (blue = north, red = central, green = south). B) Map of interpolated individual scores (blue = low, red = high) based on the first PC (first global score). Contour lines that are closer together indicate greater genetic differentiation between localities. C) Plot of decomposed genetic variation into variance (due to population structure) and Moran's *I* (spatial autocorrelation) (blue = global structure, red = local structure), with the top global score circled. D) Distribution of eigenvectors from the most positive (blue = global structure) to negative (red = local structure). 68

Figure 3.2-6: Estimated effective migration rates calculated by EEMS. A) Plot of interpolated posterior mean migration rates *m* ($\log_{10}$ scale) showing areas of higher than expected (blue) or lower than expected migration rates (orange). Circles indicate sampling locations with size depicting the number of samples per deme. B) Plot showing highlighted regions (white outline) where effective migration rates are significantly higher (blue) or lower (orange) than the overall average rate of migration. 69

Figure 3.2-7: Stairway plot showing historical changes in the median effective population size (Ne) of black-throated finches in the southern Desert Uplands Bioregion, over A) the last million years and B) the last 1,000 years, with shaded polygons representing 95% C.I.'s (light grey) and 75% C.I.'s (dark grey). Demographic events inferred by ðaði (best-fit model = 3 epoch) are indicated in orange. Results for both approaches are shown based on a mutation rate per site per generation of 5.85×10^{-9} and a generation length (g) of 2 years. 72

Figure 3.3-1: Study area of the colour band resight program. 77

Figure 3.3-2: Distribution of maximum distance from the capture location for southern-black throated finch with one or more resighting or recapture. 79

Figure 3.3-3: Mist-netting recaptures and colour band resightings of black-throated finch. 80

Figure 3.3-4: Relationship between the distance from the initial capture location compared to the time from first capture for mist-netting and colour band resightings of southern-black throated finch. A loess regression (blue line) was fitted, with 95% confidence intervals (grey). 81

Figure 3.3-5: Distribution of the maximum distance from the initial capture location for adult (n = 157) and immature (n = 11) southern black-throated finch, recorded at the time of their first capture. 82

Figure 3.3-6: Ratio of juvenile to adult SBTF captured during mist-netting split by season 83

Figure 3.4-1: Overview of the study area 89

Figure 3.4-2: Sex inference results for black-throated finches. A-C) correlations between pairs of sex-associated features used in the analysis, coloured by sex inferred with mclust clustering method (green = males, yellow = females). D) Assignment probability of samples to each cluster (cluster 1 = female, cluster 2 = male). Males are ZZ in birds, and thus are expected to have higher Z heterozygosity, a higher W missing rate and a higher Z/W ratio. 90

Figure 3.4-3: Photo of SBTF throat patch 91

Figure 3.4-4: Home ranges of female (n = 25) and male (n = 29) southern black-throated finch separated in the dry (red, n = 34) and wet (blue, n = 20) seasons). Lines show a 95% confidence interval of the mean. Home ranges were calculated using a 95% autocorrelation corrected kernel density estimate (van Osta *et al.*, 2024). A random horizontal jitter has been added to aid visualisation of the distribution of data. 94

Figure 3.4-5: Maximum distance female (n = 23) and male (n = 29) southern black-throated finch were resighted from their initial capture location. A random horizontal jitter has been added to aid visualisation of the distribution of data. 94

Figure 3.4-6: Interaction effects of taxon and sex on the proportion of DNA reads for crop and faecal samples of female (n = 34) and male (n = 25) southern black-throated finch. Means and interactions effects have been derived using a generalised linear model with an ordered beta distribution. Lines show a 95% confidence interval of the mean. Taxa that contributed less than 1% of DNA reads were excluded from the analysis. 95

Figure 3.5-1: A calibration curve showing the average difference in SBTF count between a water source watch and remote camera results grouped by various observation periods, ranging from 3 seconds to 30 minutes. 106

Figure 3.5-2: SBTF records throughout the day at water source monitoring sites during the 2024 dry and wet season surveys. The number of BTF records are not directly comparable between the surveys due to differences in the number of cameras deployed and sites at which they were deployed. 108

Figure 4.1-1: Study area for black-throated finch research undertaken within the Desert Uplands bioregion of Queensland. 115

Figure 4.1-2: Layout of 100 x 50 m vegetation plots that were surveyed to characterise black-throated finch habitat throughout the study area. 119

Figure 4.1-3: Box plots of nest heights for the most common black-throated finch nest host species in the Desert Uplands 124

Figure 4.1-4: Visibility of black-throated finch nests in the most commonly recorded host species 125

Figure 4.1-5: Predicted probabilities (solid line) with 95% confidence intervals (grey area) of linear mixed model for black-throated finch nest presence with respect to: (a) NDVI; (b) canopy height; (c) shrub cover; (d) grass cover; and (e) bare ground cover. 127

Figure 4.1-6: Predicted probabilities (solid line) with 95% confidence intervals (grey area) of black-throated finch nest presence with respect to grass cover at three different bare ground cover classes: (a) 9% bare ground cover; (b) 22% bare ground cover; and (c) 35% bare ground cover. 128

Figure 4.2-1: Study area for black-throated finch research undertaken within the Desert Uplands bioregion of Queensland. 136

Figure 4.2-2 Layout of 100 x 50 m vegetation plots surveyed at 200 sites in order to characterise black-throated finch habitat throughout the study area. 138

Figure 4.2-3: Predicted probabilities (solid line) with 95% confidence intervals (grey area) of black-throated finch foraging sites with respect to NDVI in: (a) dry season (June to November); and (b) wet season (December to May). 144

Figure 4.2-4: Predicted probabilities (solid line) with 95% confidence intervals (grey area) of black-throated finch foraging sites with respect to distance to permanent water in: (a) dry season (June to November); and (b) wet season (December to May). 145

Figure 4.2-5: – Predicted probabilities (solid line) with 95% confidence intervals (grey area) of black-throated finch foraging sites with respect to site-scale variables: (a) native grass richness; (b) native grass cover; (c) shrub cover; and (d) bare ground. 146

Figure 4.2-6: Predicted probabilities (solid line) with 95% confidence intervals (grey area) of black-throated finch foraging sites with respect to the cover of native grass species: (a) *Digitaria ciliaris*; (b) *Digitaria brownii*; (c) *Triodia pungens*; (d) *Heteropogon contortus*; (e) *Themeda triandra*; (f) *Panicum effusum*; (g) *Aristida* spp,; (h) *Chrysopogon fallax*; (i) *Paspalidium rarum*; (j) *Enneapogon* spp.; (k) *Schizachyrium fragile*; (l) *Urochloa* spp.; (m) *Eriachne mucronata*; (n) *Sehima nervosum*; and (o) *Cymbopogon bombycinus*. Red lines indicate species that had a significant effect ($p<0.05$) in Model 5. 148

Figure 4.2-7: Predicted probabilities (solid line) with 95% confidence intervals (grey area) of black-throated finch foraging sites with respect to the cover of exotic ground cover species: (a) *Stylosanthes scabra*; and (b) *Cenchrus ciliaris*. 148

Figure 5.1-1: Study area for black-throated finch research undertaken within the Desert Uplands bioregion of Queensland. 156
Figure 5.1-2 Box plots comparing the cover (%) of grass species and bare ground recorded at black-throated finch foraging site (blue) and systematic vegetation plots (orange) across the study area: (a) bare ground; (b) *Cenchrus ciliaris*; (c) *Digitaria brownii*; (d) *Digitaria ciliaris*; (e) *Enneapogon virens*; (f) *Heteropogon contortus*; (g) *Panicum effusum*; (h) *Paspalidium rarum*; (i) *Sehima nervosum*; (j) *Themeda triandra*; and (k) *Urochloa subquadripara*. 163
Figure 5.1-3: Histogram depicting the average percentage of *Digitaria ciliaris* cover per site (top) and number of sites (bottom). Results are divided by whether the data was captured via a systematic vegetation plot consisting of 10 1x1 m quadrats per site or an opportunistic foraging plot consisting of a 1x2 m quadrat undertaken after an SBTF was observed foraging. 165
Figure 5.1-4: Foraging observations of southern black-throated finch feeding on *Digitaria ciliaris* compared to other species. 166
Figure 5.1-5: Contribution of insects and arthropods to the black-throated finch diet during the dry season (red) and wet season (blue) based on DNA metabarcoding records. 169
Figure 5.1-6: Principle Coordinate Analysis (PCoA) based on Bray Curtis dissimilarity metrics showing the variance of the diet composition of black-throated finch, based on DNA analysis of crop and faecal samples, between the wet (blue) and dry (red) seasons in the Desert Uplands. 170
Figure 5.1-7: Jacobs' Index calculated for 28 grass species that black-throated finch were observed foraging on during the wet season between 2021 and 2023. 171
Figure 5.1-8: Jacobs' Index calculated for the wet and dry seasons of 20 of the genera that DNA metabarcoding identified in black-throated finch crop and faecal samples collected between 2022 and 2023. 172
Figure 5.1-9: Relationship between Jacobs Index and metabolisable energy content of the seeds of 18 grass species that occur in the study area. 173
Figure 5.1-10: Relationship between Jacobs' Index and crude protein content of the seeds of 18 grass species that occur in the study area. 174
Figure 5.1-11: Relationship between Jacobs' Index and seed length content of the seeds of 20 grass species that occur in the study area. 175
Figure 5.2-1: Daily rainfall (green) and soil moisture (blue) recorded within the study area. 180
Figure 5.2-2: Mean seed availability held on inflorescences throughout an average climatic year (black line), depicted using a LOESS regression with 95% confidence intervals (shaded region). 184
Figure 6.1-1: Study area showing the locations of vegetation monitoring transects and grazing management units 191
Figure 6.1-2: Locations of burns undertaken during the study period 192
Figure 6.1-3: Effect sizes of stocking rate and time since the last burn on the cover of ground layer components. The central point (dark blue) shows the median effect size and lines (light blue) show 95% credible intervals of the median. All parameter estimates have been calculated from the posterior distribution of a Bayesian zero-inflated beta regression model. Values above 0 on the x-axis represent an increase in cover of the ground layer component with increasing stocking rate, whereas effect sizes below 0 represent reductions. Values above 0 on the y-axis represent an increase in cover of the ground layer component with increasing time since the last burn, whereas effect sizes below 0 represent reductions. Values further from 0 represent larger effect sizes than values closer to 0. 196
Figure 6.1-4: Effect of stocking rate over the 4-year study period and time since last burn on the cover of each taxa. Parameter estimates are shown with 95% credible intervals. Parameter estimates have been sampled from the posterior distribution of a Bayesian zero-inflated beta regression model. Values above 0 represent an increase in cover of the ground layer component with increasing stocking rate or time since burning, whereas parameter estimates below 0 represent reductions. Values further from 0 represent larger effect sizes than values closer to 0. 197
Figure 6.1-5: Effect sizes of stocking rate and time since the last burn on the species richness of growth form categories. The central point (dark blue) shows the median effect size and lines (light

blue) show 95% credible intervals of the median. All parameter estimates have been calculated from the posterior distribution of a Bayesian Poisson regression model. Values above 0 on the x-axis represent an increase in species richness of the growth form category with increasing stocking rate, whereas effect sizes below 0 represent reductions. Values above 0 on the y-axis represent an increase in species richness of the growth form category with increasing time since the last burn, whereas effect sizes below 0 represent reductions. Values further from 0 represent larger effect sizes than values closer to 0. 198
Figure 6.2-1: Ecological effects of the provision of artificial water sources and potential impacts on SBTF 204
Figure 6.2-2: Location of 1 km transects carried out during the study 206
Figure 6.2-3: Raised water trough (left) and cattle trough (right) installed within the study area 208
Figure 6.2-4: Location of artificial water sources monitored during the study 209
Figure 6.2-5: Abundance of birds recorded during 1 km transects in relation to their distance to permanent water. Negative values on the left represent species that have higher abundances near permanent water compared to afar. A value of 0 (red dotted line) represents birds that were recorded approximately randomly through the landscape in relation to the distance to permanent water. 211
Figure 6.2-6: SBTF detection rates at water sources monitored by remote cameras 212
Figure 6.2-7: Percentage of species records from raised water sources 214
Figure 6.2-8: The five species which had the highest percentage of records for each raised water source 215
Figure 6.3-1: Water sources that restrict southern black-throated finch access. Thick edged concrete troughs (left) and no tree cover near the water's edge (right). 219
Figure 6.3-2: Examples of water trough retrofits that are southern black-throated finch friendly. 220
Figure 7.1-1 Survey effort by method through the 12 years of monitoring. Survey periods where survey effort was not recorded have been excluded. 222
Figure 7.1-2 The average number of SBTF recorded per survey for monitoring methods utilised during the monitoring program 226
Figure 7.2-1 Study area 242
Figure 7.2-2 Properties surveyed 247
Figure 7.2-3: Regrowth vegetation on Fleetwood (left) and Jerico subregion (right) 249
Figure 7.2-4: Ground layers impacted by overgrazing on Wambiana (left) and Jerico subregion (right) 250
Figure 7.2-5: Grazing piospheres surrounding a trough on Longton (left) and a dam on Maynard (right) 251
Figure 7.2-6: Results of fire exclusion recorded during the survey, including build-up of biomass in Jerico subregion (left) and woody shrub encroachment due to overgrazing on Longton (right) 252
Figure 7.3-1: Water sample collection method at CWAT99 256
Figure 7.3-2: Environmental DNA sampling locations 258
Figure 7.3-3:Water sources where eDNA sampling detected use by southern black-throated finch, including TT01 (left) and CWAT99 (right) 259
Figure 7.3-4:Water sources that were in consistent use by southern black-throated finch but had a negative detection result using eDNA, including CWAT301 (left) and CWAT302 (right) 259
Figure 7.4-1:Spatial distribution of acoustic recording units within the study area. Filled and hollow symbols represent recordings used for training and testing data, respectively. Recorders were placed within remnant woodlands within the study area, focussing on areas that comprised broadly suitable habitat for southern black-throated finch. 265
Figure 7.4-2: Example audio frame following data pre-processing steps that was used as input into the convolutional neural network. The audio frame shows a SBTF call with a dominant frequency (stronger signal) and a fundamental harmonic (weaker signal) included in the frame. 266

Figure 7.4-3: CNN model development pipeline that incorporates active learning iterations to develop a model for identifying calls of the southern black-throated finch. 267
Figure 7.4-4: Evaluation performance of each model iteration. Error bars show 95% confidence intervals. 271
Figure 7.4-5: Pairwise comparison of inter-annotator agreement among the primary annotator (PA), experts one and two (E1 and E2) and the final model (FM). Error bars show 95% confidence intervals. 272
Figure 7.4-6: Comparison of agreement between the primary annotator and the final model when the evaluation process has allowed 'uncertain' audio frames (a), compared to when 'uncertain' audio frames, as assessed by the primary annotator, have been excluded (b). Error bars show 95% confidence intervals. Confusion matrixes are presented below each option as heat plots. The heat plot shading and numbers represent the count of audio frames labelled in each class by the primary annotator compared to the final model. 273
Figure 7.4-7: Agreement among all annotators, including the primary annotator and two experts, for audio frames captured using solar-powered bioacoustic recorders (Frontier Labs) and Audiomoths (Hill *et al.*, 2019). Error bars show 95% confidence intervals. 274
Figure 7.5-1: Localisation methods for automated radio telemetry systems, which estimate location by: A) finding the intersection of distance estimates that are inferred from relative signal strengths (RSSs) on omni-directional receivers; B) finding the intersection of multiple angles of arrival (AOA) produced by directional receivers; C) using the AOA and distance estimates inferred from the RSSs of directional receivers; and D) developing a radio fingerprint map of each receivers' capture area (denoted by the colours red, green and blue), using ground-truthed RSS data. 280
Figure 7.5-2: Map of the study area showing the locations of automated radio towers (n = 27) and radio tracked southern black-throated finch (SBTF) (n = 232), which were used for training and testing the localisation methods. 282
Figure 7.5-3: Automated radio telemetry receiver with four directional antennas installed within the study area 283
Figure 7.5-4: Data pipelines for the training of the location fingerprinting model (A) and inference of transmitter location (B). 285
Figure 7.5-5: Comparison of the mean positional errors of each localisation method. Bars show 95% confidence intervals. 291
Figure 7.5-6: Effect of four covariates on the positional error of localisations. The relationships depicted include the: (A) mean distance to the receiver; (B) mean relative signal strength (RSS) of the detections; (C) pulse interval of the transmitter; and (D) number of receivers on which the signal was recorded. All figures show the mean estimate with 95% confidence intervals. A random jitter has been applied to the points of (D) to aid data visualisation. 292
Figure 7.5-7: Interaction effects between localisation method and the: (A) pulse interval; (B) mean distance to the receiver; and (C) mean relative signal strength (RSS) of the detections. All figures show the mean estimate with 95% confidence intervals. 293
Figure 7.5-8: Positional error of localisation estimates compared to the number of samples used to train the model. Bars show the standard deviation around the mean positional error. The y-axis scale is the same for both plots and does not start at zero. 294
Figure 8.4-1: Conceptual model of SBTF response to habitat management factors (blue) and habitat attributes (green) 302
Figure 9.2-1: Bioacoustic monitoring and water source design 324

Appendices

Appendix A Compliance with approval requirements
Appendix B Assay optimisation and eDNA analysis for black-throated finch

Definitions

Term	Definition
The Project	The Carmichael Coal Mine Project
Project Area	The Black-throated Finch Research Program study area, including areas of the Mining Lease and the Moray Downs West offset area (Figure 1.1-1).
Study area	The Black-throated Finch Research Program includes many research components, arranged into Sections 3 to 8 of this report. The 'study area' is specific to each research component and typically covers the spatial extent of data included in that component. For example, the 'study area' of the landscape genetics research (Section 3.2) has been expanded outside of the Project Area to incorporate genetic sampling on properties adjacent to the Project Area.

Abbreviations

Abbreviation	Description
Bravus	Bravus Mining and Resources
BTFMP	Black-throated Finch Management Plan (EcoLogical Australia, 2023)
BTFRP	Black-throated Finch Research Program (EcoLogical Australia, 2023)
DETSI	The Department of Environment, Tourism, Science and Innovation (DETSI) , formerly the Department of Environment and Science (DES) and the Department of Environment and Heritage Protection (DEHP)
E2M	E2M Pty Ltd
The EA	Environmental Authority (EA) EPML01470513
ML	Mining Lease
MDWOAMP	Moray Downs West Offset Area Management Plan (CO2 Australia, 2019)
RE	Regional Ecosystem
SBTF	Southern black-throated finch (*Poephila cincta cincta*)
SE	Standard error of the mean is a measure of how far the estimated mean is likely to be from the true value. Standard error is calculated as the standard deviation of a sample (σ) divided by the square root of the number of samples (n): $\sigma/\sqrt{n}$
sp.	Singular species. For example, *Eucalyptus* sp. refers to a single species of *Eucalyptus*
spp.	Multiple species. For example, *Eucalyptus* spp. refers to multiple species of *Eucalyptus*

Summary

Bravus Mining and Resources (Bravus) is constructing the Carmichael Coal Mine (the Project), located in the Galilee Basin, approximately 160 km north-west of Clermont, Queensland. As part of their State and Commonwealth Government environmental approvals for the Project, Bravus were conditioned to undertake a Research Program on the southern black-throated finch (SBTF; *Poephila cincta cincta*). This report presents the results of the Research Program, which is the culmination of five years of research and collaboration between Bravus, E2M Pty Ltd (E2M) and Griffith University.

Broadly, the Research Program aimed to determine the critical ecological requirements for BTF within the Project Area and use this understanding to inform management and mitigation measures for the Project. The key findings and recommendations are summarised below.

Home range and movement ecology

Radio-tracking, leg band resighting and recapture, and genetic analyses were used to examine the home ranges and movement ecology of SBTF within the Project Area. The results from all methods identified a high site philopatry among SBTF. Short-term movements (less than one month), assessed through radio-tracking, found few landscape-scale movements of more than 4.5 km. Long-term movements (up to three years) were identified through leg band resightings and recaptures, only among areas of interconnected habitat. Despite the high site philopatry of SBTF, we identified large home ranges (mean 319 ha), through which they move daily to meet basis to meet their foraging, water and shelter requirements.

Genetic structure

Genetic analyses of 158 individuals identified a fine-scale population structure, with restricted gene flow between northern and southern subpopulations that are separated by approximately 16 km. The South subpopulation exhibited lower effective migration rates and distinct genetic characteristics. While effective population size estimates ranged from 940 to 1,440 overall, the South subpopulation estimate ranged from 290 to 360, which is approaching the minimum threshold required to maintain long-term evolutionary potential. Demographic modelling revealed two historical bottlenecks, approximately 60–100 and 700 years ago, which are likely to have contributed to range contraction and potentially genetic isolation of the South subpopulation within the Project Area.

Habitat use

At a landscape scale, radio-tracking found that SBTF habitat use was strongly associated with eucalypt-dominated grassy woodlands, particularly those featuring *Eucalyptus melanophloia*. Habitat use was closely linked to the proximity to permanent water sources, with the majority of records occurring with 3 km of permanent water. SBTF avoided non-remnant vegetation dominated by exotic grasses such as *Cenchrus ciliaris*, which lack suitable forage and nesting structures.

At a local scale, SBTF foraging observations (n = 447) were concentrated in patches of *Eucalyptus melanophloia* woodlands that had high vegetation productivity (NDVI), native grass richness, bare ground, and cover of key grass species, with minimal shrub and weed cover. These conditions optimise energy intake and accessibility of grass seeds while minimising foraging effort and predation risk.

Nests (n = 175) that were observed incidentally and through radio tracking, were primarily found in *Eucalyptus melanophloia* and *Eucalyptus similis* savanna woodlands with sparse canopy and shrub layers. Within these woodlands, nests were placed predominantly above 2 m in *Acacia sericophylla* (58%), with *Eucalyptus melanophloia*, *Melaleuca nervosa* and *Bursaria incana* also commonly used. Similar to their selection of foraging habitat, nests were preferentially placed in areas of high vegetation productivity (NDVI), low shrub cover, a mix of grass and bare ground and often situated further from water sources. Nests were well concealed, likely as a strategy to reduce predation risk.

Dietary requirements

DNA analysis of crop and faecal samples from 214 individuals identified 46 plant taxa in the diet. However, a strong preference was evident for a small subset of high-quality grass species which offer higher metabolisable energy, crude protein, and reduced handling time, particularly *Digitaria*, *Paspalidium* and *Panicum* species. These species were most favoured during the wet season when seed availability peaks. However, as seed availability declines through the dry season, SBTF increasingly consume more common but less nutritious grasses, including *Themeda triandra*, *Heteropogon contortus* and *Aristida* species. This seasonal shift in diet, combined with the species movement ecology, of a large home range with high site philopatry, suggests that SBTF respond to seasonal resource scarcity with dietary diversification and tracking of high-foraging efficiency areas across their home range.

A high richness of grass species, particularly of species bearing nutritious seeds, is an important habitat requirement for SBTF. High grass species richness affords phenological diversity, staggering the seed production among grass species with different strategies, including early-seeding perennials, late-seeding perennials, annuals and a mast-seeding species. Phenological diversity is likely to be further increased by heterogenous vegetation communities and fire histories, supporting year-round foraging by the SBTF.

Management strategies for SBTF habitat, particularly regarding fire, grazing and water.

The Research Program included a four-year trial of the effects of grazing and fire on ground layer vegetation composition and species richness across the 75,000 ha Project Area. Over four years, 900 vegetation plots were undertaken across 239 sites stratified by fire history and stocking rates. The results demonstrated that higher stocking rates were associated with increased cover and richness of non-native graminoids, particularly *Cenchrus ciliaris*, and a decline in native perennial grasses such as *Themeda triandra* and *Heteropogon contortus*. In contrast, recent burning promoted native annual graminoids and forbs, while long-unburnt sites favoured exotic and perennial species. Controlled low-intensity burns, led by First Nations rangers, were effective in enhancing native ground layer diversity and suppressing invasive species. The findings support integrated fire and grazing management strategies, specifically, low stocking rates combined with mosaic burning, which maintain ecological integrity and improve the conservation value of SBTF habitat.

Within the Project Area, SBTF rely heavily on artificial water sources, particularly in the dry season, when ephemeral water sources are unavailable. The Research Program trialled an additional six raised water sources, which were designed to provide accessible water while minimising grazing impacts and predator presence. We found that three of the six were used, but only in areas that were known to comprise existing SBTF foraging habitat. Despite this partial success, additional water sources should be considered judiciously due to potential threats imposed by new water sources, including increased grazing pressure, increased abundance of aggressive and predatory species, and disease transmission risks. Design of new water sources should promote use by SBTF while mitigating these risks carefully, with features that include:

- Narrow perching edges (suitable for small passerines)
- Reliable water levels, with adequate flushing to reduce disease transmission risks
- Exclusion fencing to prevent access to herbivorous and predatory species (cattle, sheep, macropods, European rabbits, hares, pigs, cats, foxes, birds of prey and corvids); and
- Placed near (within 5 m) of shelter, particularly tree cover, which SBTF preferentially use for predator avoidance over shrubs.

Recommendations for the management of SBTF habitat

There are three pillars of habitat management through which the majority of SBTF habitat quality maintenance and improvement can be achieved:

- Low to no stocking rates
- A burning program of low-intensity patchy burns; and
- Access to reliable water sources.

These habitat management factors promote habitat attributes preferred by SBTF, namely:

- Open grassy woodlands, with a tussock grass understorey
- A rich diversity of native grass species
- Low shrub cover
- Patches within the ground cover that promote foraging efficiency; and
- Accessible water sources.

Additional interventions may be required where fire, grazing and water source regimes are unable to promote SBTF habitat suitability, such as in highly degraded areas where rehabilitation is required due to:

- Land clearing
- Dense infestations of exotic plants (e.g. buffel grass or stylo); and
- Heavily modified ground layers that contain few native grass species.

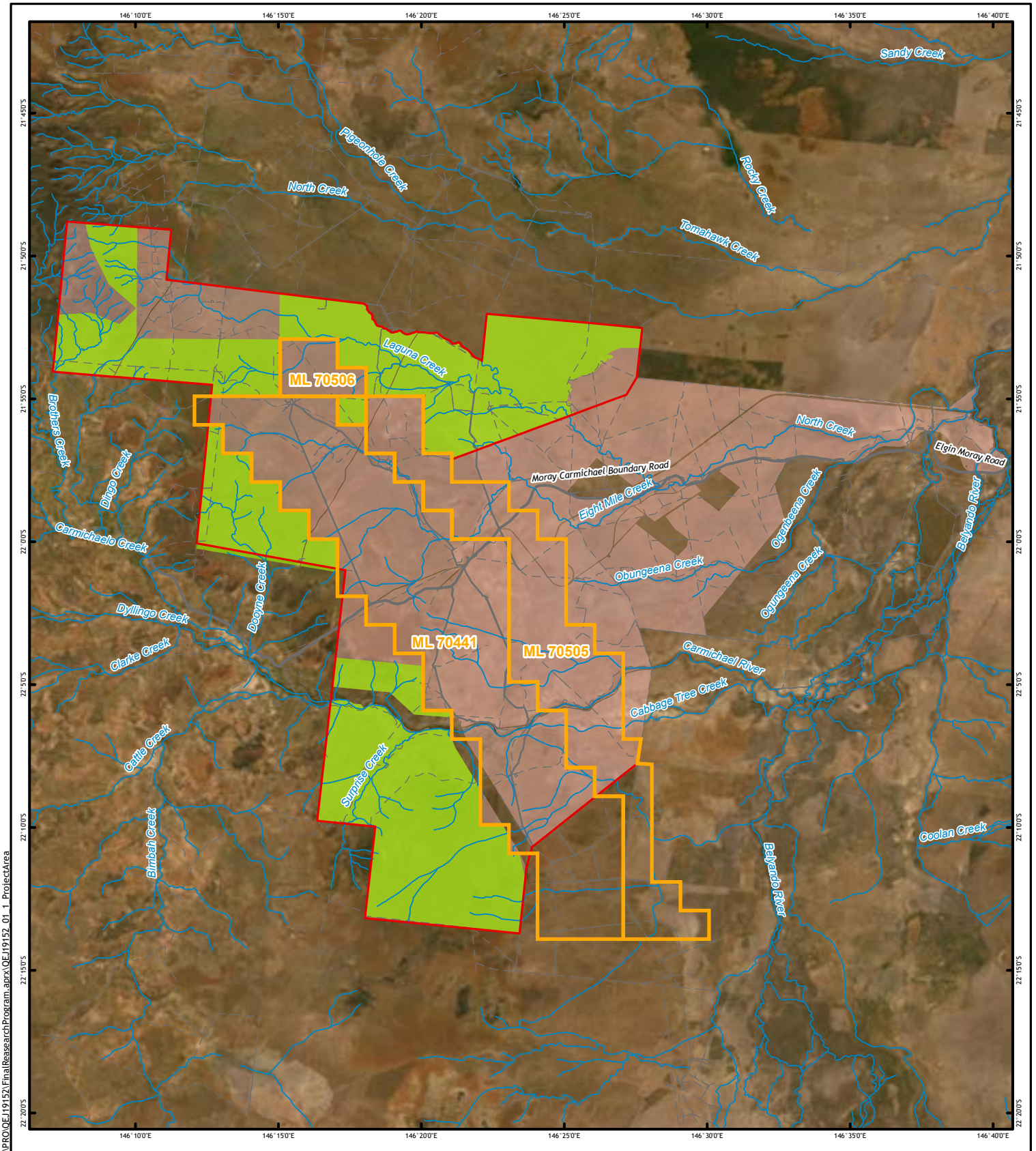
1 Study background

1.1 Project background

Bravus Mining and Resources (Bravus) is developing the Carmichael Coal Mine, herein referred to as 'the Project', which was granted conditional approval by the Queensland Coordinator-General on 8 May 2014 (Environmental Authority [EA] EPML01470513) and the Commonwealth Minister for the Environment on 14 October 2015 (EPBC 2010/5736).

The EA and *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act) approval conditions contain numerous requirements relating to management, monitoring and research of the southern black-throated finch (SBTF) (*Poephila cincta cincta*) that occur within the Mining Leases and Moray Downs West offset area, herein referred to as the 'Project Area' (Figure 1.1-1). Moray Downs West was identified as a priority offset area to fulfill the majority of the Project's offset obligations. Located outside the mining lease boundary, this area is specifically managed to protect and conserve matters of national environmental significance and matters of state environmental significance, ensuring compliance with regulatory requirements.

Commonwealth and State approval conditions also required the development of a Black-throated Finch Research Program (BTFRP) as a part of the Black-throated Finch Management Plan (BTFMP) for the Carmichael Coal Mine (EcoLogical Australia, 2023). The BTFRP aims to improve knowledge of the ecological requirements of SBTF populations that may be directly and indirectly impacted by the Project. The BTFRP Project Plan (Adani, 2020) was approved by the Department of Environment and Science (DES) on 10 February 2020 and identifies the research activities and milestones to achieve the requirements of the EA conditions and the approved BTFRP. As outlined in the BTFMP, independent reviews of BTFRP deliverables are required to provide further scientific rigour and inform the adoption of research findings into ongoing management practices.



Legend

- Major Road
- Minor Road
- Track
- Watercourse
- Mining Lease
- Project Area
- Moray Downs Wet Offset Area
- Moray Downs

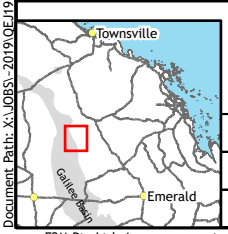
N

Scale 1:335,000 (A4)

0 5 10 15

Kilometres

Coordinate System: GDA2020



Notes:
Aerial Imagery: © ESRI 2021
Cadastre: © DNRME 2021
Watercourse: © Geoscience Australia 2012
Road: © DNRME 2021

Rev	Description	Drawn	Approved	Date
2	Issued for Use	JY	JMC	30/10/2025
0	Issued for Use	GO	JMC	08/08/2025

FIGURE 1.1-1: PROJECT AREA

BTFRP Final Report
Bravus

Map Number	Job Number	Rev
1 of 1	QEJ19152	2

Document Path: X:\JOBS\-2019\QEJ19152\GIS\Final Research Program Report\PRO\QEJ19152\FinalReasearchProgram.aprx\QEJ19152_01_1_ProjectArea

E2M Pty Ltd gives no warranty in relation to the data (including accuracy, reliability, completeness or suitability) and accepts no liability for any loss, damage or costs (including consequential damage) relating to any use of the data in this map.

1.2 Research objectives

The EA (EPML01470513), BTFRP (EcoLogical Australia, 2020) and BTFRP Project Plan (Adani, 2020) established the aims and objectives of the research presented in this document. The broad objectives of the research as specified within the BTFRP are copied below:

> *"The objective of the Research Program is linked to conditions of approvals and involves determining the critical ecological requirements for the BTF. The research is focussed on the spatial and temporal movement patterns, habitat requirements and population dynamics, dietary requirements, home range and nesting requirements for the BTF as well as the relationship between water sources, woody habitat and food sources. The research will also aim to determine the species response to grazing management, fire management and water body locations within the mining lease area and approved offset areas, to facilitate appropriate management of these areas.*
>
> *The results of the research will inform and improve management and mitigation measures adopted for the Project to minimise impacts to the local population of BTF as well as enhance habitat within the Moray Downs West offset area in such a way to increase the carrying capacity for BTF in these areas. The Research Program will establish whether Ten Mile Bore and surrounding areas are high value habitat for BTF and establish management actions to maintain the current BTF population in this area."* (EcoLogical Australia, 2020)

Appendix A identifies all approval conditions and objectives that are required to be addressed by the BTFRP and details how they are addressed within this report.

1.3 Structure of this report

The BTFRP has four approved aims, which include:

- Aim 1: Estimate the home range and movements of SBTF in the Project Area
- Aim 2: Determine the foraging and breeding habitat requirements and the relationship with vegetation structure and composition and water
- Aim 3: Determine the dietary requirements for SBTF in the Project Area; and
- Aim 4: Identify management strategies for SBTF habitat in the Project Area regarding fire, grazing and water.

The structure of this report follows these four aims, with subsections addressing the objectives that comprise each aim (EcoLogical Australia, 2020). In addition, this report includes: a general introduction to the study's background, an overview of the research program design and concludes with recommendations for the ongoing management and monitoring of SBTF.

Sections of this report have been prepared into scientific papers that have/will be submitted to peer reviewed scientific journals. All papers have been published as open access under creative commons licence CC BY 4.0, which allows the sharing and adaptation of this work, provided appropriate attribution is given. Furthermore, Doctor of Philosophy theses are being completed by Brad Dreis and John van Osta through Griffith University. All scientific papers and manuscripts have been reproduced within the report to afford consistent formatting, style and a single reference point for the BTFRP outcomes. However, some information has been repeated among sections of this report to allow these sections to be submitted to scientific journals as standalone publications.

A conceptual diagram of the structure of this report is provided in Figure 1.3-1.

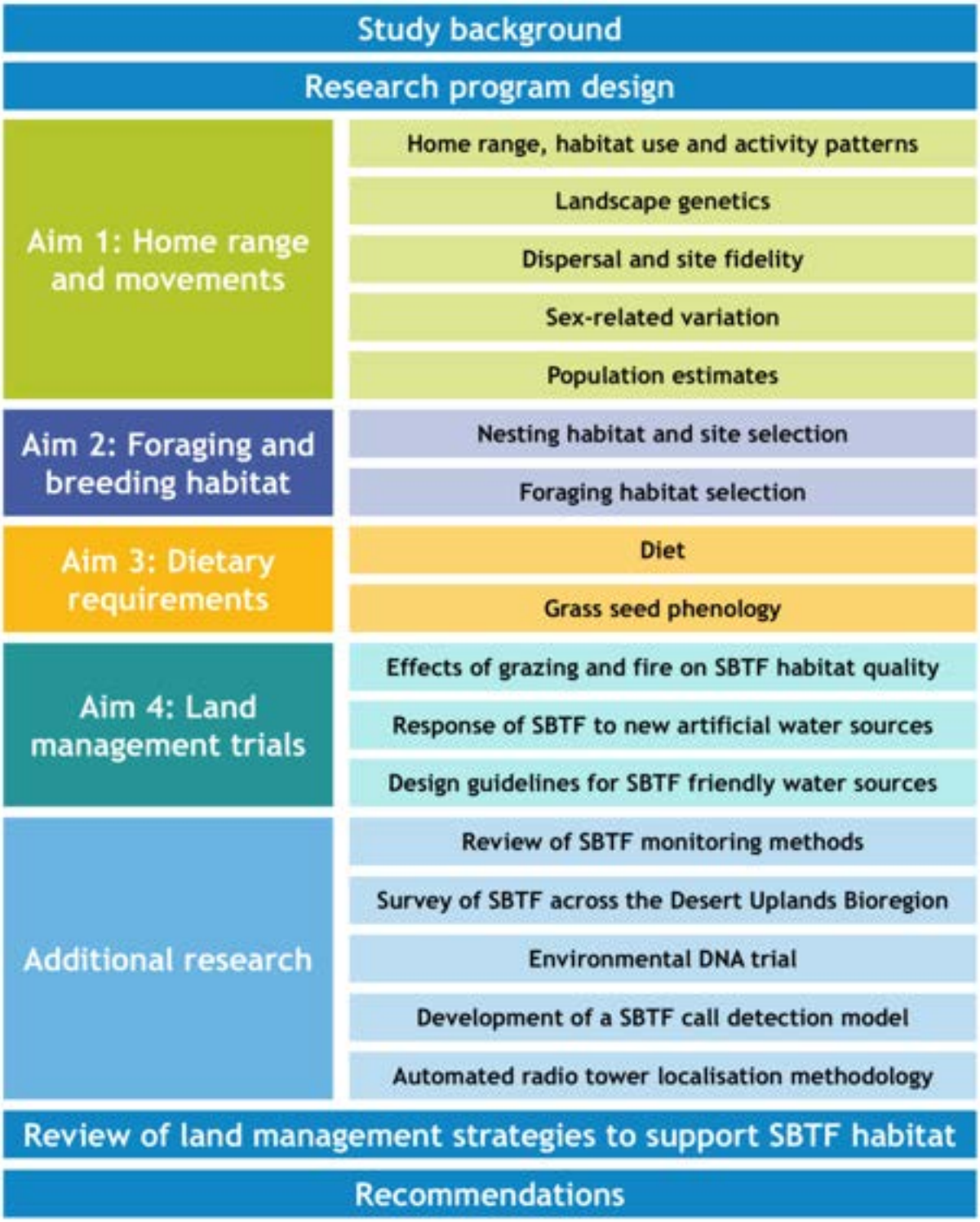


Figure 1.3-1: Conceptual diagram of the structure of this report

1.4 Southern black-throated finch species profile

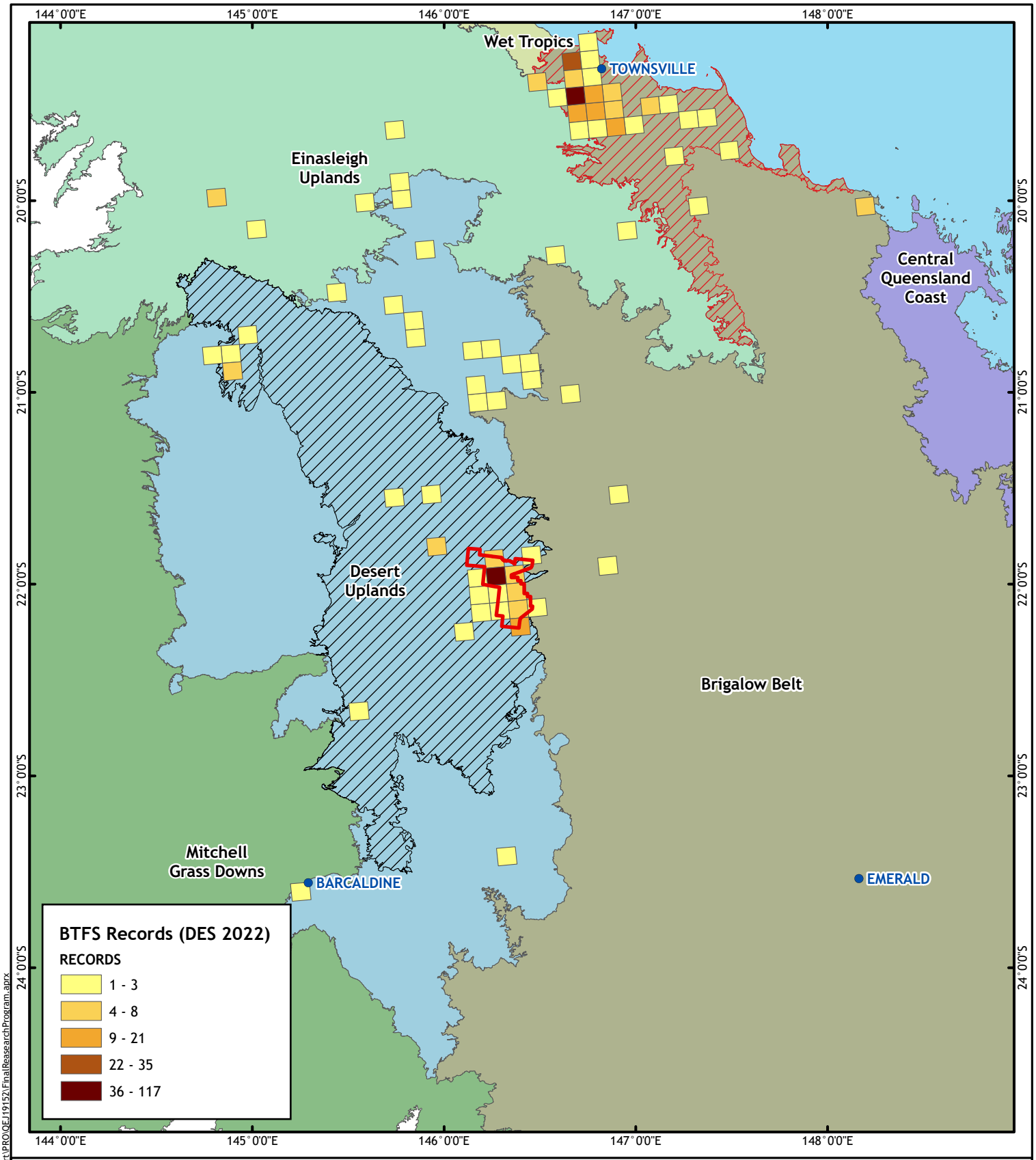
The SBTF, *Poephila cincta cincta*, is a grassfinch species (family Estrildidae), endemic to north-eastern Australia (Higgins, Peter and Cowling, 2006). The SBTF inhabits open grassy woodlands, within which the species primarily feeds on the seeds of native and introduced grass species (Higgins, Peter and Cowling, 2006; Mula Laguna *et al.*, 2019; Williams *et al.*, 2020).

The SBTF is listed as Endangered under the Commonwealth EBPC Act, with the species having undergone an extensive range contraction of over 80% from its historical distribution (Department of Agriculture, Water and the Environment (DAWE), 2023). The spread of pastoralism and associated changes to land management practices is the primary driver of the species' decline, with associated impacts including clearing or degradation of the species' preferred habitat, altered fire regimes, changes to grazing pressure and the spread of introduced grasses (Mula Laguna *et al.*, 2019; DAWE, 2023). The range and extent of SBTF has declined more severely than most other granivorous birds in Australia, despite experiencing similar pressures from land condition changes caused by pastoralism in Australia's savannas and rangelands (Franklin, 1999; Franklin *et al.*, 2005; Garnett, Szabo and Dutson, 2011).

Historically, SBTF occurred from the Inverell-Ashford district in north-eastern New South Wales, with outlying records south to Gilgandra, through eastern Queensland to a hybridisation zone with the northern subspecies *Poephila cincta atropygialis* at the headwaters of the Burdekin and Lynd River catchments (Ford, 1986a; A. Ley and Cook, 2001; Mula Laguna *et al.*, 2019). Post-2000 sightings are concentrated in two remaining strongholds, namely the Townsville Coastal Plains biogeographic subregion and the Desert Uplands Biogeographic Region, with sporadic sightings in the Undara-Toomba Basalts subregion of the Einasleigh Uplands Bioregion (Vanderduys *et al.*, 2016; Mula Laguna *et al.*, 2019; DAWE, 2023).

Previous research on the SBTF has primarily concentrated on the Townsville Coastal Plains population, while published studies from the Desert Uplands are scarce (Mula Laguna *et al.*, 2019). This disparity in research effort could be attributed to two key factors: the Desert Uplands are relatively inaccessible compared to the Townsville Coastal Plains, and the land tenure of SBTF habitats differs significantly in these regions. In the Desert Uplands, the majority of SBTF habitats are situated on leasehold and freehold lands used for grazing, which pose additional challenges for gaining research access and implementing conservation actions (Vanderduys *et al.*, 2016; DAWE, 2023).

The two stronghold populations of the SBTF are situated in regions with substantially different climate and habitat characteristics, which may lead to unique habitat requirements and movement patterns critical for the species' conservation (Mula Laguna *et al.*, 2019). The Townsville Coastal Plains is within the tropical savanna with a dry winter Köppen-Geiger climate zone, while the Desert Uplands is within the hot semi-arid climate zone (Peel, Finlayson and McMahon, 2007). Other differences between these regions include limited overlap in vegetation communities (DESI, 2021); fewer freshwater wetlands and artificial dams in the Desert Uplands compared to the Townsville Costal Plains (DES, 2013c); and reduced fragmentation of woody vegetation due to less land clearing in the Desert Uplands (Sattler and Williams, 1999). These differences highlight the need for region-specific conservation strategies to effectively support each SBTF population.



Legend

- Study Area
- **Sub-bioregion**
  - Alice Tableland
  - Townsville Plains
- **Bioregion**
  - Brigalow Belt
  - Central Queensland Coast
  - Desert Uplands
  - Einasleigh Uplands
  - Mitchell Grass Downs
  - Wet Tropics
  - Ocean

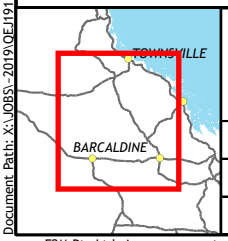
N

Scale 1:3,000,000 (A4)

0 50 100 150

Kilometers

Coordinate System: GCS GDA 1994



Notes:
Aerial Imagery: © ESRI 2021
Cadastre: © DNRME 2021
Watercourse: © Geoscience Australia 2012
Road: © DNRME 2021

Rev	Description	Drawn	Approved	Date
2	Issued for Use	JY	JMC	30/10/2025
0	Issued for Use	GO	JMC	08/08/2025

FIGURE 1.4-1: REGIONAL CONTEXT

BTFRP Final Report
Bravus

Map Number	Job Number	Rev
1 of 1	QEJ19152	2

Document Path: X:\JOBS\-2019\QEJ19152\GIS\Final Research Program Report\PRO\QEJ19152\FinalReasearchProgram.aprx

E2M Pty Ltd gives no warranty in relation to the data (including accuracy, reliability, completeness or suitability) and accepts no liability for any loss, damage or costs (including consequential damage) relating to any use of the data in this map.

2 Research project design

2.1 Project Area description

This study was conducted in a 75,000 ha section of the Moray Downs property, which is located 300 km south of Townsville and 250 km north-west of Emerald in Queensland, Australia (Figure 1.4-1). The Project Area is within the Alice Tableland subregion of the Desert Uplands and includes the largest aggregation of contemporary SBTF records (Figure 1.4-1; DAWE, 2023). Land use within the Project Area comprises a combination of low-intensity cattle grazing, and native vegetation managed as an environmental offset under the National *Environment Protection and Biodiversity Conservation Act 1999* (Figure 2.1-1). The Project Area includes a portion of the Carmichael Coal Mine mining lease (Figure 2.1-1).

The majority of the Project Area (84%) is covered by remnant vegetation (Figure 2.1-1). Eucalypt woodlands represent 71% of these remnants, predominantly featuring communities dominated by *Eucalyptus melanophloia* (39%) and *E. similis* (22%). Additional habitats include woodlands and shrublands dominated by *Acacia* and *Melaleuca* species. Areas not classified as remnant vegetation constitute 16% of the Project Area, largely cleared for uses such as grazing, mining infrastructure and roads.

One major watercourse, the Carmichael River, traverses the Project Area as well as the smaller North Creek (Figure 2.1-1). Artificial bores and dams have been established within the Project Area to support cattle grazing (Figure 2.1-1).

Much of the Desert Uplands Bioregion has a history of low-intensity cattle grazing since European settlement of the area in the mid-1800s (Paterson, 2005). This historically low grazing intensity can be attributed to the large property sizes with expansive paddocks, limited permanent water sources that create areas inaccessible to livestock, and the presence of *Gastrolobium grandiflorum* (heart-leaf poison bush), a toxic plant that has discouraged intensive grazing (Kutt and Woinarski, 2007). The Project Area is entirely located within a leasehold property, which prior to establishment of mining activities, was entirely used for grazing (EcoLogical Australia, 2023), consistent with the historical land use patterns of the Desert Uplands. From 2019 to present the maximum stocking rates for the Project Area were capped to the stocking rates specified in the BTF Management Plan (EcoLogical Australia, 2023) and Moray Downs West Offset Area Management Plan (CO2 Australia, 2019), which in SBTF habitat range from a maximum stocking rate of 25 ha/Animal Equivalent (AE) to 10 ha/AE. These stocking rates are lower than, or comparable to, the region's typical light to moderate rates of approximately 8–10 ha/AE (McIvor, 2012).

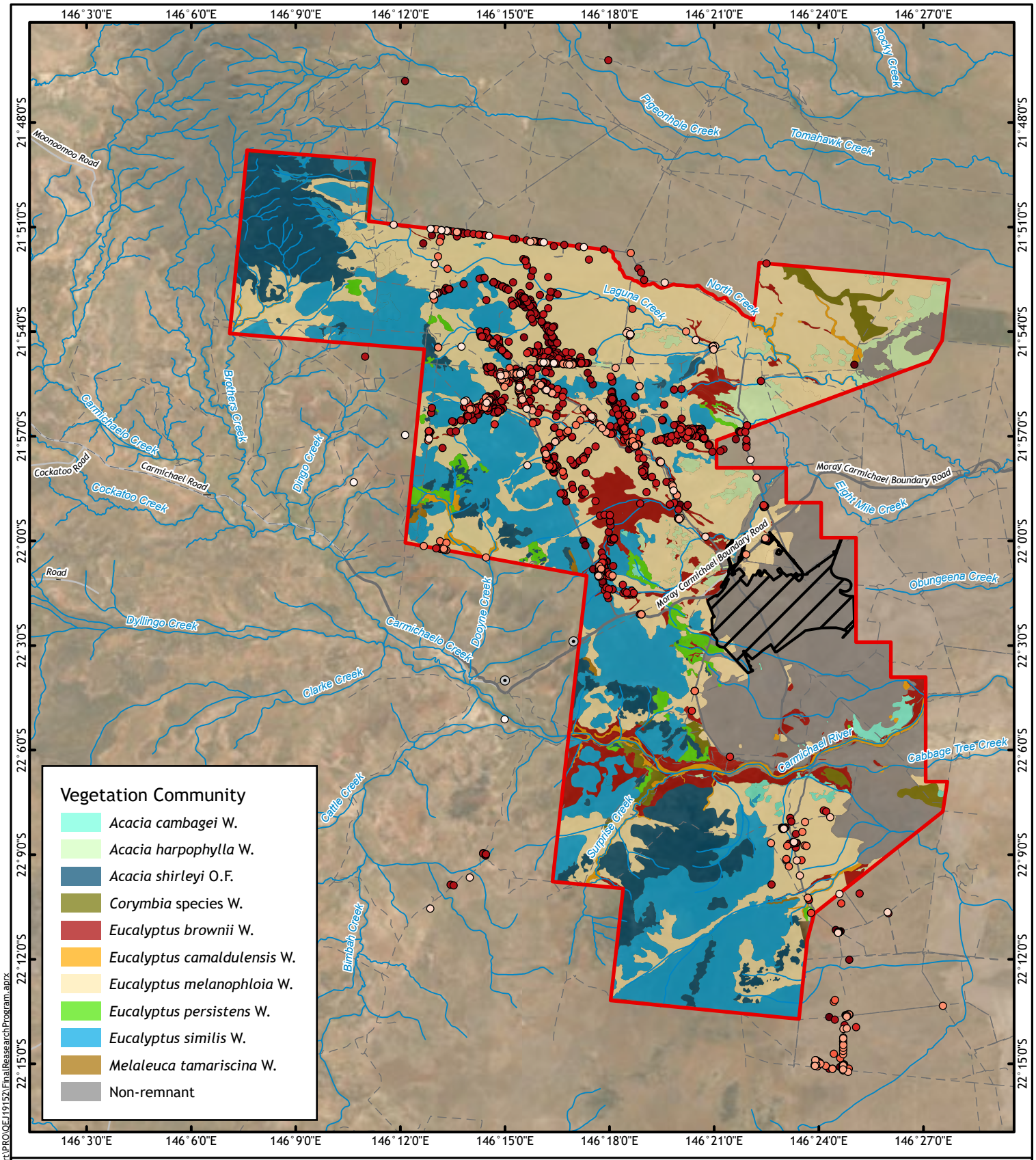
Fire is a common feature of woodlands in the Desert Uplands, including both controlled and uncontrolled burns (Paton and Tyler, 2017). The majority of vegetation within the Project Area comprises fire-adapted eucalypt dominated woodlands (Burrows *et al.*, 2010), which are recommended to be burnt every 5 to 10 years (Queensland Herbarium, 2024). Prior to the establishment of the BTFRP and implementation of an ecological and cultural burning program, the Project Area had a mosaic of fire history. Prior to 2011, fires were common across the woodlands within the Project Area, with most eucalypt-dominated woodlands having burned between three and five times over the preceding 20-year period (Figure 2.1-3). However, from 2011 to 2022, fire was largely excluded from the Project Area due to changed management (Figure 2.1-2; WES, 2021). As a result, at the commencement of the BTFRP, much of the Project Area had been long-unburnt, with high ground layer fuel loads. The early focus of the ecological and cultural burning program was therefore to regain a mosaic of fire histories (WES, 2021).

A species distribution model was developed in 2020 to assist in the stratification of survey sites that were systematically placed across the Project Area. These included the stratification of survey sites for habitat and phenology assessments and 1 km transects. The species distribution model was developed using MaxEnt, which is a widely used method for presence only data (Elith, Phillips, Hastie, Dudik, *et al.*, 2011).

The species distribution model used SBTF records sourced from data collected for the Carmichael Coal Mine Project between 2011 and 2020 to predict the distribution of SBTF habitat within the Project Area based on correlation with environmental variables. The environmental variables included within the species distribution model included:

- Climatic factors (annual mean temperature, temperature seasonality, maximum temperature of warmest month, minimum temperature of coldest month, annual precipitation, precipitation seasonality, precipitation of wettest quarter and precipitation of direst quarter)
- Topographic factors (digital elevation model)
- Vegetation characteristics (regional ecosystem)
- Fire regimes (cumulative fire scars and time since last burn); and
- Water availability (distance to perennial water sources and distance to all water sources).

The result of the species distribution model, which were used to inform the initial planning of research design, is presented in Figure 2.1-4.



Legend

Year: No date, 2011, 2012, 2013, 2014, 2015, 2016, 2017, 2018, 2019, 2020, 2021, 2022, 2023, 2024

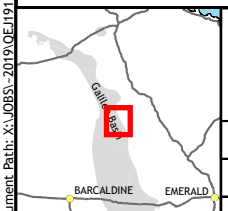
Project Area
Mining Infrastructure
Watercourse
Major Road
Minor Road
Track

N

Scale 1:275,000 (A4)

0 5 10
Kilometers

Coordinate System: GDA2020



Notes:
Aerial Imagery: © ESRI 2021
Cadastre: © DNRME 2021
Watercourse: © Geoscience Australia 2012
Road: © DNRME 2021

Rev	Description	Drawn	Approved	Date
2	Issued for Use	JY	JMC	30/10/2025
0	Issued for Use	GO	JMC	08/08/2025

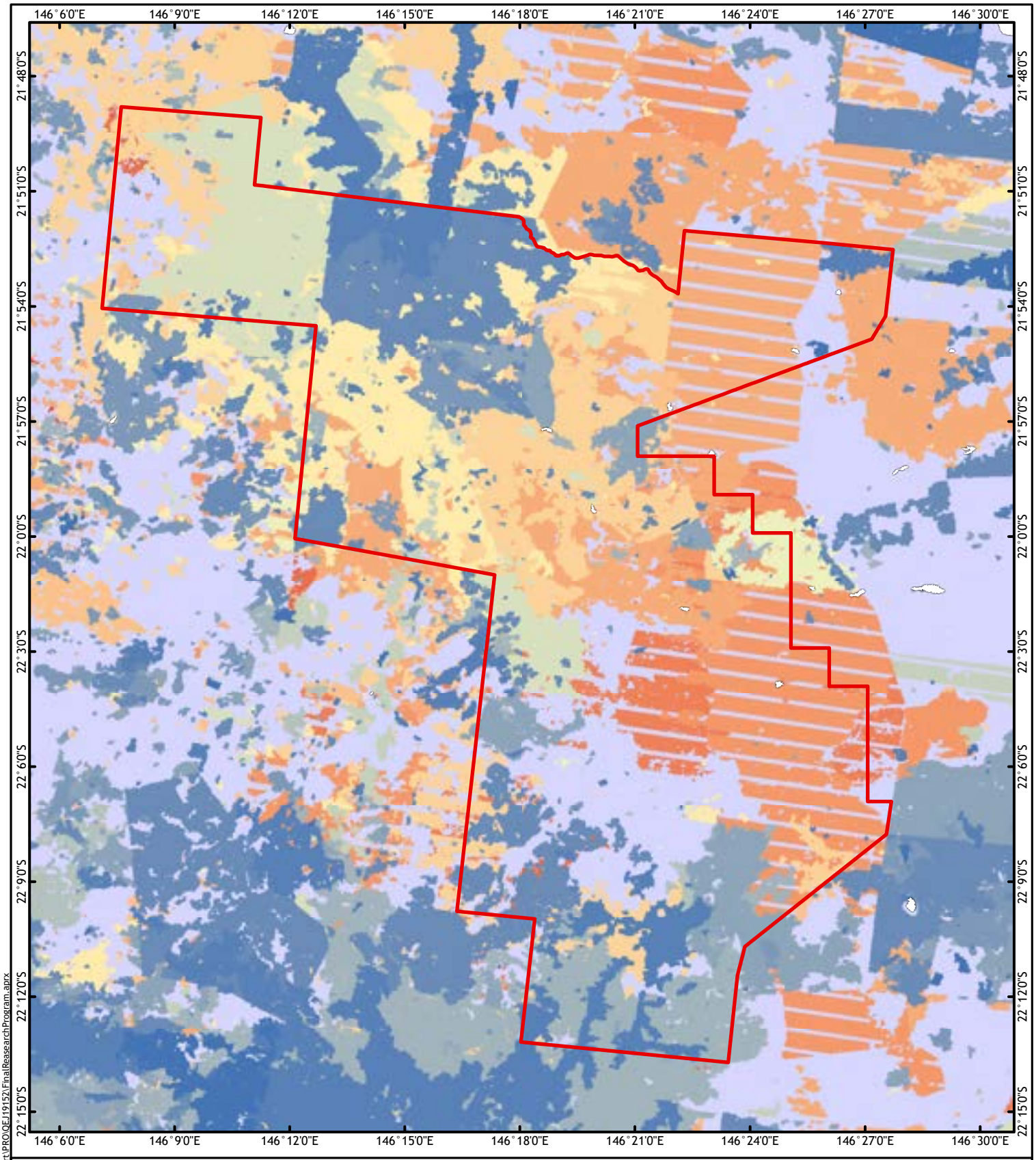
FIGURE 2.1-1: SBTF RECORDS AND VEGETATION COMMUNITIES

BTFRP Final Report
Bravus

Map Number	Job Number	Rev
1 of 1	QEJ19152	2

Document Path: X:\JOBS\-2019\QEJ19152\GIS\Final Research Program Report\PRO\QEJ19152\FinalReasearchProgram.aprx

E2M Pty Ltd gives no warranty in relation to the data (including accuracy, reliability, completeness or suitability) and accepts no liability for any loss, damage or costs (including consequential damage) relating to any use of the data in this map.



Project Area

Time Since Last Burn (Years before 2016)

0 1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 No burn recorded

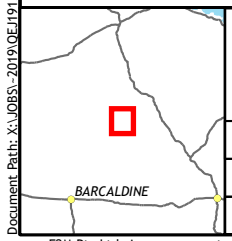
N

Scale 1:250,000 (A4)

0 4.05 8.1 12.15

Kilometers

Coordinate System: GCS GDA 1994



Notes:
Aerial Imagery: © ESRI 2021
Cadastre: © DNRME 2021
Watercourse: © Geoscience Australia 2012
Road: © DNRME 2021

Rev	Description	Drawn	Approved	Date
2	Issued for Use	JY	JMC	30/10/2025
0	Issued for Use	GO	JMC	08/08/2025

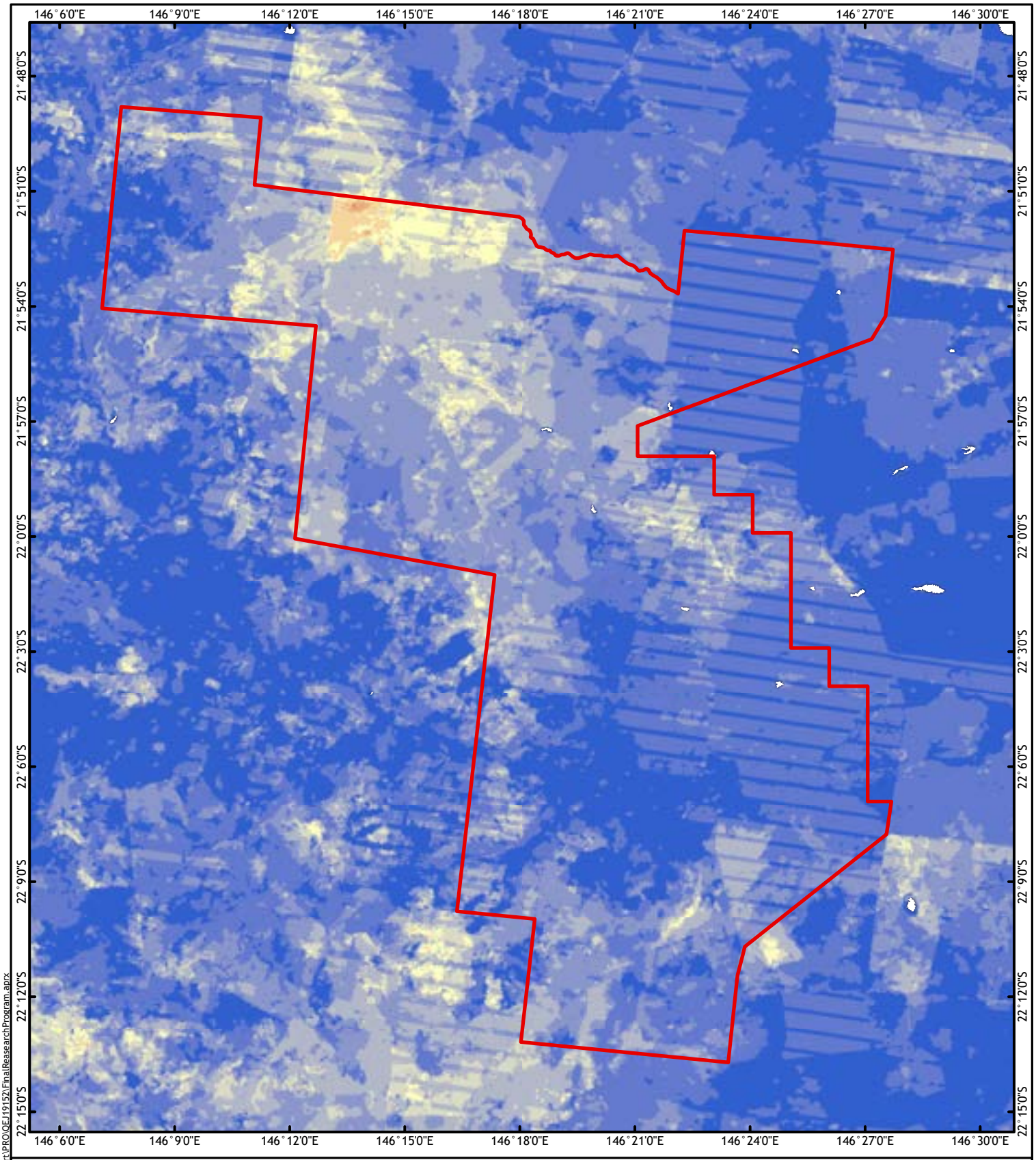
FIGURE 2.1-2: TIME SINCE LAST BURN

BTFRP Final Report
Bravus

Map Number	Job Number	Rev
1 of 1	QEJ19152	2

Document Path: X:\JOBS\-2019\QEJ19152\GIS\Final Research Program Report\PRO\QEJ19152\FinalReasearchProgram.aprx

E2M Pty Ltd gives no warranty in relation to the data (including accuracy, reliability, completeness or suitability) and accepts no liability for any loss, damage or costs (including consequential damage) relating to any use of the data in this map.



Project Area

Cumulative Total of Fire Scars

8 7 6 5 4 3 2 1 0

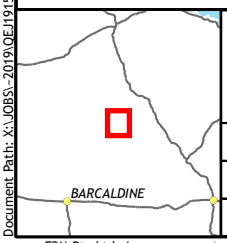
N

Scale 1:250,000 (A4)

0 4.05 8.1 12.15

Kilometers

Coordinate System: GCS GDA 1994



Notes:
Aerial Imagery: © ESRI 2021
Cadastre: © DNRME 2021
Watercourse: © Geoscience Australia 2012
Road: © DNRME 2021

Rev	Description	Drawn	Approved	Date
2	Issued for Use	JY	JMC	30/10/2025
0	Issued for Use	GO	JMC	08/08/2025

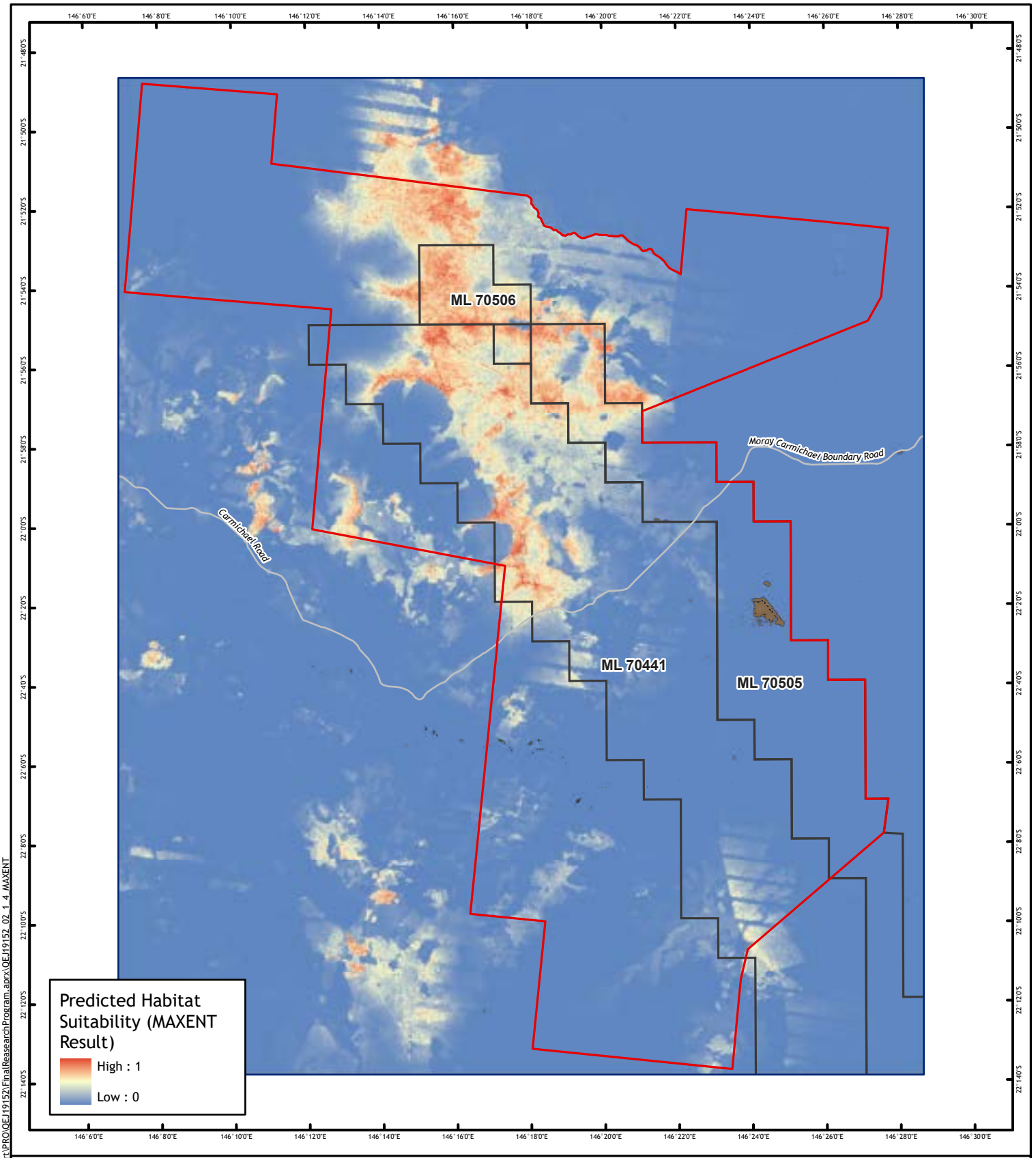
FIGURE 2.1-3: FIRE FREQUENCY

BTFRP Final Report
Bravus

Map Number	Job Number	Rev
1 of 1	QEJ19152	2

Document Path: X:\JOBS\-2019\QEJ19152\GIS\Final Research Program Report\PRO\QEJ19152\FinalReasearchProgram.aprx

E2M Pty Ltd gives no warranty in relation to the data (including accuracy, reliability, completeness or suitability) and accepts no liability for any loss, damage or costs (including consequential damage) relating to any use of the data in this map.



Legend

- Study area
- Model Extent
- Mining Lease

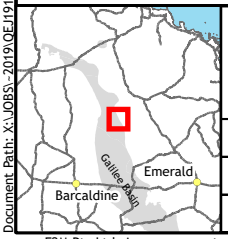
N

Scale 1:240,000 (A4)

0 10

Kilometers

Coordinate System: GDA 1994 MGA Zone 55
Projection: Transverse Mercator



Notes:
Aerial Imagery: © ESRI 2021
Cadastre: © DNRME 2021
Watercourse: © Geoscience Australia 2012
Road: © DNRME 2021

Rev	Description	Drawn	Approved	Date
2	Issued for Use	JY	JMC	30/10/2025
0	Issued for Use	GO	JMC	08/08/2025

FIGURE 2.1-4: MAXENT

BTFRP Final Report
Bravus

Map Number	Job Number	Rev
1 of 1	QEJ19152	2

Document Path: X:\JOBS\-2019\QEJ19152\GIS\Final Research Program Report\PRO\QEJ19152\FinalReasearchProgram.aprx\QEJ19152_02_1_4_MAXENT

E2M Pty Ltd gives no warranty in relation to the data (including accuracy, reliability, completeness or suitability) and accepts no liability for any loss, damage or costs (including consequential damage) relating to any use of the data in this map.

2.2 Permits and approvals

Research activities for the BTFRP were undertaken under the following permits and approvals:

- Animal Ethics Permit (CA 2020/07/1392), issued by the Queensland Department of Agriculture and Fisheries
- Research Permit/Scientific Purposes Permit (permit number WA0025814), issued by the Queensland Department of Environment and Science
- Australian Bird and Bat Banding Scheme Project Approval (banding authority 2832-01), issued by the Australian Government Department of the Environment; and
- Scientific Use Registration (registration number SUR000555), issued by the Queensland Department of Agriculture and Fisheries.

2.3 Surveys and timing

Data for this research program were collected through 13 surveys under the BTFRP and were supplemented with additional SBTF monitoring and habitat assessment data collected under the BTFMP (EcoLogical Australia, 2023). Survey events that have contributed data to these research program results are detailed in Table 2.3-1.

Table 2.3-1: Survey events, timing and methods employed

Survey event	Timing	Methods employed									
		Mist-netting	Colour banding	Radio tracking	1 km transects	Water source watches	Foraging obs.	Habitat assessment	Phenology assessment	Crop and feacal samples	Genetic samples
2020 wet (pilot trial)	25 March to 5 April 2020					✓	✓	✓			
2020 dry	27 August to 16 September 2020	✓	✓	✓	✓	✓	✓	✓			
2021 early wet	27 January to 15 February 2021	✓	✓	✓	✓	✓	✓				
2021 wet	9 to 27 April 2021	✓	✓	✓		✓	✓	✓			
2021 dry	24 August to 26 October 2021	✓	✓	✓	✓	✓	✓	✓	✓		✓
2022 early wet	7 February until 4 March 2022	✓	✓	✓	✓	✓	✓	✓	✓		✓
2022 wet	1 to 22 June 2022	✓	✓	✓		✓	✓	✓	✓		✓
2022 dry	5 to 31 October 2022	✓	✓	✓		✓	✓	✓	✓	✓	✓
2023 early wet	22 February to 7 March 2023	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
2023 wet	4 May to 24 June 2023	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓

Survey event	Timing	Methods employed									
		Mist-netting	Colour banding	Radio tracking	1 km transects	Water source watches	Foraging obs.	Habitat assessment	Phenology assessment	Crop and feacal samples	Genetic samples
2023 dry	5 to 12 October 2023	✓			✓	✓	✓	✓	✓	✓	
2024 early wet	6 to 15 February 2024				✓	✓	✓	✓	✓		
2024 wet	5 April to 31 May 2024				✓	✓	✓	✓	✓		

2.4 Research methods overview

A range of methods have been used to achieve the BTFRP's objectives identified in Section 1.2. The key methods used and their contributions to achieving the BTFRP's objectives are summarised in Table 2.4-1. Detailed descriptions of methods specific to each of the research objectives are contained within the subsequent sections of this report.

Table 2.4-1: Summary of methods employed for the BTFRP

Method	Description	Contribution to research objectives
Mist-netting	Method to capture SBTF for the purpose of: • Banding (including colour banding) • Radio tagging • Morphology measurements • Genetic sample collection • Mark-recapture • Crop and feacal samples	Mist-netting surveys contributed broadly to the research objectives, including: • Home range, habitat use and activity patterns (through radio tracking) (Section 3.1) • Landscape genetics (Section 3.2) • Dispersal and site fidelity (Section 3.3) • Sex-related variation (Section 3.4) • Population estimates (mark-resight and genetic estimates) (Section 3.5) • Diet (Section 5.1)
Colour banding	SBTF were fitted with uniquely identifiable coloured bands, which afforded a means to identify individual SBTF without requiring recapture.	Colour banding and subsequent resightings of SBTF provided data key to the research objectives: • Dispersal and site fidelity (Section 3.3) • Population estimate (mark-resight) (Section 3.5)
Radio tracking	The radio tracking program included an array of 27 automated radio telemetry towers and manual radio tracking used to track radio tagged SBTF that were mist-netted over 9 survey events, covering three years.	Radio tracking data was used to identify the home range, habitat use and activity patterns of SBTF within the Project Area (Section 3.1). Furthermore, radio tracking was used to support data collection on SBTF nest sites (Section 4.1), foraging sites (Section 4.2) and diet through foraging observations (Section 5.1).
1 km transects	Walked surveys that followed a 1 km fixed distance transect (Buckland *et al.*, 2015), which were randomly located within SBTF habitat in the Project Area and included in repeat surveys.	1 km transect data were used to estimate the population of SBTF within the Project Area (Section 3.5.1). In addition, 1 km transects contributed to the assessment of avifauna composition with respect to distance to water (Section 6.2) and contributed to SBTF sighting data used for nest, diet and foraging observations (Sections 4.1, 4.2 and 5.1).

Method	Description	Contribution to research objectives
Water source watches	Surveys of SBTF at water sources, which were primarily used to identify colour-banded SBTF.	Water source watches were used to estimate the population of SBTF within the Project Area (Section 3.5.1) and contributed data to understand the dispersal and site fidelity of SBTF (Section 3.3).
Habitat assessments	Systematic surveys of vegetation within the Project Area. Habitat transects were based on a BioCondition transect (Eyre *et al.*, 2015) that was modified to record additional information on the ground cover composition.	Habitat assessments contributed to various research objectives, including: • Nesting habitat selection (Section 4.1) • Foraging habitat selection (Section 4.2) • Grass seed phenology (Section 5.2) • Effect of grazing and fire on SBTF habitat (Section 6.1)
Phenology assessments	Phenology data were collected during habitat assessments from the 2021 dry season onwards. Phenology data included recording the proportion of grasses holding mature seeds as well as the life-stages of each species (green growth, flowering, mature seeds, etc.)	Phenology assessments were used to describe the seasonal seed availability of grasses within the Project Area (Section 5.2)
Foraging observations	Visual observations of SBTF foraging activity and species eaten. Habitat assessments were also undertaken opportunistically at SBTF foraging sites.	Visual observations of SBTF foraging contributed to the assessment of SBTF foraging habitat selection (Section 4.2) and diet (Section 5.1).
Crop and feacal samples	Crop and feacal samples were collected from SBTF caught in mist-nets to collect diet information. DNA metabarcoding was undertaken on the crop and feacal samples by the Griffith University DNA Sequencing Facility	DNA metabarcoding of crop and feacal samples informed an assessment of SBTF diet (Section 5.1).
Genetic samples	Genetic samples were collected as blood-pricks stored on Flinders Technology Associates cards. Genetic samples were sent to Diversity Arrays Technologies Laboratory for single nucleotide polymorphism sequencing and resulting data analysed by Macquarie University's Conservation Genetics Research Group.	Genetic analyses were used to understand the landscape genetic structuring and population size of SBTF (Section 3.2). Genetic analyses also informed an assessment of sex-related differences in SBTF morphology, home range sizes and diet (Section 3.4).

3 Aim 1: Estimate the home range and movements of SBTF in the Project Area

3.1 Home range, landscape scale habitat selection and daily activity patterns

This section has been published as a scientific paper in the journal *PLOS One*. Bibliographic details of this scientific paper are:

> van Osta, J. M., Dreis, B., Grogan, L. F., & Castley, J. G. Local resource availability drives habitat use by a threatened avian granivore in savanna woodlands. *PLOS One*. 19:8, e0306842. https://doi.org/10.1371/journal.pone.0306842

3.1.1 Abstract

Conserving threatened species relies on an understanding of their habitat requirements. This is especially relevant for granivorous birds, whose habitat use and movement patterns are intricately linked to the spatial and temporal availability of resources such as food and water. In this study, we investigated the habitat use, home range and daily activity patterns of the Endangered southern black-throated finch (SBTF; Poephila cincta cincta) within a 75,000 ha savanna woodland study area in northeastern Australia. This semi-arid region is one of the key remaining strongholds for the species and is characterised by substantially different habitat and climatic conditions than areas where previous research on this species has been undertaken. We radio tracked 142 SBTF using both manual tracking and an array of 27 automated radio towers, which revealed a strong preference for eucalypt-dominated grassy woodland communities. The preference for these habitats also increased with decreasing distance to permanent water. While SBTF occupied large home ranges, individual SBTF were largely sedentary during the radio tracking period (21.8, range = 0.83–120 days), with few landscape-scale movements of more than 4.5 km. Daily foraging activity of SBTF exhibited bimodal peaks in the early morning and late afternoon, while other activities were greatest from the late morning to the early afternoon. Compared to other estrildid finches, our research suggests that SBTF track resources at a local scale across a large home range. We postulate that in times of resource scarcity SBTF may use dietary diversification, instead of landscape or regional-scale nomadic movements, to meet their resource needs. The species' movement patterns underscore the importance of local scale habitat management to facilitate resource availability throughout the year. Furthermore, our research helps target monitoring designs for granivorous birds that focus on the species' diurnal activity patterns.

3.1.2 Introduction

Habitat use is a key focus of research effort as a species' habitat preferences can ultimately influence survival and population persistence (Matthiopoulos *et al.*, 2015). Understanding an animal's habitat use, home range and movements helps to identify how animals may be influenced by the availability and distribution of resources such as food, water, and breeding habitats (Jeltsch *et al.*, 2013; Fraser *et al.*, 2018). Central to habitat use studies is the analysis of habitat preferences, home range and movement patterns (Van Moorter *et al.*, 2016). This knowledge becomes even more critical for species that are threatened and reliant on targeted conservation efforts (Allen & Singh, 2016; Rechetelo et al., 2016). In Australia, granivorous birds, especially ground feeders, are disproportionately represented in threatened species lists and have experienced significant range declines due to changes in resource availability (Franklin, 1999; Garnett, 2021). These declines are primarily attributed to the spread of pastoralism,

which leads to land clearing, altered vegetation composition, changed fire regimes, increased grazing pressure, as well as predation by introduced species (Reid and Fleming, 1992; Franklin *et al.*, 2005).

The granivorous bird assemblage in the tropical savanna biome of northern Australia is rich in estrildid finch species (n=18), many of which have overlapping distributions (Schidelko *et al.*, 2013). These sympatric species exhibit a wide diversity of mobility and dietary behaviours that reduce their interspecific competition (Franklin *et al.*, 2017; Collett *et al.*, 2022b). Species range from double-barred and crimson finches (*Stizoptera bichenovii* and *Neochmia phaeton*) that occupy small home ranges and are dietary generalists through to Gouldian finch (*Erythrura gouldiae*) and pictorella mannikin (*Heteromunia pectoralis*) that are highly nomadic dietary specialists (Franklin *et al.*, 2017; Collett *et al.*, 2022b). Understanding the movement, habitat use and activity patterns of these species is important for identifying the resource niches that are required by species in order to appropriately target management actions. This is particularly important for three of the savanna-inhabiting estrildid finches that are listed as threatened at a national level, namely Gouldian finch, eastern star finch (*Neochmia ruficauda ruficauda*) and southern black-throated finch (*Poephila cincta cincta*) (DCCEEW, 2024). Understanding the movement and habitat use patterns of these species helps to inform conservation actions, such as the management of fire and grazing that are key habitat modifiers in the savanna woodlands (Kutt and Woinarski, 2007; Legge *et al.*, 2019; Woinarski, Andersen and Murphy, 2020).

The southern black-throated finch (SBTF) is one of the nationally threatened estrildid finches that has suffered extensive range decline since European settlement and is listed as Endangered under Federal and State legislation (Mula Laguna *et al.*, 2019; Commonwealth of Australia, 2023). The species inhabits open grassy woodlands (Higgins, Peter and Cowling, 2006; Shephard, Pridham and Forshaw, 2012), and is primarily granivorous (Higgins, Peter and Cowling, 2006; Mula Laguna *et al.*, 2019). Its habitat has also been subjected to extensive degradation, primarily due to the spread of pastoralism and associated changes in land management practices, including habitat clearing, altered fire regimes, changes in grazing pressure and the spread of introduced grasses (Franklin *et al.*, 2005; Mula Laguna *et al.*, 2019).

The contemporary SBTF population is concentrated in two remaining strongholds, namely, the Townsville Plains sub-bioregion and the Desert Uplands Bioregion (Mula Laguna *et al.*, 2019; Commonwealth of Australia, 2023). Published movement and habitat use research on SBTF has focused primarily on the population located on the Townsville Plains (Rechetelo *et al.*, 2016a; Mula Laguna *et al.*, 2019), while similar research from the Desert Uplands Bioregion is sparse. Importantly, the two areas exhibit substantially different climate and habitat characteristics and could lead to different patterns of habitat use by SBTF in the Desert Uplands. The Townsville Plains is within the tropical savanna with a dry winter Köppen-Geiger climate zone, while the Desert Uplands Bioregion is within the hot semi-arid climate zone (Peel, Finlayson and McMahon, 2007). In addition, the two regions share little overlap in vegetation communities (DESI 2021), and the Desert Uplands has less than half the area of freshwater wetlands and artificial dams (DES, 2013a, 2013b).

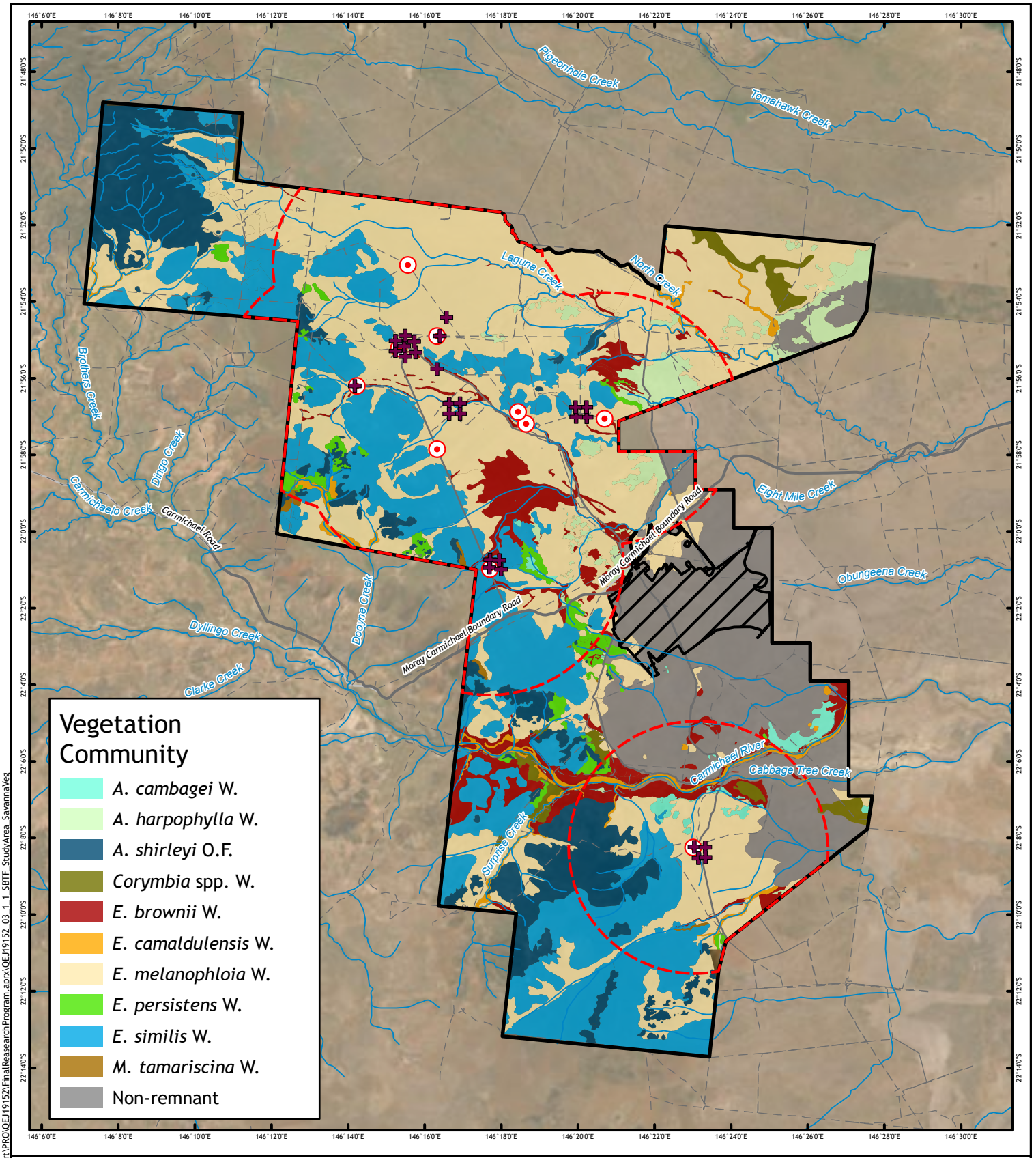
The aim of this study was to identify the habitat use, movement and diel activity patterns of SBTF within the Desert Uplands Bioregion and to test for seasonal variation in these factors. This study builds on previous research from the Townsville Plains by Rechetelo et al. (2016a), and increases the sample size of tracked SBTF and the period over which tracking is undertaken. More broadly, our research aims to improve our understanding of the ecology of the SBTF and aims to contribute to conservation planning and monitoring for the Endangered SBTF and other granivorous birds with similar threatening processes.

3.1.3 Methods

3.1.3.1 Study area

We conducted this study in a 75,000 ha section of the Moray Downs property within the Desert-Uplands Bioregion of Queensland, Australia (Figure 3.1-1). The study area receives an average annual rainfall of 541 mm, 80% of which occurs between November and April, marking a characteristic wet season (Bureau of Meteorology, 2025). Land use within the study area includes a combination of native vegetation managed for conservation as an environmental offset, low-intensity cattle grazing and an active coal mine. The majority of permanent water sources are artificial bores and dams that were established for cattle grazing as well as permanent water associated with a major watercourse, the Carmichael River (Figure 3.1-1).

Remnant vegetation covers approximately 79% of the study area and includes 10 vegetation communities that we assigned based on similarities in dominant canopy species and soil attributes (Figure 3.1-1, Table 3.1-1). These vegetation communities are based on the Queensland Government's regional ecosystem framework (Neldner *et al.*, 2022), which was extensively ground-truthed throughout the study area. Broadly, the study area's vegetation comprises semi-arid savanna woodlands that are dominated by eucalypt and Acacia species (Woinarski, Andersen and Murphy, 2020). Eucalypt woodlands account for 71% of the remnant vegetation, with the most common communities being those dominated by silver-leaved ironbark (*Eucalyptus melanophloia*) (39%) and yellow jacket (*E. similis*) (22%) on deep red earths (Figure 3.1-1, supporting information Table 3.1-1). Other remnant habitats include lancewood (*Acacia shirleyi*) on skeletal Tertiary plateaus, *E. camaldulensis* and *E. brownii* woodlands on alluvial plains associated with watercourses and drainage features, brigalow (*A. harpophylla*) and gidgee (*A. cambagei*) open forests on clay plains, and small areas of *Melaleuca* dominated shrublands (Figure 3.1-1, supporting information Table 3.1-1). In addition, non-remnant areas, which have been cleared of native vegetation (Neldner *et al.*, 2022), cover 21% of the study area. This includes 16% of the study area used for cattle grazing pastures and 5% occupied by mining infrastructure and roads (Figure 3.1-1).



Legend

- Automted Radio Tower
- Mist-Netting Site
- Watercourse
- Street/Local - only provides property access
- Major Road
- Minor Road
- Track
- Mining Infrastructure
- Project Area
- Sample Area

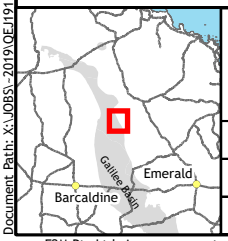
N

Scale 1:250,000 (A4)

0 10

Kilometers

Coordinate System: GDA2020



Notes:
Aerial Imagery: © ESRI 2021
Cadastre: © DNRME 2021
Watercourse: © Geoscience Australia 2012
Road: © DNRME 2021

Rev	Description	Drawn	Approved	Date
2	Issued for Use	JY	JMC	30/10/2025
0	Issued for Use	GO	JMC	08/08/2025

FIGURE 3.1-1: STUDY AREA FOR SBTF IN SAVANNA VEGETATION

BTFRP Final Report
Bravus

Map Number	Job Number	Rev
1 of 1	QEJ19152	2

Document Path: X:\JOBS\-2019\QEJ19152\GIS\Final Research Program Report\PRO\QEJ19152\FinalReasearchProgram.aprx\QEJ19152_03_1_1_SBTF_StudyArea_SavannaVeg

E2M Pty Ltd gives no warranty in relation to the data (including accuracy, reliability, completeness or suitability) and accepts no liability for any loss, damage or costs (including consequential damage) relating to any use of the data in this map.

Table 3.1-1: Vegetation communities within the study area

Short name	Vegetation community	Regional ecosystems*	Area (ha)
A. shirleyi open forest	*Acacia shirleyi* open forest on skeletal soils associated with sandstone ranges and laterite	10.10.1 and 10.7.3	7,382
Corymbia spp. woodland	*Corymbia dallachiana*, *C. plena* and/or *C. terminalis* woodland on sandy soils	10.3.12 and 10.5.2	994
E. camaldulensis woodland	*Eucalyptus camaldulensis* and/or *E. coolabah* open woodland to open forest associated with major water courses	10.3.15, 10.3.13, 10.3.14, 11.3.3, 11.3.25 and 11.3.27	1,296
A. cambagei woodland	*Acacia cambagei* woodland on heavy clay soils	10.3.4, 10.4.5, 11.3.5 and 11.4.6	541
E. brownii woodland	*Eucalyptus brownii* woodland on sandy plains	10.3.6, 10.5.5x2 and 11.3.10	4,152
A. harpophylla woodland	*Acacia harpophylla* woodland on heavy clay soils	10.3.3, 10.4.3 and 11.4.9	2139
E. similis woodland	*Eucalyptus similis* and/or *Corymbia brachycarpa* woodland on sandy plains	10.5.1	16,450
E. melanophloia woodland	*Eucalyptus melanophloia* woodland on sandy plains	10.5.5a	29,388
E. persistens woodland	*Eucalyptus persistens* and *E. thozetiana* woodland on lateritic soils	10.7.4 and 10.7.5	1,098
M. tamariscina woodland	*Melaleuca tamariscina* open shrubland to low woodland on skeletal lateritic soils	10.7.7	107
Non-remnant	Non-remnant areas that have been cleared of native vegetation. These areas include areas converted to improved pasture dominated by exotic grasses, particularly buffel grass (*Cenchrus ciliaris*) and areas occupied by mining infrastructure and roads.	N/a	11,945

* Regional ecosystems form the basis of vegetation mapping in Queensland, Australia (Neldner *et al.*, 2022), and were ground-truthed throughout the study area. Detailed regional ecosystem descriptions are available from: https://apps.des.qld.gov.au/regional-ecosystems/.

3.1.3.2 Ethics statement

Mist-netting, banding and radio tracking were conducted under the animal ethics permit CA 2020/07/1392, issued by the Queensland Department of Agriculture and Fisheries; research permit WA0025814, issued by the Queensland Department of Environment, Science and Innovation; and Australian Bird and Bat Banding Scheme authority 2832-01, issued by the Australian Government Department of Climate Change, Energy, the Environment and Water.

3.1.3.3 Mist-netting and transmitter attachment

Nine mist-netting and radio tracking surveys were conducted between August 2020 and May 2023, with approximately three per year to align with the dry season (May to November), early wet season (December to February) and wet season (February to April). Mist-netting was conducted at 10 locations where SBTF were observed, which included permanent and ephemeral water sources and a single foraging location remote from water. A total of 142 SBTF were fitted with Lotek NTQB2-2 Nanotag radio transmitters, which weighed 0.32 g, approximately 2.1% of the average SBTF weight. An average of 15.8 ± 5.59 transmitters were applied per survey period ($\bar{x}$ ± 1 SD, range = 7–23) from 4.89 ± 1.27 independent locations ($\bar{x}$ ± 1 SD, range = 3–7) per survey. Across the entire study period 14.2 ± 11.0 transmitters were applied per survey location ($\bar{x}$ ± 1 SD, range = 1–38)[1]. Transmitters were attached to the scapular region of the birds by trimming a small patch of feathers level with the skin and using cyanoacrylate glue (Johnson, Pebworth and Krueger, 1991; Diemer, Wheeler and Nocera, 2014; Rechetelo *et al.*, 2016a). Prior to release, transmitter function and signal detection were tested using a handheld receiver (Lotek SRX1200 M2). All of the radio transmitters were set to 150.6 MHz and emitted uniquely coded radio pulses, allowing multiple transmitters to be recorded simultaneously using both manual tracking as well as the automated radio telemetry arrays. Transmitters with a 13 second pulse interval (n = 73) and 97 day battery life were initially used based off an estimated transmitter retention time (Diemer, Wheeler and Nocera, 2014). Following initial field surveys and evaluation of tag retention times, we changed to a 3 second pulse interval and 29-day battery life (n = 69) to better reflect the tag retention that was being achieved, which was 21.8 ± 22.5 days ($\bar{x}$ ± 1 SD , range = 0.83–120 days) and improve manual radio tracking efficiency. All SBTF captured for the purpose of the radio tracking study (n = 142) were also fitted with a uniquely coded Australian Bird and Bat Banding Scheme supplied alloy band.

3.1.3.4 Manual radio tracking

We used a combination of tracking approaches by combining manual tracking with an automated radio telemetry system to assess the movement and habitat use of 142 SBTF over a three-year period. We excluded tracking data obtained within 24 hours of transmitter attachment from all statistical analyses to allow a post-release acclimation period.

We manually radio-tracked SBTF using a handheld 3-element Yagi-Uda antenna connected to a Lotek SRX1200-M2 receiver. We aimed to capture the location of each tracked SBTF approximately twice per day. Where field time allowed, SBTF were also tracked at a higher temporal resolution, of up to one location per 15 minutes until the signal was lost, which provided additional information on diel activity patterns. Tracking bouts ranged from single sightings to approximately 8 hours of tracking per bird. We collected the location of tracked SBTF after visually resighting the individuals, where possible. Care was taken to avoid disturbing the SBTF while radio-tracking to limit behavioural changes associated with tracking. For higher temporal resolution tracking, SBTF positions were occasionally estimated from the maximum radio signal strength and direction when achieving a visual resight was likely to flush foraging birds. We recorded the GPS locations of tracked SBTF using a Trimble TDC600 from the position where the radio-tracked bird was prior to moving off. A positional error of 20 m was assigned to the manual radio tracking locations, approximating the uncertainty of locating the exact position that the radio-tracked SBTF was observed, as well as additional GPS error.

[1] Treatment information and home range sizes of SBTF have been published within an open access database: https://doi.org/10.6084/m9.figshare.26043391.

3.1.3.5 Automated radio telemetry system

We used an array of 27 automated radio telemetry (ART) towers to remotely and continuously estimate the location of radio tracked SBTF when they were within the range of one or more towers. Site-based testing of the towers found a typical detection distance of approximately 300 m when SBTF were foraging on the ground and 800 m when SBTF were perching, which they typically do 2–10 m above ground height. We installed the ART towers within remnant vegetation in areas of known foraging, drinking and nesting habitats for SBTF. The setup included four arrays of four towers each, and an additional array of 11 towers that comprised seven units in the centre surrounded by four units in peripheral areas (Figure 3.1-1). This design allowed simultaneous detections of SBTF across multiple towers, particularly in the centre of an array area. The ART towers were on average 10.3 km (range: 5.03–14.9 km) from the nearest mining infrastructure.

Automated radio telemetry data require extensive quality assurance steps to ensure reliable location estimates (Kays *et al.*, 2011; Gottwald *et al.*, 2019). Our quality assurance steps included an initial filter of the ART data to include only those tag detections that were recorded after the time of release of the bird and within the maximum possible transmitter battery life of approximately 29 days or 97 days, depending on tag pulse rate. Field observations found that transmitters typically exceeded the manufacturers specification by approximately 10–40% and as such we multiplied the life of the transmitter published by the manufacturer (Lotek) by 1.5, to ensure that all true tag detections were captured. After filtering, we arranged the detections by tower ID and time and manually screened for outliers. The minimum convex polygons (MCP) for each tag ID were calculated using ArcGIS Pro (ESRI), and the outliers were manually verified. We then checked for dropped transmitters by plotting the received signal strength of each transmitter on each antenna and manually screening for a time after which the signal strength did not change, suggesting that the transmitter was stationary for an extended period.

Transmitter positions were estimated from the ART data using a data pipeline that is described in Section 7.5. The data pipeline used site-specific training data collected manually by radio-tracking the SBTF to develop a model to estimate transmitter positions (Section 7.5).The median error of the ART position data was estimated at 235 m using a test dataset (Section 7.5), which was factored into the home range estimates discussed below.

3.1.3.6 Landscape scale habitat use

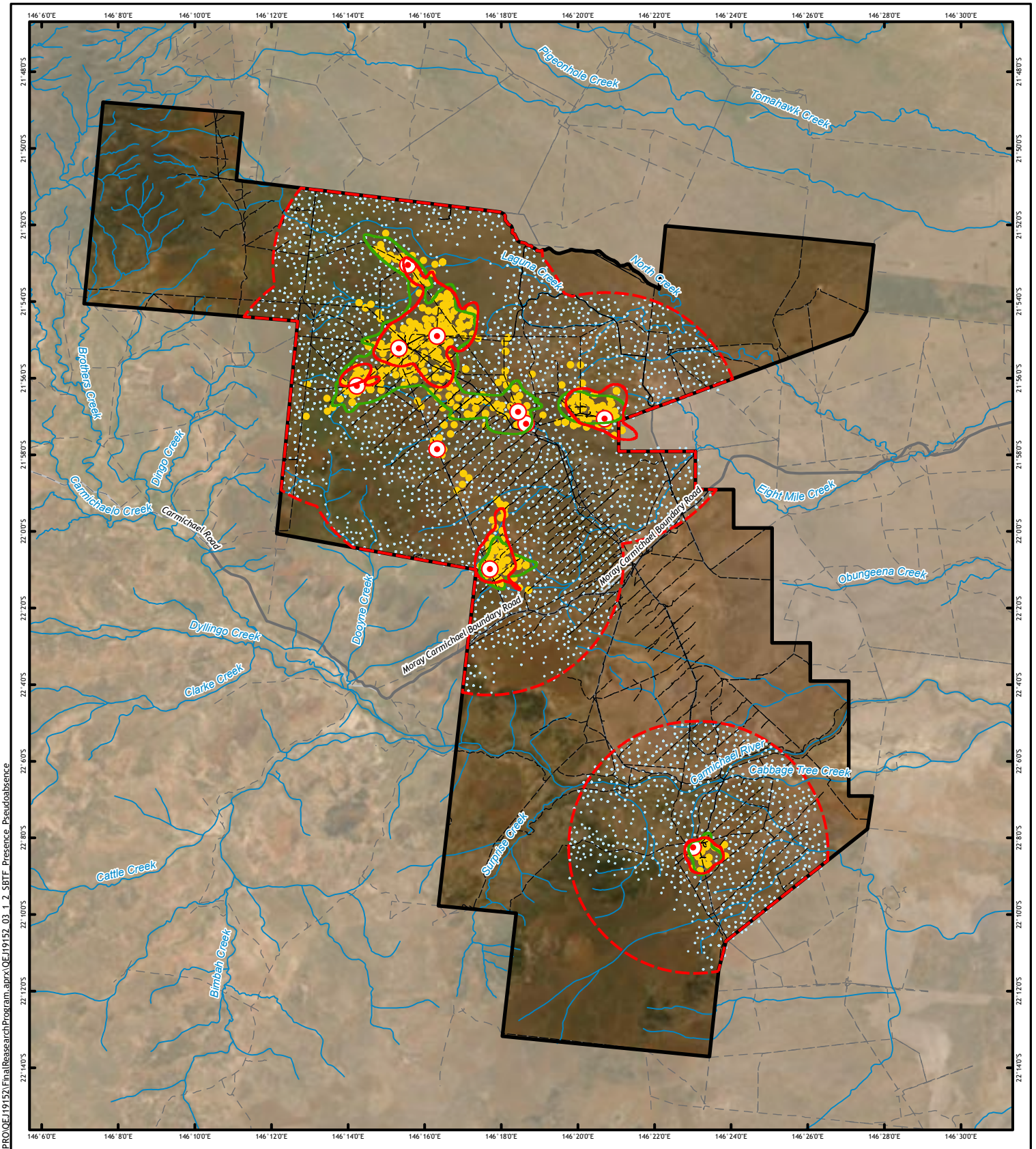
Animals select habitat at multiple spatiotemporal scales (Mayor *et al.*, 2009a). We chose to assess SBTF habitat use at the 'landscape scale', which is the area that an individual chooses to inhabit on a time-scale of days to months (Mayor *et al.*, 2009a). This scale is the most suitable for the intensity and sampling duration at which we obtained radio tracking data. The landscape scale is also suitable for the 75,000 ha study area, which represents an area that is much larger than the individual home ranges of SBTF, while being small enough that a SBTF could, if it chose to, access any part of the study area over a timescale of days (Mayor *et al.*, 2009a; Paton and Matthiopoulos, 2016).

To assess landscape scale habitat use, we developed a resource selection function that used presence sites from manual radio tracking and ART locations and generated pseudoabsence sites (Boyce *et al.*, 2002; Manly *et al.*, 2007). Presence sites were randomly subsampled to a maximum of two sites per day from each of the manual radio tracking and ART locations to minimise bias from auto-correlation of radio tracking data (Rechetelo *et al.*, 2016a; Alston *et al.*, 2023). The minimum time allowed between ART subsamples was 15 minutes, which matched the minimum sampling frequency of manual radio tracking. We generated pseudoabsence sites within a subset of the study area that we defined as the 'sample area' (Figure 3.1-1) at a 10x multiple of presence sites using ArcGIS Pro (ESRI), which followed recommendations by Barbet-Massin et al. (2012). The sample area was a circle centred on each SBTF trapping site that had a radius equal to the maximum distance that these birds were observed to travel

from the trapping site during the tracking period (6.07 km). Areas on adjacent properties within these buffers were excluded due to access restrictions. Since detection of radio-tracked SBTF was likely to be highest near vehicle tracks, which is a common constraint of land-based radio-telemetry studies (Kenward, 2000), we randomly generated pseudoabsence sites at the same proportional distribution with distance from vehicle tracks as our presence sites (Figure 3.1-3 and Figure 3.1-2) (Barbet-Massin *et al.*, 2012). The coverage of pseudoabsence sites therefore closely approximated the effective radio-tracking area coverage of our study.

The independent variables selected for the model included vegetation community (supporting information Table 3.1-1), distance to permanent water and season. We mapped the distributions of permanent water sources with a high level of confidence using a combination of Queensland Government mapping, information supplied by landholders, interpretation of high-resolution aerial imagery and manual verification during field surveys. Additionally, we tested the effect of season on habitat use by adding first order interaction effects between season and vegetation community and season and distance to permanent water. The study area's 'wet season' typically occurs between November and April; however, there is year to year variability in the onset and cessation of rainfall that defines the wet season. We therefore categorised records as wet or dry season based on the yearly timing of rainfall, availability of ephemeral water and growth of annual grasses.

We used Firth's logistic regression as the resource selection model, which was implemented in the logistf R package (Heinze and Schemper, 2002). Firth's logistic regression was selected due to complete separation of the response variable (SBTF presence/pseudoabsence) among some vegetation communities. Firth's method is robust to convergence issues in the presence of separation while still producing unbiased model estimates (Heinze and Schemper, 2002; Puhr *et al.*, 2017). We converted model estimates to odds ratios using the sjPlot package (Lüdecke, 2023), which is a standard approach to facilitate interpretation of logistic regression coefficients (Gelman, Hill and Vehtari, 2020). Our analysis accounted for differences in the relative abundance of vegetation communities by comparing SBTF presence sites with randomly generated pseudoabsence sites. Vegetation communities with larger areas generate proportionally more pseudoabsences, which ensured that the model adjusts for the area of each vegetation community when assessing SBTF habitat preferences.



Legend

- Mist-Netting Site
- Pseudoabsence site
- Presence site
- Watercourse
- Major Road
- Minor Road
- Track
- Street/Local - only provides property access
- Dry season mean 95% AKDE
- Wet season mean 95% AKDE
- Sample Area
- Project Area

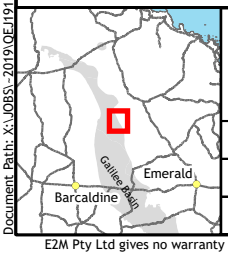
N

Scale 1:250,000 (A4)

0 10

Kilometers

Coordinate System: GDA2020



Notes:
Aerial Imagery: © ESRI 2021
Cadastre: © DNRME 2021
Watercourse: © Geoscience Australia 2012
Road: © DNRME 2021

Rev	Description	Drawn	Approved	Date
2	Issued for Use	GO	JMC	30/10/2025
0	Issued for Use	GO	JMC	08/08/2025

FIGURE 3.1-2: SBTF WET AND DRY SEASON HOME RANGES AND PRESENCE AND PSEUDOABSENCE SITES

BTFRP Final Report
Bravus

Map Number	Job Number	Rev
1 of 1	QEJ19152	2

Document Path: X:\JOBS\-2019\QEJ19152\GIS\Final Research Program Report\PRO\QEJ19152\FinalReasearchProgram.aprx\QEJ19152_03_1_2_SBTF_Presence_Pseudoabsence

E2M Pty Ltd gives no warranty in relation to the data (including accuracy, reliability, completeness or suitability) and accepts no liability for any loss, damage or costs (including consequential damage) relating to any use of the data in this map.

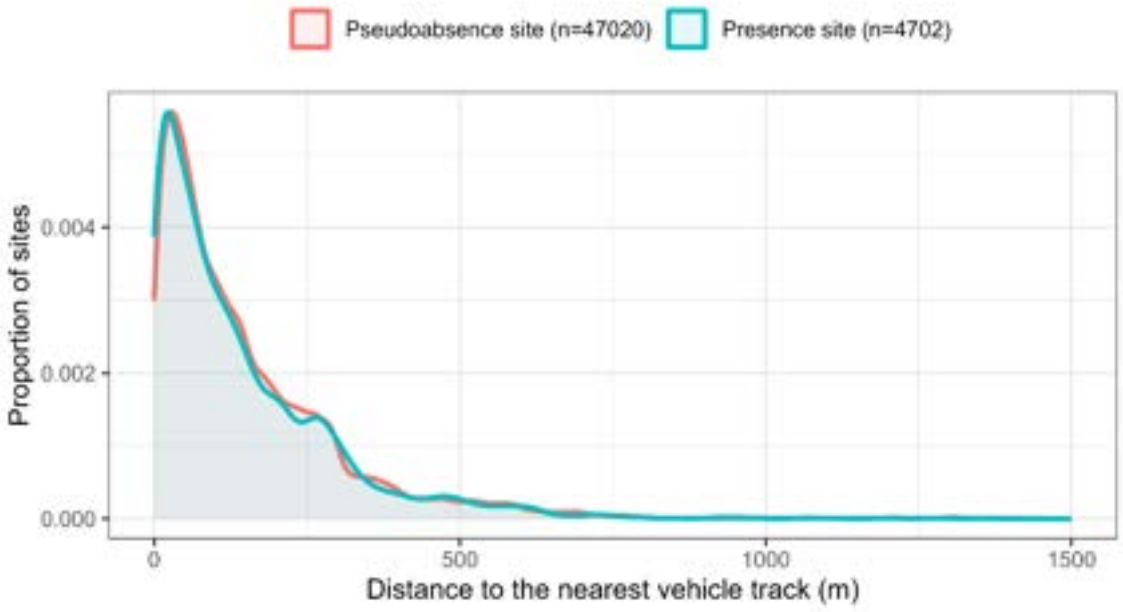


Figure 3.1-3: Distribution of presence sites and pseudoabsence sites relative to their distance to the nearest vehicle track.

3.1.3.7 Home range

We calculated autocorrelated kernel density estimate (AKDE) home ranges using the ctmm package version 1.2.0 (Calabrese, Fleming and Gurarie, 2016). We used the pHREML wAKDE$_c$ method, following recommendations of Silva et al. (2022), which applied bias correction methods for inconsistent sampling intervals, autocorrelation of locations and small sample sizes. For each transmitter, we initially trialled the inclusion of all manual radio-tracking and ART location estimates within the model. However, despite the autocorrelation corrections applied by the model, we found that the home ranges were heavily biased to the ART data. To account for this, we subsampled the ART location estimates to two random locations per day, with a minimum of 15 minutes between subsamples, which approximated the sampling frequency of the manual radio tracking dataset.

We included estimated positional errors for each location in the AKDE home range calculations. These positional errors were 20 m for manual radio tracking locations and 235 m for ART locations, which are described in more detail in the respective methods sections above. Home ranges and core ranges were estimated using the 95% and 50% AKDE contours, respectively (Laver and Kelly, 2008). Home range estimates were plotted against their effective sample sizes, which are the number of statistically independent samples within the data (Fleming *et al.*, 2019). We used these plots to verify that the autocorrelation and sample size correction methods were sufficient to avoid small effective sample sizes underestimating the SBTF home ranges (Silva *et al.*, 2022).

We tested the impact of season on the 95% AKDE home range of SBTF using a generalised linear mixed effects model with a log link gamma distribution using the glmmTMB package version 1.1.7 (Magnusson *et al.*, 2023). Wet and dry seasons were categorised following the same method as the landscape scale habitat use method. The mist-netting site was added as a random effect to this model.

We also calculated minimum convex polygons from all manual radio tracking locations and ART locations for each transmitter using ArcGIS Pro (ESRI), which is a coarse home range tool that we selected to provide a comparative method to SBTF home range estimates for the Townsville Plains population produced by Rechetelo et al. (2016a).

3.1.3.8 Diel activity patterns

Where possible during manual radio tracking, observers categorised SBTF activity as foraging, drinking, within a nest, flying, perching or preening. The flying, perching and preening categories were combined for statistical analysis as these activities were often undertaken in association with each other.

We used the activity package version 1.3.4 (Rowcliffe, 2023) to standardise the time to sunrise and sunset using a double anchoring method (Vazquez *et al.*, 2019). This method anchors the time to equinox sunrise and sunset times for the study area (Vazquez *et al.*, 2019). We transformed the clock times so that equinox sunrise occurred at 6 am and sunset occurred at 6 pm. Anchoring time to both sunrise and sunset is important for SBTF, which arrive and depart their nest based on sunrise and sunset times (Zann, 1972).

There was unequal survey effort across seasons and times of day, making a direct analysis of counts inappropriate. We instead created an unbiased metric for activity by calculating the proportion of each activity relative to other activities for categorical combinations of season and time of day. The time of day was assigned to five categorical bins, which included early morning (6:30–8:30 am), late morning (8:30–10:30 am), midday (10:30 am–1:30 pm), early afternoon (1:30–3:30 pm) and late afternoon (3:30–5:30 pm). Night and twilight hours (within 30 minutes of dawn and dusk) were excluded due to a limited sample size of manual tracking records. Wet and dry seasons were categorised following the same method as the landscape scale habitat use method. We then used a linear model to test the impact of time of day and season on the proportion of observations made for each activity. The model included interaction effects between activity and time of day, as well as between activity and season. All data analyses were undertaken using R statistical language version 4.3.0 (R Core Team, 2023). Unless otherwise specified, data in the results section below are reported as mean ± SD (range).

3.1.4 Results

3.1.4.1 Radio tracking

For each of the 142 radio tracked SBTF, we recorded an average of 1890 ± 4180 ART locations (range = 0–45600) and 11.9 ± 11.9 manual radio tracking locations (range = 1–57). All SBTF were detected through manual tracking at least once, while eight SBTF were not detected on the ART receivers post-release. The average number of manual radio tracking detections per day that the individual was detected was 3.3 (range = 1–30).

Following the data subsampling steps described in the methods, which aimed to ensure independence of the data, we used an average of 39.9 ± 32.4 locations per SBTF (range = 5–199) to estimate home ranges and 33.1 ± 29.5 locations per SBTF (range = 2–198) to test landscape scale habitat use.

Radio tracked SBTF were resighted at an average of 810 ± 730 m from their initial capture location (range = 0–6070 m). All resightings except seven (99.9 ± 0.05%; $\bar{x}$ ± 1 SE) were within 4.5 km of their initial capture location, while 70.8 ± 0.6% ($\bar{x}$ ± 1 SE) of resightings were within 1 km of their initial capture location (Figure 3.1-4).

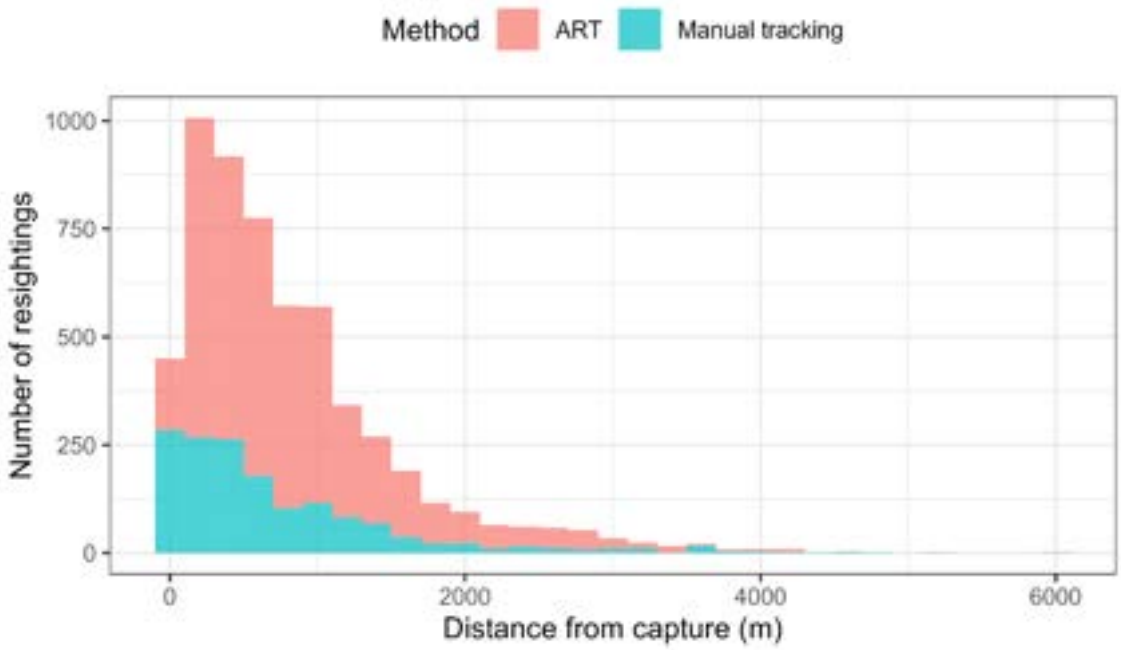


Figure 3.1-4: Histogram showing the distribution of radio tracked southern black-throated finch in the Desert Uplands region of Queensland, Australia. Resightings are relative to the distance from their point of capture and include resightings from automated radio telemetry (ART, red) and manual radio tracking (blue).

3.1.4.2 Landscape scale habitat use

At the landscape scale, we observed that SBTF strongly preferred three vegetation communities (Figure 3.1-5). These preferences were for *Eucalyptus melanophloia* woodland (odds ratio [OR] = 22.8, 95% confidence interval [95% CI]: 8.06–109), *E. brownii* woodland (OR = 17.5, 95% CI: 6.14–83.9) and *E. similis* woodland (OR = 17.7, 95% CI: 6.24–84.8). Conversely, SBTF strongly avoided non-remnant vegetation (OR = 0.021, 95% CI: 0.00015–0.259). There was little seasonal difference in the use of most vegetation communities (Figure 3.1-5). However, there was a weak but statistically significant decrease in the use of *E. brownii* woodland and *E. similis* woodland during the wet season compared to the dry season (OR = 0.774, 95% CI: 0.630–0.949; OR = 0.599, 95% CI: 0.499–0.719; respectively; Figure 3.1-5).

We resighted radio tracked SBTF at an average of 1240 ± 868 m from permanent water (range = 5.34–4210 m). Southern black-throated finch habitat use was strongly related to the distance to permanent water, with SBTF preferentially using areas closer to permanent water (OR = 0.525, 95% CI: 0.498–0.553; Figure 3.1-5). Their use of these areas did not differ significantly among seasons (OR = 1.02, 95% CI: 0.945–1.10; Figure 3.1-5).

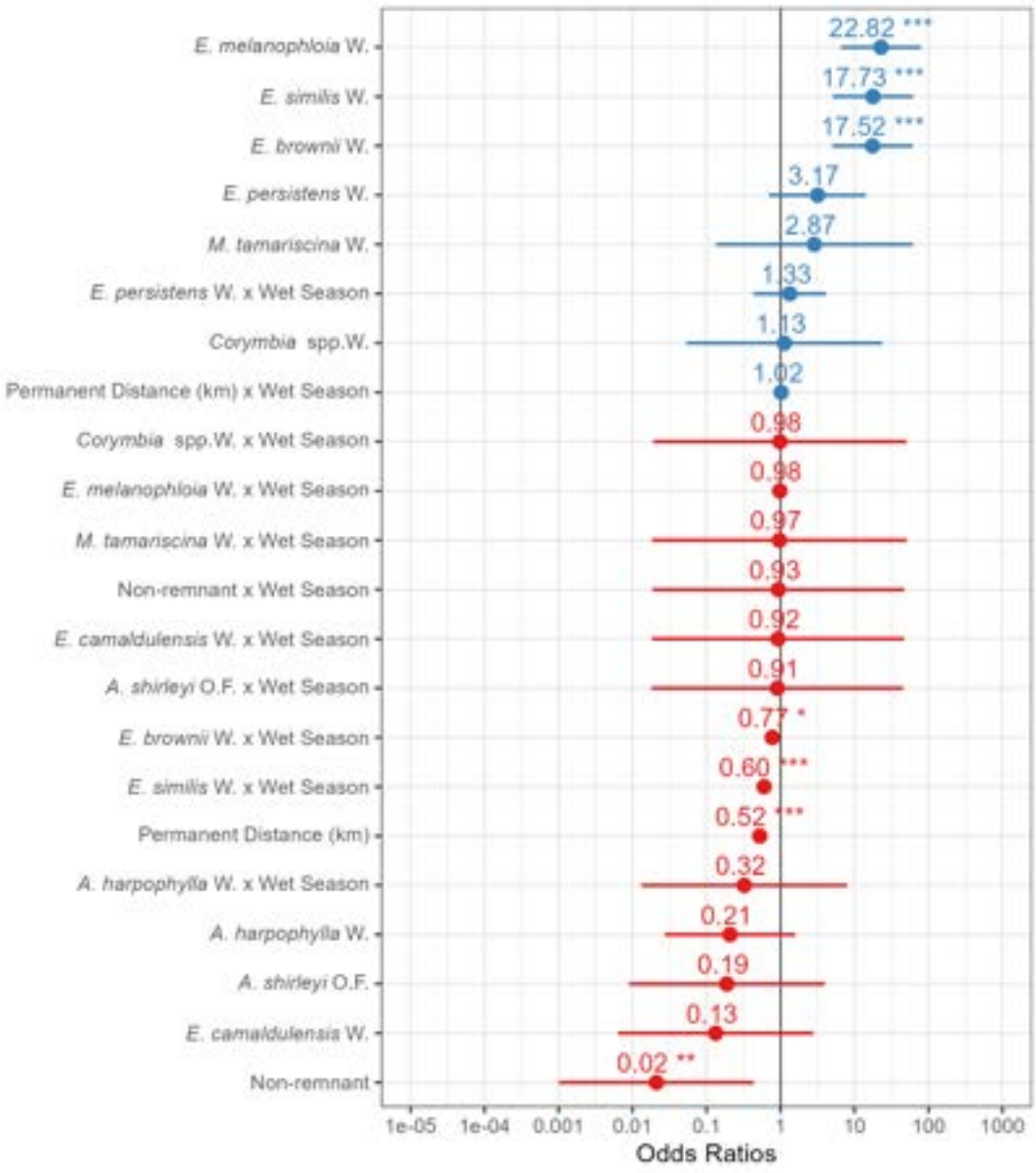


Figure 3.1-5: Odds ratios of the predictor variables of a logistic regression resource use model in the Desert Uplands region of Queensland, Australia. Odds ratios above 1 represent a use preference by southern black-throated finch (blue), whereas odds ratios lower than 1 represent avoidance (red). The error bars show 95% confidence intervals. ***p < 0.001, **p < 0.01, *p < 0.05. Abbreviations include A = *Acacia*, E = *Eucalyptus*, M = *Melaleuca*, O.F. = open forest and W = woodland.

3.1.4.3 Home range

We were able to fit utilisation distributions (UDs) in ctmm for 121 SBTF that had a sufficient effective sample size (Fleming and Calabrese, 2017; Fleming *et al.*, 2019). The average tracking period for these UDs was 21.8 ± 22.5 days (range = 0.827–120 days), and the average number of positions per individual was 42.4 ± 33.1 (range = 5–199).

The mean 95% AKDE home range estimate was 319 ± 43.2 ha ($\bar{x}$ ± 1 SE, range = 8.58–3210 ha) and 50% core range estimate was 72.4 ± 10.2 ha ($\bar{x}$ ± 1 SE, range = 1.89–752 ha). The mean MCP home range estimate

was 176 ± 17.2 ha ($\bar{x}$ ± 1 SE, range = 2.10–1220 ha). There was little difference in the 95% home range sizes of SBTF between the wet and dry seasons (GLMM: ß = 0.27, 95% CI: -0.17–0.71, Table 3.1-2). Furthermore, the distribution of SBTF home ranges within the study area was broadly similar between the dry and wet seasons (Figure 3.1-2).

The 95% and 50% AKDE home range areas did not significantly correlate with the effective sample size (linear regression: R^2 = 0.01, n = 121, p = 0.118; and R^2 = 0.0007, n = 121, p = 0.30; respectively; Figure 3.1-6), indicating that the autocorrelation and small sample size correction measures employed were successful in mitigating sample size bias in home range estimates.

Table 3.1-2: Estimated regression parameters, standard errors, z-values and p values for a generalised linear mixed effects model relating home range size to season.

Predictor	Estimate	Std. error	z-value	p
(Intercept)	5.55	0.233	23.8	< .0001
Wet Season	0.27	0.223	1.21	0.225
Observations	121			
R2 conditional/R2 marginal	0.201/0.0152			

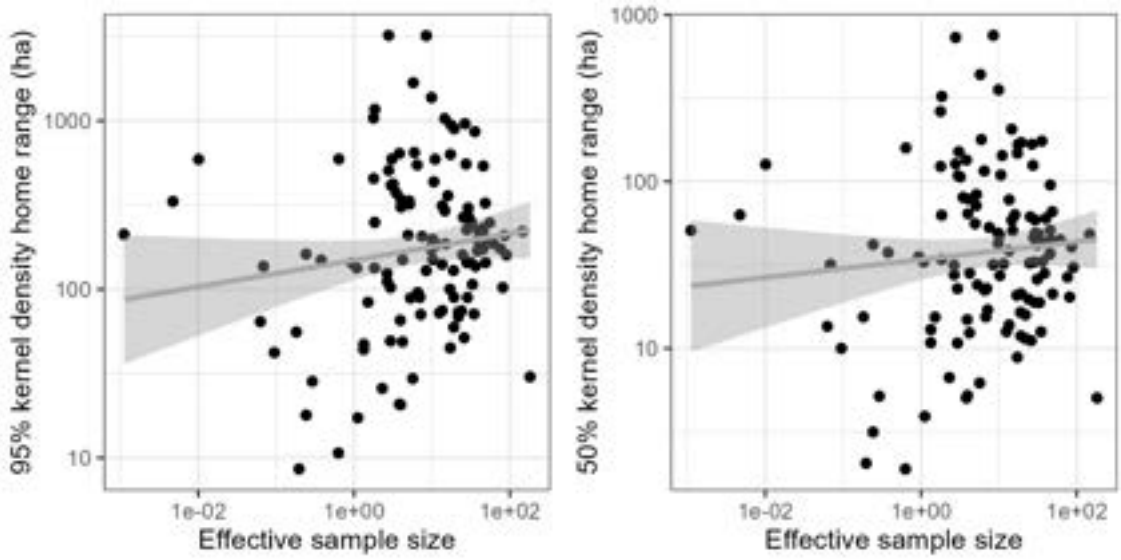


Figure 3.1-6: 95% autocorrelated kernel density estimate (AKDE) home ranges and 50% AKDE core ranges compared to the effective sample sizes that are calculated within the ctmm package.

3.1.4.4 Diel activity patterns

We recorded 1054 SBTF activity observations through manual radio tracking, of which 990 were during diurnal hours, between 6:30 am and 5:30 pm (solar corrected times). Most of these records were SBTF foraging (48.0%, n = 480) or a combination of perching, preening or flying (42.8%, n = 426). Observations of drinking (4.66%, n = 46) and birds within nests (4.55%, n = 47) were less frequent. All nocturnal observations (6:30 pm to 5:30 am) were of SBTF within nests (n = 27; Figure 3.1-7).

Dawn and dusk radio tracking indicated that SBTF typically leave their nests around first light approximately 30 minutes before sunrise. In the evening, they engaged in extended perching and preening sessions before moving close to their nests at sunset and entering them around last light. Activities observed at dawn and dusk were mixed and included perching, preening, and flying (51.4%, n = 19), foraging (21.6%, n = 8) and birds within nests (27%, n = 10) (Figure 3.1-7).

There were significant diurnal patterns in SBTF activities (Figure 3.1-8). The early morning and late afternoon periods had similar proportions of foraging (β = -0.071, 95% CI: -0.17–0.026) and perching, preening and flying (β = 0.059, 95% CI: -0.038–0.16) (Figure 3.1-8, supporting information Table 3.1-3). Compared to the early morning, there were significantly lower proportions of foraging in the late morning (β = -0.19, 95% CI: -0.28–-0.09), midday (β = -0.18, 95% CI: -0.28–-0.085) and early afternoon (β = -0.26, 95% CI: -0.36–-0.17). In contrast, perching, preening and flying was significantly higher in the late morning (β = 0.13, 95% CI: 0.028–0.22), midday (β = 0.13, 95% CI: 0.35–0.23) and early afternoon (β = 0.24, 95% CI: 0.15–0.34) than compared to the early morning and late afternoon (Figure 3.1-8, supporting information Table 3.1-3).

Drinking was most common at midday (β = 0.073, 95% CI: -0.024–0.17); however, the confidence intervals overlapped with the late morning (β = 0.038, 95% CI: -0.059–0.13), early afternoon (β = 0.036, 95% CI: -0.061–0.13) and late afternoon (β = 0.046, 95% CI: -0.051–0.14). No drinking observations were recorded during the early morning period.

The only diurnal activity exhibiting seasonal variation was records of birds within nests, with these being significantly higher in the wet season than the dry season (95% CI: 0.0041–0.13; supporting information Table 3.1-3).

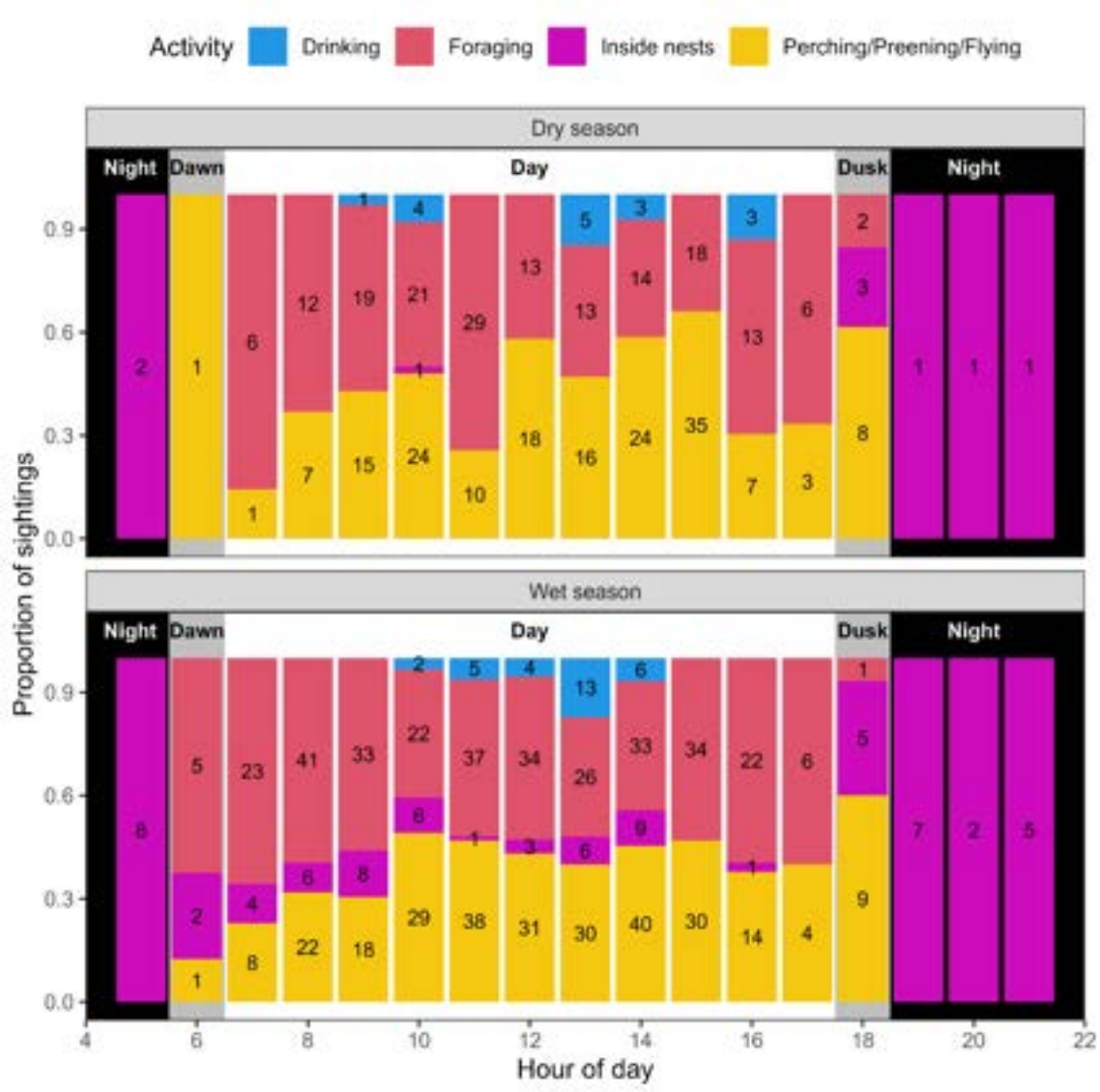


Figure 3.1-7: Daily activity records of southern black-throated finch by time of day and season. Numbers show a count of the number of observations for each bar.

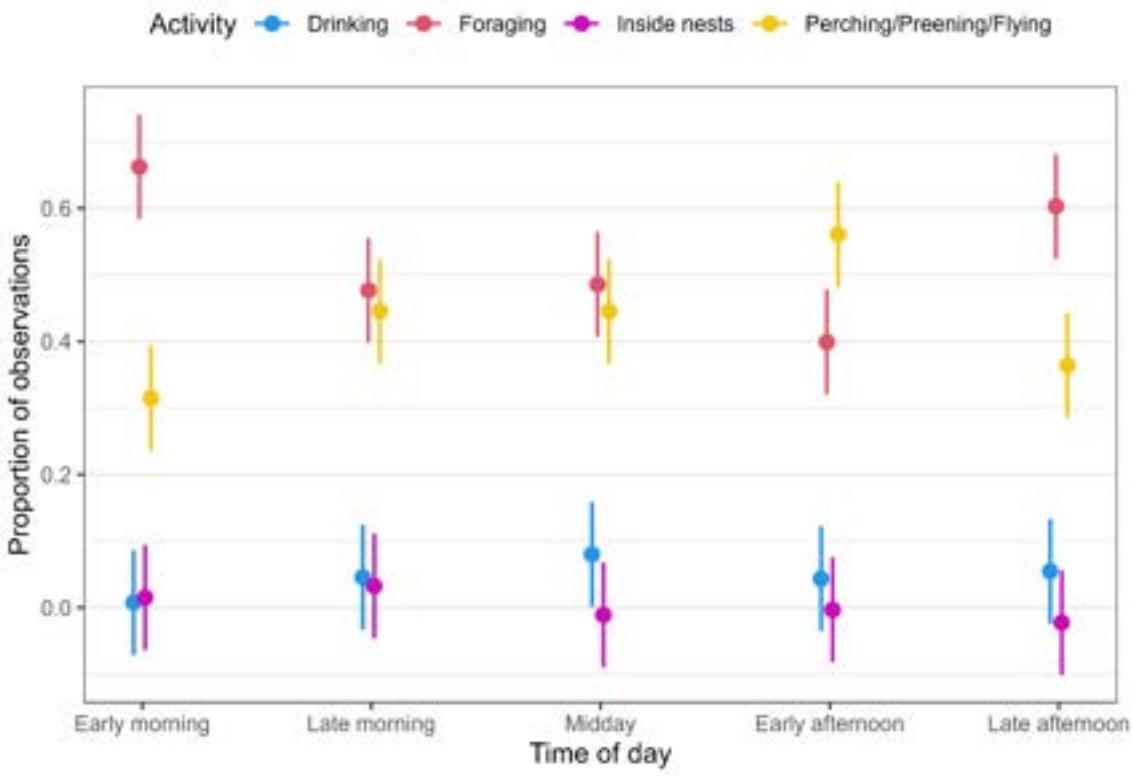


Figure 3.1-8: Diel activity patterns of southern black-throated finch. The bars show confidence intervals established using a linear model. Dusk and night periods were excluded from the linear model due to sample size constraints.

Table 3.1-3: Estimated regression parameters, standard errors, t-values and P-values for the linear model assessing the relationship among activity, season and time of day.

Predictor	Estimate	Std. error	t-value	p
(Intercept)	0.00771	0.0353	0.218	0.83
Activity (foraging)	0.656	0.05	13.1	< .001
Activity (nesting)	0.00337	0.05	0.0674	0.947
Activity (perching/preening/flying)	0.31	0.05	6.2	< .001
Activity (drinking) x Season (wet)	-0.0154	0.0288	-0.534	0.6
Activity (foraging) x Season (wet)	-0.0135	0.0288	-0.469	0.645
Activity (nesting) x Season (wet)	0.0652	0.0288	2.26	0.038
Activity (perching/preening/flying) x Season (wet)	-0.0363	0.0288	-1.26	0.226
Activity (perching/preening/flying) x Time of day (late morning)	0.125	0.0456	2.74	0.0147

Predictor	Estimate	Std. error	t-value	p
Activity (perching/preening/flying) x Time of day (midday)	0.131	0.0456	2.88	0.0109
Activity (perching/preening/flying) x Time of day (early afternoon)	0.243	0.0456	5.32	< .001
Activity (perching/preening/flying) x Time of day (late afternoon)	0.0587	0.0456	1.29	0.217
Activity (drinking) x Time of day (late morning)	0.0375	0.0456	0.822	0.423
Activity (drinking) x Time of day (midday)	0.0725	0.0456	1.59	0.131
Activity (drinking) x Time of day (early afternoon)	0.0359	0.0456	0.786	0.443
Activity (drinking) x Time of day (late afternoon)	0.0455	0.0456	0.997	0.334
Activity (foraging) x Time of day (late morning)	-0.187	0.0456	-4.11	< .001
Activity (foraging) x Time of day (midday)	-0.182	0.0456	-3.99	0.00105
Activity (foraging) x Time of day (early afternoon)	-0.264	0.0456	-5.8	< .001
Activity (foraging) x Time of day (late afternoon)	-0.0711	0.0456	-1.56	0.139
Activity (nesting) x Time of day (late morning)	0.0251	0.0456	0.551	0.589
Activity (nesting) x Time of day (midday)	-0.0218	0.0456	-0.477	0.64
Activity (nesting) x Time of day (early afternoon)	-0.0141	0.0456	-0.309	0.761
Activity (nesting) x Time of day (late afternoon)	-0.0331	0.0456	-0.725	0.479
Observations	40			
R2 adjusted	0.962			

3.1.5 Discussion

3.1.5.1 Landscape scale habitat use

Resource availability is a key driver of habitat use (Mayor *et al.*, 2009a; Tischler, Dickman and Wardle, 2013). The distribution of preferred vegetation communities and drinking sites is therefore expected to affect habitat use by granivorous birds. We found that at a landscape scale, SBTF habitat use was strongly associated with three vegetation communities and the availability of permanent water. The SBTF strongly selected woodlands dominated by *E. melanophloia, E. brownii* and *E. similis*, which within the study area comprise habitats with a high richness of native tussock grass species that comprise the majority of the SBTF diet (Mitchell, 1996; Mula Laguna et al., 2019; Williams et al., 2020). Conversely, non-remnant vegetation was strongly avoided. Within the study area, non-remnant vegetation is dominated by buffel grass (*Cenchrus ciliaris*), an introduced species that outcompetes native grasses and forbs (Eyre *et al.*, 2009) and provides little forage value for SBTF (Wright *et al.*, 2021). In addition, non-remnant areas have been historically cleared and lack canopy trees in which SBTF typically roost, nest and escape predators (Zann, 1972). At the landscape scale, SBTF also preferred sites close to permanent water, suggesting that

the species' requirement for regular access to drinking water promotes the use of habitats requiring less travel costs. Close associations with permanent water have been identified for the SBTF population of the Townsville Plains, with Rechetelo et al. (2016a) resighting the majority of radio tracked SBTF within 200 m of water. While our study found a significant increase in SBTF use of habitat near water, the effect size was smaller than that suggested by Rechetelo et al. (2016a), with SBTF being resighted on average 1.2 km and up to 4.2 km from water. Our findings clearly demonstrate the importance of identifying the habitat preferences of threatened species, in order to target these habitats for conservation actions (Fraser *et al.*, 2018).

The availability of resources varies through time and space across the Australian savanna woodlands, primarily driven by variable rainfall, habitat and land management (Kutt and Woinarski, 2007; Woinarski, Andersen and Murphy, 2020). Many granivorous birds in these areas track resources seasonally in response to this variation (Weier *et al.*, 2019; Hodder, 2020; Collett *et al.*, 2023). At the landscape scale, we found little seasonal change in SBTF habitat use. A small but statistically significant shift away from *E. brownii* and *E. similis* woodlands during the wet season was recorded, which suggests that SBTF may concentrate in *E. melanophloia* woodlands at times of high seed abundance and spread their activity over a wide range of vegetation in times of relative scarcity. However, broadly, this study suggests that at the landscape scale habitat use by SBTF remains largely similar across seasons. At least on short timeframes, of less than one month, SBTF movement at the regional scale is also limited, with almost all radio tracked bird resightings occurring within 4.5 km of the initial capture location. Our results therefore suggest that the majority of SBTF resource tracking occurs on the finer local scale, which is the use of foraging, nesting and drinking sites within an individual's home range over hourly to daily timeframes (Mayor *et al.*, 2009a; Rechetelo *et al.*, 2016a).

The scale of habitat use is an important factor in the conservation management of threatened species (Monroe *et al.*, 2022). Conservation strategies must occur on spatial scales that are aligned with the species habitat use and resource requirements (Fraser *et al.*, 2018). For instance, in savanna woodlands, best practice fire management regimes are characterised by patchy and low-intensity fires, which create a fine-scale mosaic of fire histories (Russell-Smith and Yates, 2007). This approach is particularly advantageous for relatively sedentary granivores, like the SBTF, that depend on a diverse array of grass species within their local habitats to meet their nutritional needs throughout the year (Franklin, 1999).

3.1.5.2 Home range

Our study identified that SBTF predominately exhibit sedentary behaviour with largely stationary home ranges over short timeframes (less than one month). This finding contributes to our understanding of niche separation among sympatric estrildid finches, which are largely differentiated based on differences in their diet and mobility (Franklin *et al.*, 2017; Collett *et al.*, 2022b). Notably, the SBTF's home range and movement characteristics are similar to the closely related and morphologically similar long-tailed finch (*Poephila acuticauda*). Like SBTF, long-tailed finch occupy a similar habitat niche in the savannas of northwestern Australia (Tidemann *et al.*, 1992), have large mostly sedentary home ranges and have similar diel drinking patterns (Collett *et al.*, 2022a). In addition, SBTF share similar wing morphology to long-tailed finch, particularly the wing width-length ratio (Sheard *et al.*, 2020), which is a strong indicator of mobility among estrildid finches (Franklin *et al.*, 2017). Collett et al. (2022b) reported that long-tailed finch diversified their diet in times of resource restriction, which was in contrast to the findings of the sympatric Gouldian finch, a dietary specialist that uses regional nomadism to meet its resource requirements (Bolton *et al.*, 2016; Collett *et al.*, 2023). The home range patterns and wing morphology of SBTF suggest a similar mobility niche to that of long-tailed finch, which is characterised by local-scale resource tracking within large home ranges (on average 319 ha) and limited broader landscape or regional movements (Franklin *et al.*, 2017; Collett *et al.*, 2022b).

A sedentary home range pattern of SBTF with possible diet diversification in times of food scarcity is also supported by our finding that home ranges were stable across seasons. Rechetelo et al. (2016a) similarly reported that there was no significant seasonal variation in home range size of SBTF in the Townsville Plains. The diet of SBTF comprises a wide variety of grass seeds and a minor dietary component of insects when available (Higgins, Peter and Cowling, 2006; Williams *et al.*, 2020). This dietary flexibility suggests that SBTF are likely to use dietary diversification within their local home range rather than broader landscape scale movements to meet their nutritional needs throughout the year (Collett *et al.*, 2022b).

Theory predicts that home range size is proportional to the availability and quality of suitable habitat, with larger home ranges required for areas with a sparser distribution of resources (Doligez & Boulinier, 2008; Haskell et al., 2002; Mitchell & Powell, 2004). We recorded mean AKDE home range and core range estimates that were 6–7 times greater than those recorded by Rechetelo et al. (2016a) for the Townsville Plains population. In addition, our MCP home range estimates obtained using a simple and comparable method were 3.8 times greater than those recorded by Rechetelo et al. (2016a). The observed difference in home range areas may be a function of resource density (Haskell, Ritchie and Olff, 2002). Compared to the Townsville Plains, the Desert Uplands has a sparser distribution of permanent water sources (DES, 2013c), which may require SBTF to extend their home range to gain water (Brandt and Cresswell, 2008). The Desert Uplands also has a lower annual rainfall, which may reduce the seed production of grassy woodlands (Moore *et al.*, 2018). These two environmental differences may require SBTF to have a larger home range to fulfil their resource requirements (Haskell, Ritchie and Olff, 2002; Doligez and Boulinier, 2008).

While the methods used to estimate MCP home ranges were consistent between this study and Rechetelo et al. (2016a), we adopted a more contemporary kernel density estimation (KDE) home range method, which was specifically designed to correct biases introduced by autocorrelated data and small sample sizes (Fleming *et al.*, 2019; Silva *et al.*, 2022). Fleming et al. (2019) used simulation tests to demonstrate that small effective sample sizes, which are common in radio tracking datasets, bias KDE home range estimates lower than true values. The autocorrelation and small sample size weighting methods that we used may explain why our KDE estimates were 6–7 times greater than those of Rechetelo et al. (2016a), while the MCP estimates were less than four times greater.

3.1.5.3 Diel activity patterns

Our results revealed distinct diurnal activity patterns in SBTF. There was a bimodal peak in SBTF foraging activity, with increased foraging in the early morning and late afternoon. A bimodal peak in forging activity has been identified for other Australian estrildid finches (Zann, 1972). However, SBTF specific studies have reported varying results. Using 659 activity observations of SBTF, Mitchell (1996) did not find a daily pattern to foraging activity, while in contrast, a coarse analysis of 620 activity observations by Rechetelo et al. (2016a) suggested a pattern of bimodal foraging similar to our study. For tropical granivorous finches, this bimodal pattern has been suggested to be driven by energy and water conservation requirements during the hottest periods of the day (Brandt and Cresswell, 2009; Manu and Cresswell, 2013).

Drinking activities were highest around the midday period, followed by lower rates of drinking in the late morning, early afternoon, and late afternoon; however, the rates of drinking were not significantly different from the early morning observations in the linear model. This lack of significance may be due to a limited sample size of drinking observations through manual tracking. The timing of drinking in estrildid finches is species-specific (Evans *et al.*, 1985; Collett *et al.*, 2022a). Gouldian finch and pictorella mannikin favour drinking in the morning, while double-barred finch, masked finch (*Poephila personata*), long-tailed finch, zebra finch (*Taeniopygia castanotis*) and star finch (*Bathilda ruficauda*) have been found to drink throughout the day, sometimes with peak visitation rates occurring during the middle of the day (Evans *et al.*, 1985; Collett *et al.*, 2022a). Although we did not find a statistically significant pattern in

SBTF visitation of water sources, the diurnal timing of our drinking records supports the temporal spread of drinking behaviour in SBTF that is likely to be concentrated around the hottest parts of the day. We also found a concentration of SBTF resting (perching and preening) in the middle of the day, which was also reported by Zann (1972) for closely related estrildid finches.

The timing of departure from the nest near first light (approximately 30 minutes prior to sunrise) and re-entry to the nest at last light (approximately 30 minutes after sunset) aligns with previous ethological research on SBTF (Zann, 1972). Nest visitation was nearly only recorded at night during the non-breeding dry season, while during the wet season, there was an increased rate of nest visitation during the day, aligning with daytime nesting activities, such as incubation, occurring during the breeding season (Zann, 1972; Higgins, Peter and Cowling, 2006).

3.1.6 Conclusion

Our study identified strong landscape scale habitat selection by SBTF for grassy eucalypt woodlands where permanent water was accessible and avoidance of non-remnant habitat. This study revealed that SBTF occupy large but sedentary home ranges within which they track resources at a local scale, with limited landscape scale movements, at least over timeframes of less than a month. While our results report on the responses for a single threatened species, these have implications for other resource tracking species more broadly. Importantly, our results highlight the need for local scale habitat management that protects preferred habitats and supports a diversity of suitable forage plants required by the target species.

We found that SBTF foraging activity was most concentrated in the early morning and late afternoon. In addition, while there was no significant effect of time-of-day on drinking behaviour, the highest rates of drinking were observed during the heat of day. Current survey guidelines for SBTF recommend water source monitoring for at least three hours after first light and targeted searches for foraging individuals within close proximity (600 m) of water sources (Department of the Environment, Water, Heritage and the Arts, 2009a). Our findings recommend that the SBTF survey guidelines be updated to include water source watches during the middle of the day and targeted searches for foraging SBTF in the early morning and late afternoon.

3.2 Landscape genetics

This section has been prepared by Skye Davies and Adam Stowe from the Conservation Genetics Laboratory at Macquarie University. Simon Griffiths from the Griffiths Ecology Laboratory at Macquarie University provided supervision and advice. Note: for relevance and continuity, Appendices B and C have been omitted from this report.

3.2.1 Terms and Abbreviations

Term	Definition
Allele	Alternative states of a gene or locus present in a population or species.
Ar	Average number of alleles per locus corrected for sample size.
DAPC	Discriminant Analysis of Principle Components - DAPC teases apart the genetic variation due to differences between groups by controlling for variation within groups.
F	Fixation index – the probability that two alleles in an individual are identical by descent ($F = (H_e - H_o)/ H_e$).
Fitness	The reproductive success of a genotype, typically the number of offspring produced by an individual that survive to reproductive age relative to the average for the population.
F_{IS}	The inbreeding coefficient of an individual with respect to the local subpopulation.
F_{ST}	The inbreeding coefficient of subpopulations relative to the total population (i.e. measure of genetic differentiation between subpopulations).
Genetic drift	A mechanism of evolution by which the allele frequencies of a population change between generations due to random chance.
H_E	Expected heterozygosity – proportion of heterozygous genotypes expected under Hardy-Weinberg equilibrium, calculated as 1.0 minus the sum of the squared gene frequencies (aka, the probability that two gametes, randomly chosen from the gene pool, are of different alleles)
H_O	Observed heterozygosity - observed proportion of heterozygotes, averaged over loci.
K	Inferred number of ancestral populations (genetic clusters)
Microsatellites	Simple sequence repeats of typically 1-6 nucleotide base pairs, originating from DNA polymerase slippage during replication.
MU	Management Unit – demographically independent unit within a species for which birth/death rates rather than immigration influences population demographics.
N_e	Effective population size derived from Linkage Disequilibrium method. The size of an idealized population that would lose genetic diversity due to drift at the same rate as the population under study.
SNP	Single Nucleotide Polymorphism – loci with alleles that differ at a single base, with the rarer allele having a frequency of at least 1% in a population.

3.2.2 Abstract

We investigated the fine-scale genetic structure of the declining and Endangered southern black-throated finch (*Poephila cincta cincta*) in one of its last remaining stronghold populations in the Desert Uplands Bioregion. Using a panel of genome-wide single nucleotide polymorphisms, we found restricted gene flow and lower effective migration between subpopulations only 16 km apart, as well as evidence of isolation by distance and regional philopatry. Small contemporary effective population sizes and a recent demographic history of two genetic bottlenecks (≈ 60 – 100 and 700 years ago) suggest the Desert Uplands population may have a reduced capacity to maintain evolutionary potential in the long term. Given recent evidence from whole-genome sequencing data that the Desert Uplands population is genetically isolated from the other stronghold population in Townsville, future losses of genetic diversity may occur without effective management. Incorporating both genetic and ecological data into management plans would aid conservation planning and help prevent continued declines in the last two remaining stronghold populations of the southern black-throated finch.

3.2.3 Background and Rationale

3.2.3.1 Decline of the southern black-throated finch

The black-throated finch (*Poephila cincta*, hereafter referred to as BTF) is a granivorous (seed-eating) finch endemic to Australia, with two recognised subspecies, a northern (*P. c. atropygialis,* NBTF) and southern form (*P. c. cincta,* SBTF) (Higgins, 2006). SBTF have declined since the early 1900's with the rise of pastoralism, and their range has contracted by more than 80% since the late 1900's (Black-throated Finch Recovery Team (BTFRT), 2007; Franklin, 1999; Reside *et al.*, 2019). The southern subspecies was formerly distributed from north-eastern NSW up to the Atherton Tablelands, but its range is now highly fragmented and restricted to north-eastern QLD (Baldwin, 1976; Schodde and Mason, 1999). Two strongholds remain, the Townsville Coastal Plain, and the Desert Uplands Bioregion where the largest remaining population is found, although smaller, scattered populations outside these strongholds have been recorded (Grice *et al.*, 2023; Vanderduys, Reside, *et al.*, 2016). Although SBTF are listed as 'Least Concern' on the IUCN Red List of Threatened Species (BirdLife International, 2016), SBTF was nationally listed in 2005 as "Endangered" (Environment Protection and Biodiversity Conservation Act 1999), and state-listed in 2016 as "Presumed Extinct" in NSW (Threatened Species Conservation Act 1995). Yet despite increased environmental regulation since SBTF was first listed under these environmental protection laws, the southern subspecies has continued to decline in recent years (Reside *et al.*, 2019). Conserving the last two strongholds is crucial for ensuring the long-term persistence of SBTF.

The major threats facing SBTF are habitat loss and degradation, particularly in the south of its range where agriculture has been more intensive (Reside *et al.*, 2019). SBTF have strict habitat requirements, occupying open grassy woodlands and forests, with regular access to water (Higgins, 2006; Zann, 1976). Larger flocks are associated with generally low vegetation density (low tree and shrub cover) but with a higher vegetation density within the 0-20 cm height class and a higher cover of native grasses, avoiding heavily grazed or cleared areas with low grass diversity (Rechetelo, 2015; Rechetelo *et al.*, 2016; van Osta *et al.*, 2024). The decline of SBTF has been primarily driven by agriculture, clearing of grassy woodlands, habitat degradation by livestock grazing and the spread of invasive species (e.g., exotic grasses) (BTFRT, 2007a; Garnett, Szabo and Dutson, 2011; Shephard, Pridham and Forshaw, 2012; Woinarski and Catterall, 2004). More than half of the remaining habitat for SBTF occurs within mining lease tenures, particularly affecting the larger resident population in the Desert Uplands Bioregion, while the Townsville population is threatened by urbanization and land development (BTFRT, 2020; Mula Laguna, Reside, *et al.*, 2019; Vanderduys, Reside, *et al.*, 2016). Flock sizes have decreased in both populations, but are larger in the Desert Uplands Bioregion, likely due to a higher percentage of remnant vegetation and intact SBTF habitat compared to Townsville (Reside *et al.*, 2019; Mula Laguna *et al.*, 2019). SBTF are also considered

sedentary, maintaining small home range sizes and exhibiting limited post-breeding dispersal, which may exacerbate their risk of location extinction (Higgins, 2006; Garnett, Szabo and Dutson, 2011; Vanderduys, Reside, *et al.*, 2016; van Osta *et al.*, 2024).

3.2.3.2 Knowledge gaps in SBTF conservation

Recovery efforts over the past two decades have aimed at improving ecological knowledge available for SBTF to inform conservation status and guide management strategies (BTFRT, 2007a, BTFRT, 2020). However, the conservation of this subspecies remains hampered by limited information on its ecology and a lack of long-term monitoring in the face of ongoing habitat loss (Reside *et al.*, 2019; Mula Laguna *et al.*, 2019). Data from radio-tracking finches in the Townsville and Desert Uplands strongholds has provided some insight into the movement ecology of SBTF, including home range sizes, habitat preferences and short-term dispersal distances (Mula Laguna *et al.*, 2019; Rechetelo, 2015; Rechetelo *et al.*, 2016; van Osta *et al.*, 2024). However, information on fine-scale functional connectivity and effective dispersal (i.e., dispersal that results in successful reproduction) has been limited to the Townsville population (Tang, 2016; Tang *et al.*, 2017). Using microsatellite markers, evidence of genetic structuring over relatively short distances (10 – 20 km) was found in the Townsville population, with differences in elevation and vegetation structure, along with the Ross River Dam, contributing to altered gene flow between subpopulations (Tang, 2016; Tang *et al.*, 2017).

Further research is needed to assess the genetic diversity and structure of the larger, resident Desert Uplands population of SBTF. As the fundamental source of biodiversity, conserving genetic diversity in wild populations is vital for supporting the long-term persistence of species (Allendorf and Luikart, 2009; Lowe and Allendorf, 2010; Raffard *et al.*, 2019). Reduced gene flow, for example due to habitat fragmentation, can lead to a loss of genetic diversity and an increased risk of inbreeding depression in small, isolated populations, reducing adaptive potential and elevating extinction risk (Frankham, 2005; Frankham *et al.*, 2017). Four genetic 'Essential Biodiversity Variables' (EBVs) were identified by Hoban et al. (2022) as priorities in conservation and include: genetic diversity, genetic differentiation, inbreeding and effective population size (N_e). Quantifying N_e, the size of an idealized population that would lose genetic diversity at the same rate as the population under study, can be used to inform the risk of losing genetic diversity due to drift over the next few generations (Luikart *et al.*, 2010; Tallmon *et al.*, 2010). Population bottlenecks reduce N_e and elevate the risk of genetic erosion through the accumulation of deleterious alleles (Bijlsma and Loeschcke, 2012). Thus, assessing a population's demographic history can identify genetic signatures of bottlenecks which may have implications for long-term persistence, particularly as many avian species currently in decline have unstable demographic histories (Nadachowska-Brzyska *et al.*, 2015). Collectively, examining the genetic diversity, structure and demographic history of SBTF in the Desert Uplands population will improve our understanding of the ecology and genetic health of this subspecies.

3.2.3.3 Objectives and study scope

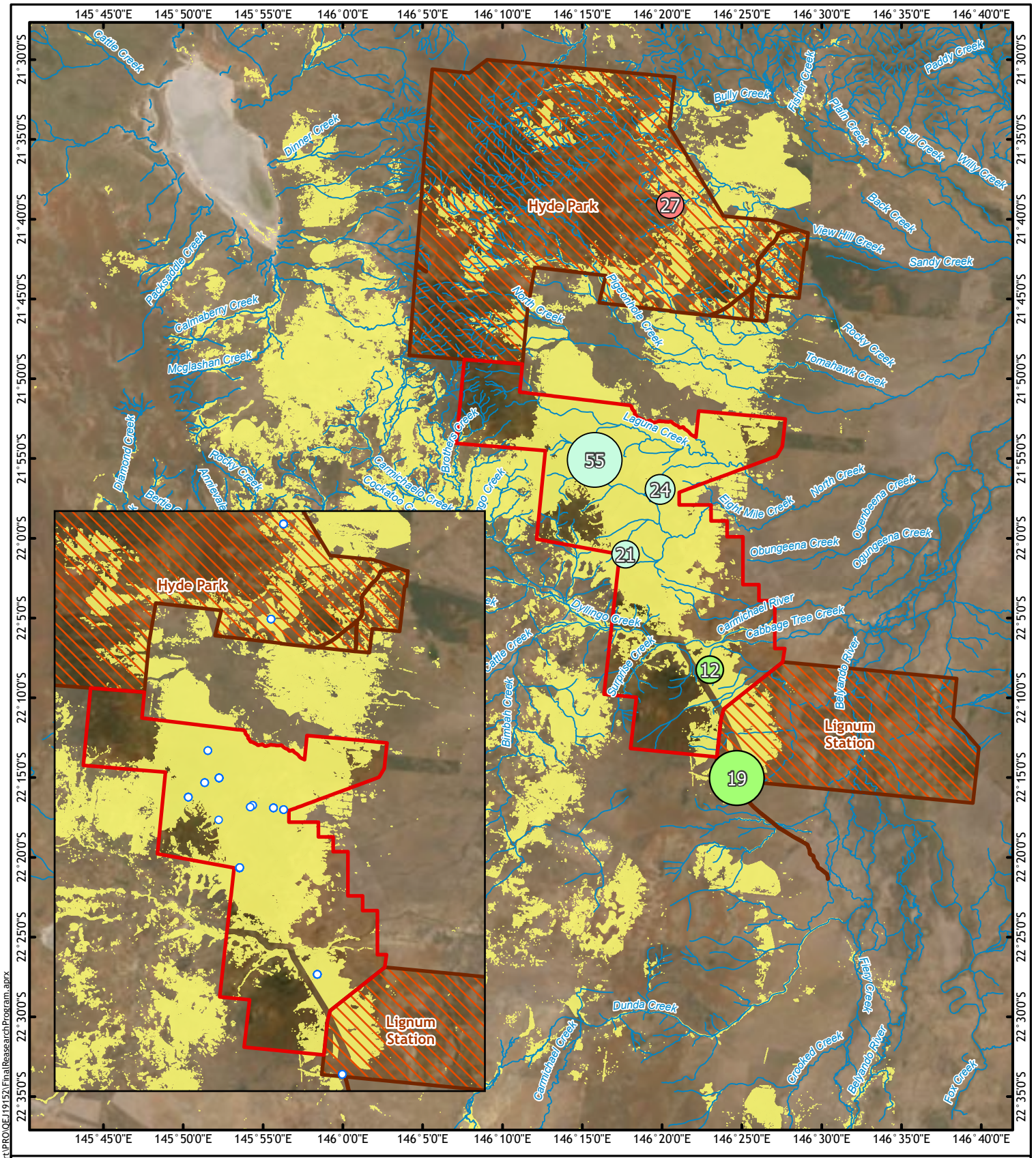
The primary aim of this study was to analyse thousands of nuclear genetic markers (single nucleotide polymorphisms, SNPs) for a population of SBTF located around the Carmichael Coal Mine within the Desert Uplands Bioregion (QLD), to address the following objectives:

1) Assess spatial genetic structure, diversity and site fidelity,
2) Estimate contemporary effective population size,
3) Infer recent demographic history.

3.2.4 Methodology

3.2.4.1 Data collection

158 SBTF DNA samples were collected from three localities (denoted North, Central and South regions, see Figure 3.2-1). Genetic samples were sent to Diversity Arrays Technology (http://www.diversityarrays.com/) for DArTseq genotyping-by-sequencing (GBS). After quality control, 14,788 neutral, autosomal SNPs were retained for 107 unrelated individuals for analyses of genetic structure, diversity, N_e and landscape resistance modelling (14,808 SNPs for 155 samples with related individuals). Sex was assigned to each bird using three sex-associated features (Figure 3.2-2). For inference of demographic history, raw reads were mapped to a reference genome (zebra finch, *Taeniopygia guttata*), retaining 4,438,791 sites after quality control (including 80,382 variant sites).



Legend

Northern Clusters
Low
High

Central Clusters
Low
High

Southern Clusters
Low
High

- Study Area
- Modelled BTF Habitat (>0.6)
- Properties
- Unclustered Genetic Sample Locations
- Watercourse

N

Scale 1:600,000 (A4)

0 9.75 19.5 29.25

Kilometers

Coordinate System: GCS GDA 1994

BARCALDINE

Notes:
Aerial Imagery: © ESRI 2021
Cadastre: © DNRME 2021
Watercourse: © Geoscience Australia 2012
Road: © DNRME 2021

Rev	Description	Drawn	Approved	Date
2	Issued for Use	JY	JMC	30/10/2025
0	Issued for Use	GO	JMC	11/08/2025

FIGURE 3.2-1: GENETIC SAMPLING LOCATIONS

BTFRP Final Report
Bravus

Map Number	Job Number	Rev
1 of 1	QEJ19152	2

Document Path: X:\JOBS\-2019\QEJ19152\GIS\Final Research Program Report\PRO\QEJ19152\FinalReasearchProgram.aprx

E2M Pty Ltd gives no warranty in relation to the data (including accuracy, reliability, completeness or suitability) and accepts no liability for any loss, damage or costs (including consequential damage) relating to any use of the data in this map.

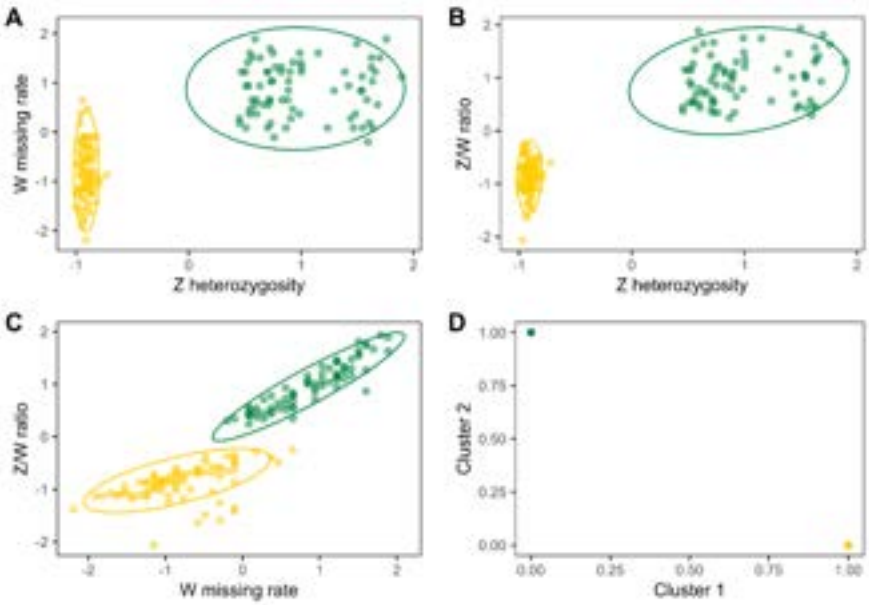


Figure 3.2-2: Sex inference results for black-throated finches. A-C) correlations between pairs of sex-associated features used in the analysis, coloured by sex inferred with *mclust* clustering method (green = males, yellow = females). D) Assignment probability of samples to each cluster (cluster 1 = female, cluster 2 = male). Males are ZZ in birds, and thus are expected to have higher Z heterozygosity, a higher W missing rate and a higher Z/W ratio.

3.2.4.2 Genetic structure and diversity

Genetic structure was analysed using four individual-based approaches: 1) a principal coordinates analysis (PCoA) with *ade4* v 1.7-22 using Euclidian genotypic distance between samples; 2) a discriminant analysis of principal components (DAPC) with *adegenet* v 2.1.10 which identifies genetic clusters by minimizing within-group genetic variation (Jombart and Ahmed, 2011); 3) snmf clustering with *LEA* v 3.14.0 to infer the number of ancestral populations; and 4) the Bayesian clustering approach implemented in STRUCTURE v 2.3 (Jombart, 2008) via *dartR*. Global F_{ST} (fixation index) and F_{IS} (inbreeding coefficient), and pairwise F_{ST} between localities or regions were estimated using *diveRsity* v 1.9.90 (Keenan *et al.*, 2013). To assess recent migration and identify individuals with mixed ancestry, STRUCTURE was run with the 'related' dataset, using regions (north, central, south) as prior site information.

Spatial genetic structure was examined using spatial PCA (sPCA) and spatial autocorrelation analysis (SAA). sPCA detects cryptic spatial structure by decomposing genetic variation into variance and spatial autocorrelation (Moran's index, *I*) with *adegenet*. SAA groups pairs of individuals into specific distance classes and computes the mean genotypic correlation (related to kinship) within each distance class, testing the null hypothesis of no spatial structure (i.e., random mating) using permutation tests. Estimated effective migration surfaces (EEMS) were mapped for the study area using *reemsplot2* v 0.1.0 (Petkova, 2024). EEMS infers deviation from continuous gene flow across the landscape to identify areas of higher or lower than average historic migration (Petkova, Novembre and Stephens, 2016).

Sex-biased dispersal was assessed using kinship permutation tests and the sex-biased dispersal tests described in Goudet et al. (2002). Mean kinship was compared between sexes, localities and regions using the KING-derived kinship coefficients with GENESIS, accounting for genetic structure. The sex-biased dispersal tests assessed whether four statistics were significantly different between males and females;

F_{ST}, F_{IS}, mean corrected assignment index ($mAIc$) and variance ($vAIc$). Sex was reshuffled for each permutation, keeping sex ratios and individual locations constant.

Genetic diversity metrics were calculated per region (North, Central, South) with *diveRsity*, including allelic richness (Ar), observed heterozygosity (H_o), unbiased expected heterozygosity (H_e), and inbreeding coefficient (F_{IS}).

3.2.4.3 Contemporary effective population size

Contemporary N_e was estimated using the bias-corrected linkage disequilibrium (LD) method with *NeEstimator* v 2.1 (Waples and Do, 2008). As LD is built up over generations, this N_e reflects the short-term harmonic mean across a few recent generations. A minor allele frequency threshold of P_{crit} = 0.02 to 0.03 was used, and bias-corrected jack-knife 95% confidence intervals were calculated about the mean (Waples and Do 2010). Given overlapping generations, this raw estimate was interpreted as the effective number of breeders in the parent generation which produced the cohorts sampled ($Raw\hat{N}_b$). $Raw\hat{N}_b$ was adjusted to account for this bias and converted to N_e using two life history traits (age at sexual maturity (α) and adult lifespan (AL)). To account for the limited ecological knowledge on life span available for SBTF, we converted N_e using α = 0.5 or 1 year, and AL = 4 to 8 years, and report results for these lower and upper estimates (Shephard, 1989). Based on the results of the genetic structure analyses, we also estimated N_e for the North-Central and South subpopulations separately. To enable more accurate comparisons between these two groups due to sample size differences (North-Central = 87, South = 20), we used a bootstrapping approach for the North-Central group by randomly subsampling 20 individuals (i.e., similar sample size to the South) for each of 100 bootstrap replicates, and recalculated N_e.

3.2.4.4 Demographic history

Demographic history was inferred using two site frequency spectrum (SFS) based approaches; Stairway Plot v 2.0 (Liu and Fu, 2015) and ðaði v 2.3 (Gutenkunst *et al.*, 2009). For both approaches, 1-dimensional SFS's were generated with ANGSD v 0.940, based on individual genotype likelihoods, and folded as we lacked information on ancestral allelic states.

Stairway Plot was run with default settings, with sequence length (L) calculated as the total number of variant and invariant sites after filtering. As the germline mutation rate (per site per generation, μ_G) is unknown for BTF, conversions used the estimated germline mutation rate (per site per generation, μ_G) for zebra finch, 5.85×10^{-9} (Bergeron *et al.*, 2023), and a generation length of 2 years (Bird *et al.*, 2020). We also tested a longer generation length of 3.5 years (Garnett, Szabo and Dutson, 2011). Demographic inference with ðaði was performed with *dadi-cli* (Huang *et al.*, 2023). Five 1D models were tested, described in Table 3.2-1. Parameters for each model were optimized using 5 independent runs and 5000 optimizations per run, until parameters converged across runs. Parameters for the best-fit model were converted into measures of N_e and time in years. Using 100 bootstrapped SFS's, we computed the 95% confidence intervals around parameter estimates.

Table 3.2-1: 1-dimensional demographic models used in the δaδi analysis to infer demographic history of black-throated finch in their southernmost range of the Desert Uplands Bioregion. T is in time of $2N_{ref}$ generations ago, and nu are ratios of effective population size relative to N_{ref}.

Model	Description	Parameters
SNM	Standard neutral model No population size change.	None
2EM	Two epoch model Instantaneous size change to $\boldsymbol{nuF}$ at time $\boldsymbol{T}$.	$\boldsymbol{nuF}$, $\boldsymbol{T}$
EGM	Growth model Exponential growth/decline to $\boldsymbol{nuF}$ beginning at time $\boldsymbol{T}$.	$\boldsymbol{nuF}$, $\boldsymbol{T}$
BGM	Bottlegrowth model Instantaneous size change to $\boldsymbol{nuB}$ followed by exponential growth/decline to $\boldsymbol{nuF}$ starting at time $\boldsymbol{T}$.	$\boldsymbol{nuB}$, $\boldsymbol{nuF}$, $\boldsymbol{T}$
3EM	Three epoch model Instantaneous size change to $\boldsymbol{nuB}$ of duration $\boldsymbol{TB}$, followed by population recovery to $\boldsymbol{nuF}$ starting at time $\boldsymbol{TF}$.	$\boldsymbol{nuB}$, $\boldsymbol{nuF}$, $\boldsymbol{TB}$, $\boldsymbol{TF}$

3.2.5 Key Findings

3.2.5.1 Fine-scale genetic structure of SBTF in the lower Desert Uplands Bioregion

Individual- and population-based approaches for inferring genetic structure indicated restricted gene flow among SBTF in the study area. Pairwise F_{ST} was highest between North and South sub-populations, but was less than 0.02, indicating drift connectivity (panmixia) between regions (i.e., > 1 migrant per generation) (Lowe and Allendorf, 2010). Pairwise F_{ST} was significantly different from 0 between all regions but was highest between North and South (F_{ST} = 0.018, p < 0.001), followed by Central and South (F_{ST} = 0.0136, p < 0.001), and North and Central (F_{ST} = 0.0057, p < 0.001). Measures of genetic diversity were similar between regions, although Ar and F_{IS} were both significantly higher in the Central sub-population Table 3.2-2.

Table 3.2-2: Measures of regional neutral autosomal genetic diversity for black-throated finches in the Desert Uplands Bioregion. Ar = allelic richness (accounting for differences in sample sizes, n), H_o = observed heterozygosity, H_e = unbiased expected heterozygosity, and F_{IS} = inbreeding coefficient. Bootstrapped 95% C.I.'s are shown in brackets (999 bootstraps). Declines in Ar (diversity at the allelic level) may impair a populations capacity to adapt to environmental changes, while declines in H_e (diversity at the genotypic level) have been linked with decreases in the average fitness of individuals (Fisher, 1930; Reed and Frankham, 2003). Management plans aimed at conserving genetic diversity should consider both types for long-term population persistence (Greenbaum *et al.*, 2014).

	n	Ar	H_o	H_e	F_{IS}
Region					
North	20	1.72 (1.67, 1.76)	0.158	0.188	0.106 (0.066, 0.098)
Central	67	1.81 (1.77, 1.83)	0.163	0.194	0.133 (0.115, 0.128)

	n	Ar	H_o	H_e	F_{IS}
South	20	1.71 (1.67, 1.75)	0.162	0.191	0.101 (0.061, 0.092)
Mean		1.74	0.161	0.191	
Global F_{IS}					0.160 (0.157, 0.163)

For individual-based analyses, the number of genetic clusters varied depending on the approach, but generally suggested finches south of the Carmichael River form a distinct genetic unit (Figure 3.2-3B & D). The assignment tests indicated mixed ancestry within the South sub-population, unidirectional gene flow from Central→South, and no recent gene flow between North and South sub-populations (Table 3.2-3). For example, all Central and most North birds assigned to the Central subpopulation, suggesting higher gene flow between these two regions, while fewer South individuals assigned to Central and none to North (Table 3.2-3). There was no strong evidence of sex-biased dispersal based on the kinship permutation tests or SAA. For Goudet's sex-biased dispersal tests, only mean variance in AIc and F_{IS} were significantly higher among females ($p < 0.05$) (based on assigning individuals to their region of origin).

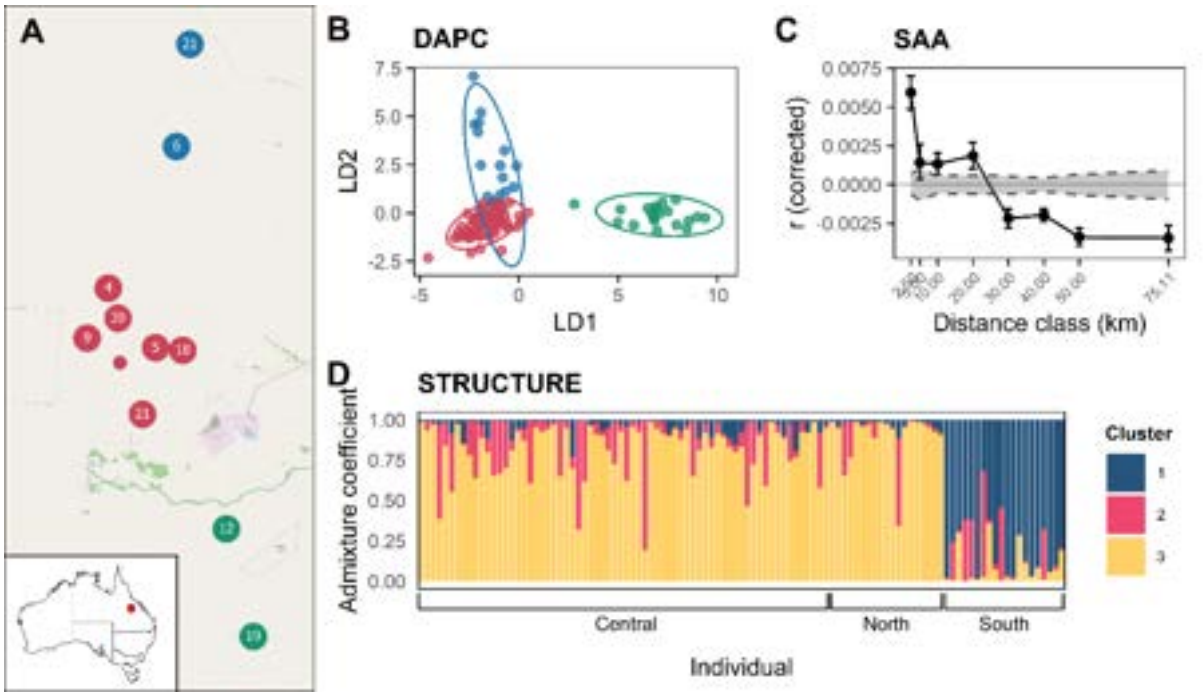


Figure 3.2-3: A) Map of southern black-throated finch samples collected around the Carmichael Coal Mine and coloured by region (blue = North, red = Central, green = South). B) Discriminant analysis of principal components (DAPC) showing the first two discriminant functions, with individuals coloured by region of origin. C) Spatial autocorrelation analysis (SAA) showing mean within distance-class genotypic correlation (related to kinship). Irregular distance classes were chosen to maximise sample size per distance class, solid error bars denote 95% bootstrapped C.I. about the mean r within each distance class (999 bootstraps), dashed line denotes the 95% C.I. for the hypothesis of 'no spatial structure' (999 permutations). D) STRUCTURE plot of admixture coefficients for each individual, assuming K = 3 ancestral populations (highest delta K).

Table 3.2-3: Percentage of samples (including related individuals) either assigning correctly to the region they were sampled in (i.e., no mixed ancestry), assigning to a different region (i.e., potential 1st generation migrant to the region they were sampled in), or having a parent, grandparent or great grandparent that migrated from a different region (i.e., recent ancestry in a different region, back to three generations ago). Only results with posterior probabilities of assignment P > 0.5 or 0.9 are shown.

	North (n = 27)		Central (n = 97)		South (n = 31)	
	P > 0.9	P > 0.5	P > 0.9	P > 0.5	P > 0.9	P > 0.5
Assigned to region of origin	7.4%	7.4%	97.9%	100.0%	22.6%	32.3%
Potential migrant from a different region						
North			0.0%	0.0%	0.0%	0.0%
Central	81.5%	81.5%			29.0%	35.5%
South	0.0%	0.0%	0.0%	0.0%		
Potential parent from a different region						
North			0.0%	0.0%	0.0%	0.0%
Central	0.0%	0.0%			0.0%	9.7%
South	0.0%	0.0%	0.0%	0.0%		
Potential grandparent from a different region						
North			0.0%	0.0%	0.0%	0.0%
Central	11.1%	11.1%			16.1%	22.6%
South	0.0%	0.0%	0.0%	0.0%		
Potential great grandparent from a different region						
North			0.0%	0.0%	0.0%	0.0%
Central	0.0%	0.0%			0.0%	0.0%
South	0.0%	0.0%	0.0%	0.0%		

After updating KING kinship coefficients to account for genetic structure, we detected 45 pairs of first-degree relatives (full-sibs or parent-offspring, $0.375 > k > 0.1875$) across 48 samples, and 61 pairs of second-degree relatives (half-sibs or grandparent-offspring, $0.1875 > k > 0.09375$) across 74 samples, following the midpoint cut offs in Iacchei et al. (2013). First-degree relatives were detected across years and wet/dry seasons, however no first- or second-degree relatives were found between samples in different regions. This may reflect oversampling family groups within a breeding season, or potentially limited recent interbreeding between finches in different regions (i.e., regional or site fidelity).

Evidence of isolation by distance IBD and high within-site kinship also supports site fidelity. Kinship permutations tests indicated significant site fidelity, at both local and regional scales, with mean within-site and within-region kinship being significantly higher than between-site and between-region kinship, respectively ($p < 0.001$, Figure 3.2-4). Mean kinship was also significantly higher in the North ($\mu = 0.0075$)

and the South (μ = 0.0070) compared to the Central region (μ = 0.0019, $p < 0.001$) (Figure 3.2-4), but note that sampling within each region varied by season (wet or dry) and sample sizes differed between regions.

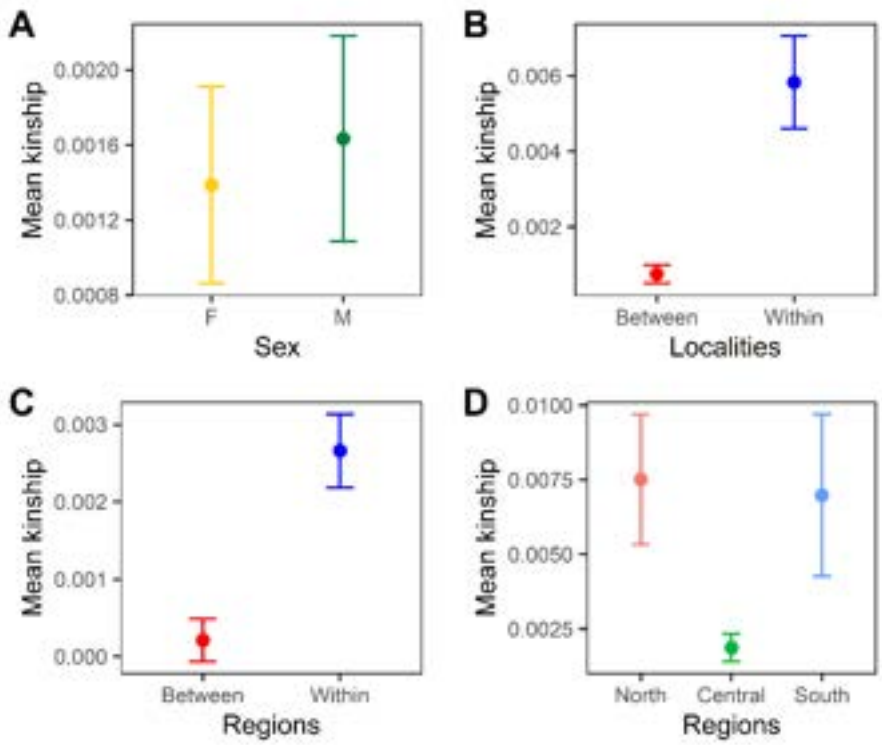


Figure 3.2-4: Mean kinship among black-throated finch compared A) between sexes, B) within versus between localities, C) within versus between regions, and D) between each region. Kinship was estimated with KING accounting for genetic structure with PC-AiR and PC-Relate, using neutral, autosomal loci and the 'unrelated' dataset (no full- or half-sibs). Note that samples size is larger in the Central region.

SAA revealed a strong signal of IBD when all 'unrelated' samples were pooled, with mean genotypic correlation '*r*' among samples within the 2.5 km distance class (i.e., within-site comparisons) being significantly higher than expected under conditions of random mating (Figure 3.2-3C). This pattern of IBD was largely driven by differences between regions but was also detectable when examining the Central subpopulation alone. The sPCA also revealed significant global structure ($p < 0.01$), indicating high variance and positive spatial autocorrelation (Figure 3.2-5).

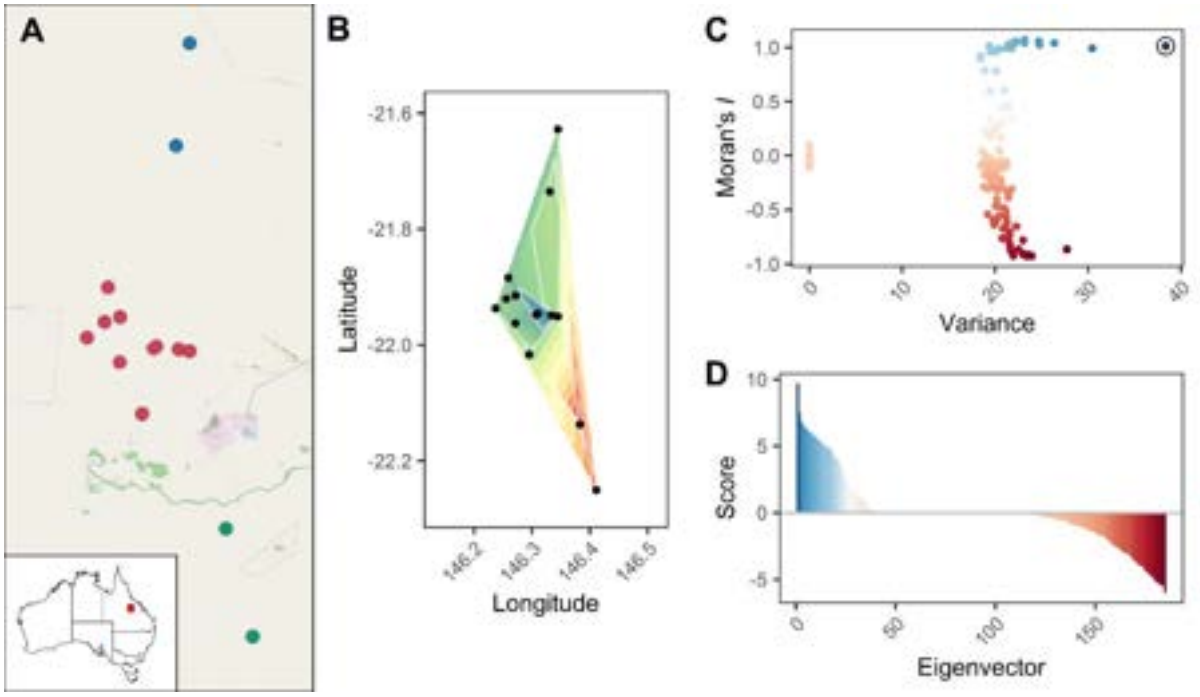


Figure 3.2-5: Spatial principal coordinates analysis (sPCA) for black-throated finches across their distribution in the lower Desert Uplands Bioregion. A) Geographic location of samples coloured by region (blue = north, red = central, green = south). B) Map of interpolated individual scores (blue = low, red = high) based on the first PC (first global score). Contour lines that are closer together indicate greater genetic differentiation between localities. C) Plot of decomposed genetic variation into variance (due to population structure) and Moran's *I* (spatial autocorrelation) (blue = global structure, red = local structure), with the top global score circled. D) Distribution of eigenvectors from the most positive (blue = global structure) to negative (red = local structure).

Using EEMS, the estimated effective migration rate was significantly lower than the average migration rate between the Central and South subpopulations (Figure 3.2-6).

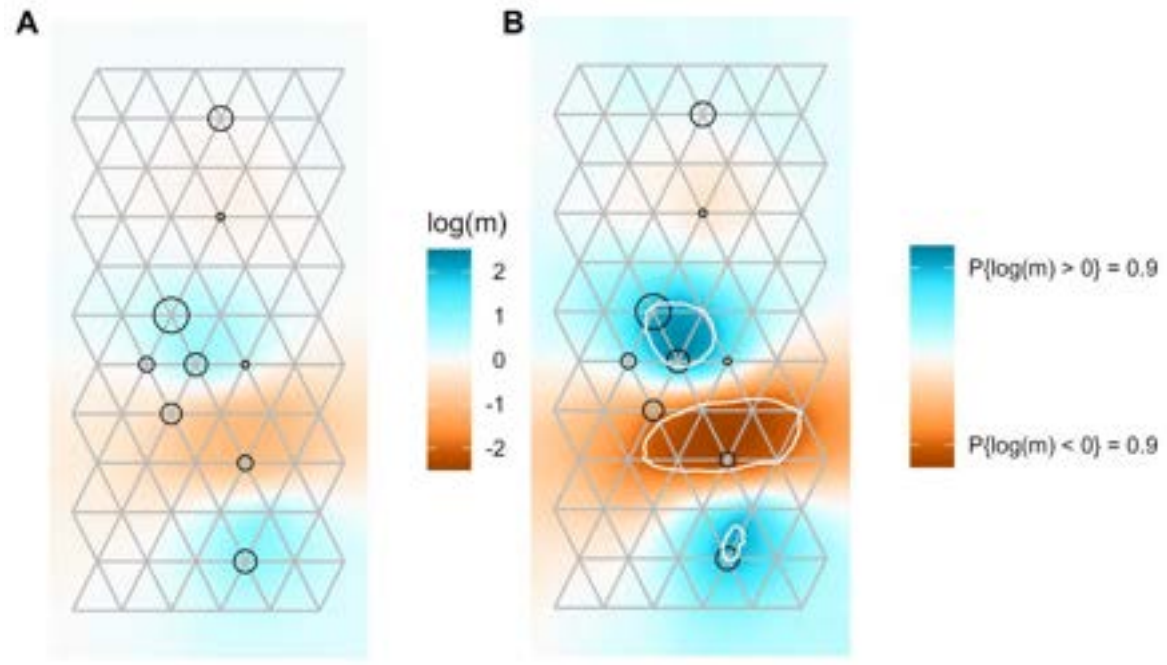


Figure 3.2-6: Estimated effective migration rates calculated by EEMS. A) Plot of interpolated posterior mean migration rates *m* (log_{10} scale) showing areas of higher than expected (blue) or lower than expected migration rates (orange). Circles indicate sampling locations with size depicting the number of samples per deme. B) Plot showing highlighted regions (white outline) where effective migration rates are significantly higher (blue) or lower (orange) than the overall average rate of migration.

Overall, these results indicate sufficient levels of migration over many generations have maintained similar allele frequencies between regions, but other factors may contribute to recent fine-scale genetic structuring over a relatively small spatial scale (70 km) for SBTF. These results are consistent with previous observations and genetic research for SBTF. Restricted gene flow was observed between SBTF in the Townsville population over relatively short distances (10-20 km) based on microsatellite markers (Tang, 2016; Tang *et al.*, 2017). SBTF are also philopatric, known to return to the same nest over multiple years (Isles, 2007; NRA, 2005), and juveniles remain within family groups for several months after fledging which may promote site fidelity through social learning (Higgins, 2006; R Zann, 1976). Home range estimates for SBTF in the Desert Uplands are around 320 ha (using a kernel density estimation method), and short-term dispersal distances are generally small (up to 4.5 km) (van Osta *et al.*, 2024). Radio tracking also suggests Desert Uplands SBTF tend to remain within 4.2 km of water and may favour dietary diversification over long-distance dispersal during the dry season (van Osta *et al.*, 2024). Given the higher level of genetic differentiation between finches in the northern and southern regions of the study area, and limited recent gene flow between regions, there is likely restricted dispersal between subpopulations of SBTF that are 16—45 km apart. However, geographic distance was similar between the North and Central regions (18 km) and Central and South (16 km), and patterns of IBD were largely driven by differences in mean kinship between regions. Thus, landscape features may play a larger role in restricting gene flow for the SBTF than geographic distance alone.

3.2.5.2 Small contemporary N_e

Contemporary N_e over a few recent generations was relatively small for the studied SBTF population. Depending on the life-history traits used to account for overlapping generations, estimates of N_b ranged

from around 1,110 – 1,610 and for N_e 940 – 1,440 (Table 3.2-4). When considering South and North-Central groups as two distinct genetic units, N_e for the South group (n = 20) ranged from 290 to 360 (lower 95% C.I. from 210 to 260; upper 95% C.I. from 490 to 610; P_{crit} = 0.03) (Table 3.2-5). For the North-Central group (n = 87), N_e ranged from 1,280 to 1,610 (lower 95% C.I. 1,010 to 1,270, upper 95% C.I. 1,750 to 2,200; P_{crit} = 0.02) (Table 3.2-5). Across 100 bootstrap replicates, the average N_e for the North-Central group using an equivalent sample size as in the South ranged from 970 to 1,220 depending on the life-history traits used. N_e for the North-Central group was larger than the South in 100% of bootstraps, and twice as large in 90% of bootstraps.

These estimates of N_e are near the recommended minimum level of 1,000 to maintain evolutionary potential in the long term (Frankham, Bradshaw and Brook, 2014). Given that adult census sizes (N_c, total number of potential breeders) are poorly known for SBTF, this study provides a baseline for continued monitoring of the effective number of breeders and genetic health of the Desert Uplands population.

Table 3.2-4: Estimates of the effective number of breeders (N_b) for all SBTF samples in the Desert Uplands population, accounting for overlapping generations, and effective population size (N_e), under different estimates of age at sexual maturity (α) and adult lifespan (AL) in years, rounded to the nearest 10. Lower and upper give the 95% jack-knifed confidence intervals.

	N_e		N_b	
Mean	α = 0.5	α = 1.0	α = 0.5	α = 1.0
AL = 4	1,260	1,440	1,470	1,360
AL = 5	1,220	1,370	1,510	1,390
AL = 6	1,190	1,320	1,550	1,420
AL = 7	1,170	1,290	1,580	1,450
AL = 8	1,150	1,260	1,610	1,470
Lower				
AL = 4	1,030	1,180	1,210	1,110
AL = 5	1,000	1,130	1,240	1,140
AL = 6	970	1,090	1,270	1,170
AL = 7	960	1,060	1,290	1,190
AL = 8	940	1,030	1,320	1,210
Upper				
AL = 4	1,610	1,840	1,880	1,740
AL = 5	1,560	1,760	1,930	1,780
AL = 6	1,520	1,690	1,980	1,820
AL = 7	1,490	1,650	2,020	1,850
AL = 8	1,470	1,610	2,050	1,880

Table 3.2-5: Mean estimates of the effective number of breeders (N_b) for the North-Central and South subpopulations of SBTF in the Desert Uplands Bioregion, accounting for overlapping generations, and effective population size (N_e), under different estimates of age at sexual maturity (α) and adult lifespan (AL) in years, rounded to the nearest 10. Estimates for the North-Central subpopulation are based on a P_{crit} of 0.02 and for the South 0.03; based on the P_{crit} rule of thumb proposed by Waples and Do (2010) for n ≤ 25 ($1/2n < P_{crit} < 1/n$).

	N_e		N_b	
North-Central	α = 0.5	α = 1.0	α = 0.5	α = 1.0
AL = 4	1,400	1,610	1,640	1,520
AL = 5	1,360	1,530	1,690	1,550
AL = 6	1,330	1,480	1,730	1,590
AL = 7	1,300	1,440	1,760	1,620
AL = 8	1,280	1,400	1,790	1,640
South				
AL = 4	320	360	370	340
AL = 5	310	350	380	350
AL = 6	300	330	390	360
AL = 7	290	320	400	360
AL = 8	290	320	400	370

3.2.5.3 Genetic signatures of recent population bottlenecks

We carried out demographic history analysis for all samples pooled together ('unrelated' dataset, subset of 20 samples per region). Although we found evidence of fine-scale genetic structure, we assumed the population was panmictic overall with respect to allele frequencies, given the relatively low global F_{ST} and pairwise F_{ST} between regions (< 0.02), indicating drift connectivity (Lowe and Allendorf, 2010). The Stairway plot analysis inferred a historical population expansion beginning ≈ 367,000 years ago and lasting ≈ 127,000 years (2× increase from ancestral N_e) (Figure 3.2-7). Median N_e stabilized until a bottleneck beginning ≈ 700 years ago (82% decrease) followed by a more recent bottleneck ≈ 100 years ago (further 63% decrease from the first bottleneck) (Figure 3.2-7).

The best fit model in ðaði (3EM model, ΔAIC > 10) inferred a historical population expansion beginning ≈ 321,000 years ago (95% C.I. = 103,000 – 539,000) to around 2× the ancestral N_e, and a recent bottleneck ≈ 60 years ago (95% C.I. = 50 – 70) (Figure 3.2-7). ðaði detected the most recent population decline compared to Stairway, with N_e dropping to ≈ 900 within the last 30 generations (assuming a generation length of 2 years). With a longer generation length of 3.5 years, the expansion period detected by Stairway began ≈ 524,000 years ago, the 1st bottleneck ≈ 1,600 years ago and the 2nd bottleneck ≈ 200 years ago. For ðaði, the expansion began ≈ 562,000 years ago and the bottleneck ≈ 100 years ago.

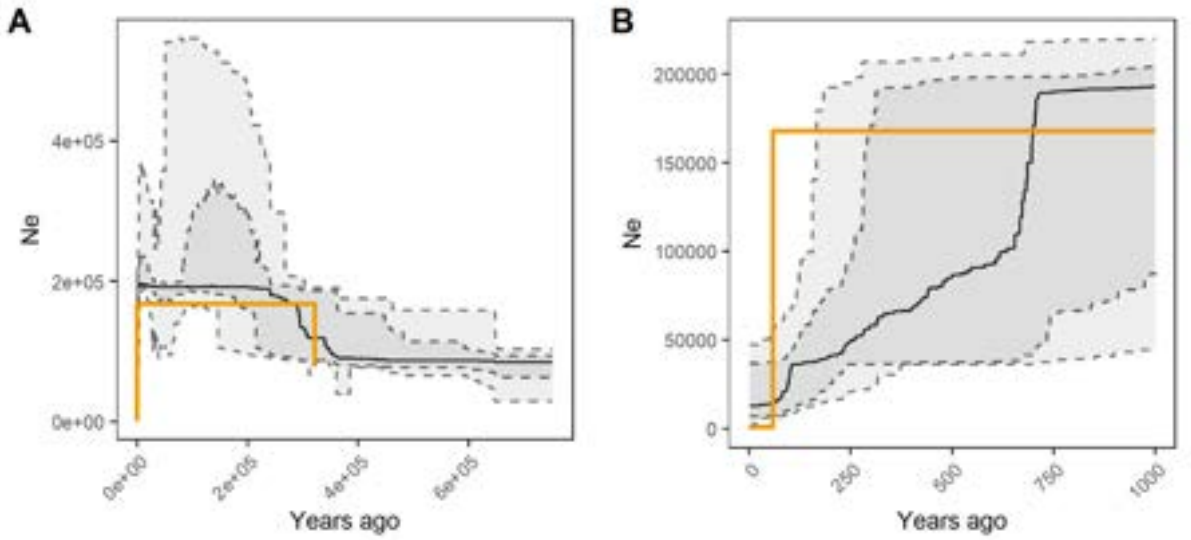


Figure 3.2-7: Stairway plot showing historical changes in the median effective population size (N_e) of black-throated finches in the southern Desert Uplands Bioregion, over A) the last million years and B) the last 1,000 years, with shaded polygons representing 95% C.I.'s (light grey) and 75% C.I.'s (dark grey). Demographic events inferred by ðaði (best-fit model = 3 epoch) are indicated in orange. Results for both approaches are shown based on a mutation rate per site per generation of 5.85×10^{-9} and a generation length (g) of 2 years.

The recent demographic history reconstructed by Stairway and ðaði reflects range contractions for SBTF. The extent of occurrence (EOO) of SBTF has contracted by 50-80% since the late 1900's, and likely over 90% since pre-European times (BTFRT, 2007; Franklin, 1999; Reside *et al.*, 2019). These range contractions are reflected in the recent genetic bottlenecks we detected, with the Stairway analysis inferring a ~60% decrease over the last 100 years, and a more distant bottleneck around 700 years ago (~80% decrease from expansion era N_e). These declines coincide with the European colonization of Australia and the rise of pastoralism over the past 200 years. A previous study on the Townsville population of SBTF using 18 microsatellite markers did not detect any signatures of the population bottlenecks we detected with thousands of SNPs (Tang, 2016; Tang *et al.*, 2017). Future research using whole genome sequencing (WGS) will enable more accurate reconstruction of the recent decline of SBTF, for example by assessing runs of homozygosity across the genome.

The population expansion that began 300,000-500,000 years ago as inferred by the Stairway plot and ðaði analyses is consistent with expansions occurring in other bird species during the early to mid-Pleistocene period, although cycles of expansion-contractions are common (Nadachowska-Brzyska *et al.*, 2015). In Australia, cyclic aridity during the Pleistocene is thought to have driven avian speciation in isolated refugia zones following range contractions (Kearns, Joseph and Cook, 2010). Based on 30 nuclear loci, population divergence estimates place the divergence between SBTF and long-tailed finches (LTF, *P. acuticauda* and *hecki*) within the Pleistocene, around 600,000 years ago (Jennings and Edwards, 2005). Thus, the expansion event we detected with Stairway and ðaði may reflect the expansion of SBTF after diverging from LTF across the Carpentaria Barrier.

3.2.6 Conclusions

In summary, the key findings of the genetic study on SBTF located around the study area in the lower Desert Uplands Bioregion were:

1) Restricted gene flow over relatively small geographic scales (16—45 km) and evidence of site or regional philopatry, with no strong evidence of sex-biased dispersal

2) Relatively small contemporary N_e, and a smaller N_e in the South subpopulation

3) Evidence of a recent and severe population bottleneck reflecting range contractions since the late 1900s

These findings provide critical insights into the population dynamics and genetic health of SBTF in the Desert Uplands population, with broader implications for the southern subspecies across its range. The relatively small N_e we estimated for this population, and evidence of severe genetic bottlenecks (around 70 – 100 and 1,000 years ago), may reflect similar declines in other populations, including smaller, scattered populations outside the two strongholds (Townsville and Desert Uplands). Given the restricted gene flow detected between subpopulations over small spatial scales in both the Desert Uplands and Townsville stronghold, loss of genetic diversity due to drift is likely to continue in the future.

3.2.7 Recommendations for management

This study provides baseline data on effective population sizes, genetic diversity and spatial patterns of genetic variation for SBTF in the lower Desert Uplands population. Management of this stronghold population would benefit from continuing to monitor levels of genetic diversity and N_e over time, particularly for the smaller, potentially demographically isolated subpopulation south of the Carmichael River. The small contemporary N_e of the South subpopulation, coupled with a lower effective migration rate between the South and North-Central groups, may increase the risk of inbreeding depression and loss of adaptive potential over time. Management should also consider the level of site or regional philopatry in SBTF, for example by weighing the relative benefits of habitat recovery (e.g., optimizing habitat corridors between the North-Central and South subpopulations) versus carrying out translocations to facilitate gene flow.

Management plans for SBTF also need to consider the scale at which management is undertaken within the Desert Uplands population and whether subpopulations need to be classified as distinct ‘management units’ (MU’s). Neutral genetic markers can be used to designate MU’s, based on whether the level of genetic divergence between two populations means they are demographically independent (Palsbøll, Berube and Allendorf, 2007). However, this threshold varies depending on the conservation context and species-specific traits. In the current study, although historical gene flow was high enough to maintain similar allele frequencies across the study area, recent interbreeding and effective migration rates were lower between the North-Central and South groups. Combining demographic and genetic modelling will enable a more accurate assessment of whether these two groups should be managed as separate units (Palsbøll, Berube and Allendorf, 2007). This will help to ascertain the level of genetic divergence needed to warrant MU designation, in order to utilize conservation resources more effectively while addressing the unique management needs of each subpopulation.

Lastly, further genetic research into the Desert Uplands SBTF population could benefit from the use of whole genome sequencing (WGS) data. For example, WGS data can be used to inform levels of both ‘realised’ and ‘potential’ genetic load within SBTF populations, i.e., the reduction in fitness due to the accumulation of deleterious mutations (Haldane, 1937; Muller, 1950). ‘Realised’ genetic load is the proportion of total genetic load expressed in the current population and is shaped by short-term demographics (e.g., recent inbreeding), while ‘potential’ genetic load informs on potential fitness losses

in future generations (Mathur and DeWoody, 2021; Mathur *et al.*, 2023). Population declines can increase the level of realised genetic load in wild populations, and thus assessing genetic load can help ensure management plans for the Desert Uplands SBTF population remain adaptive in the face of ongoing environmental change. WGS data can also be used to more accurately reconstruct recent demographic history by examining runs of homozygosity (ROH) across the genome.

3.3 Dispersal movements of southern black-throated finch over three years

3.3.1 Abstract

Understanding the mobility and dispersal capabilities of threatened species is crucial for assessing the connectivity of contemporary populations, particularly considering the escalating threats posed by habitat clearing and degradation. The Endangered southern black-throated finch (SBTF; Poephila cincta cincta) may be impacted by genetic isolation among populations; however, the specific dispersal patterns and barriers to dispersal have yet to be clearly defined. In this study, we evaluated the long-term movement and dispersal of SBTF by examining individuals recorded over a three-year mist netting and colour banding program across a 75,000 ha study area. Strong evidence was found for dispersal among water sources within a 15,000 ha area of interconnected habitat in the northern portion of the study area, while there was no evidence of movement between northern and southern populations of SBTF that are separated by approximately 16 km. Our results suggest that over long periods, of up to three years, SBTF move among water sources at a local scale, while broader landscape or regional scale movement of SBTF were rare. Our results also suggest that habitat clearing and degradation can fragment populations of a moderately mobile bird, restricting contemporary gene flow. Our research highlights the need to manage habitat to retain large interconnected tracts of habitat that promote gene flow of SBTF and habitat management of the remnant vegetation within and surrounding the study area to mitigate isolation of SBTF populations.

3.3.2 Introduction

Species' mobility and dispersal capacity are critical factors influencing their genetic diversity and potential to adapt to environmental changes (Le Galliard, Massot and Clobert, 2012). Dispersal varies from small-scale movements in sedentary species through to long-distance migratory and nomadic movements across continental scales (Sheard *et al.*, 2020). The estrildid finches are a speciose group of birds across Africa, south Asia and Australasia (Arnaiz-Villena *et al.*, 2009) that display a wide range of mobility strategies. These species range from the highly nomadic Gouldian finch (*Erythrura gouldiae*), which have movements across northern Australia (Bolton *et al.*, 2016; Collett *et al.*, 2022b) to the highly sedentary rock firefinch (*Lagonostica sanguinodorsalis*) that has home range size of approximately 8 ha (Brandt and Cresswell, 2008).

Understanding the mobility and dispersal capacity of estrildid finches is important to understand the connectivity of their contemporary populations, particularly in regions where habitat fragmentation may have disrupted the continuity of species habitats (Segelbacher *et al.*, 2010). Differences in the impact of biogeographic barriers to species movement on species' population structuring is evident among the estrildid finches of northern Australia. Rollins, Svedin and Griffith (2012) found that the Ord Arid Intrusion, which is a minor biogeographic barrier dividing savannas of north-western Australia (Bowman *et al.*, 2010), has restricted geneflow of the long-tailed finch (*Poephila acuticauda*), resulting in subspeciation. In contrast, the highly mobile Gouldian finch are able to cross this biogeographic barrier, resulting in a single panmictic population in north-western Australia (Bolton *et al.*, 2016).

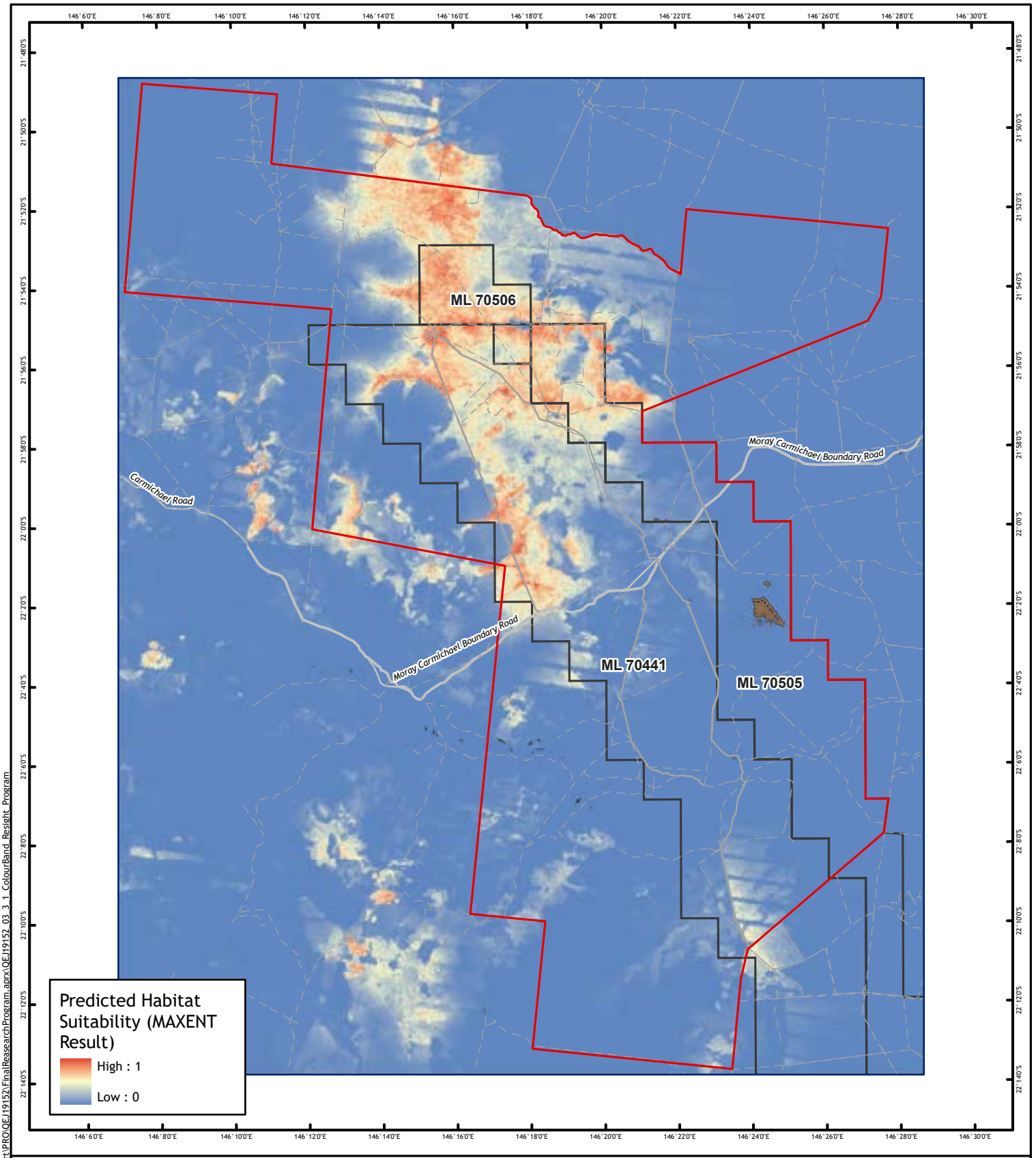
The nationally Endangered southern black-throated finch (SBTF; *Poephila cincta cincta*) is another estrildid finch that may be affected by genetic isolation of populations resulting from habitat clearing and degradation across its range (Vanderduys *et al.*, 2016a; Mula Laguna *et al.*, 2019). Tang (2016) identified spatial structuring of a SBTF population across small scales around the Ross River Dam, which was suggestive of contemporary habitat fragmentation restricting genetic flow. However, Tang (2016) also found evidence of genetic exchange across SBTF populations throughout the species distribution and occasional long distance movements of up to 17 km recorded by Rechetelo *et al.* (2016a) are evidence of

some degree of dispersal occurring in the species. However, the spatial scale at which SBTF disperse and potential barriers to their dispersal is currently unclear. The aim of this study was to evaluate the long-term movement and dispersal patterns of SBTF. To do this, we examined the movements of individual SBTF recorded over a three-year mist-netting and colour-banding program across a 75,000 ha study area.

3.3.3 Methods

3.3.3.1 Study area

We conducted this study in a 75,000 ha section of the Moray Downs property within Queensland, Australia (Figure 3.3-1). The study area occurs at the boundary of the Desert Uplands and Brigalow Belt Bioregions, with vegetation communities broadly reflecting this divide (Figure 3.3-1). Within the western portion of the study area (the Desert Uplands Bioregion), vegetation is largely uncleared and comprises extensive tracts of semi-arid savanna woodlands (Woinarski, Andersen and Murphy, 2020). These woodlands are dominated by eucalypt species including silver-leaved ironbark (*Eucalyptus melanophloia*), yellow jacket (*E. similis*) and Brown's box (*E. brownii*) on deep red earths. These eucalypt woodlands are the preferred habitat for southern black-throated finch within the study area, which are intersected by less suitable *Acacia* and *Melaleuca* dominated vegetation communities on tertiary plateaus (Section 3.1, van Osta *et al.*, 2024). The eastern portion of the study area (the Brigalow Belt Bioregion) comprises heavy clay soils that have largely been cleared for agriculture and are avoided by SBTF (Section 3.1, van Osta *et al.*, 2024).



Legend

- Major Road
- Minor Road
- Track
- Study area
- Model Extent
- Mining Lease

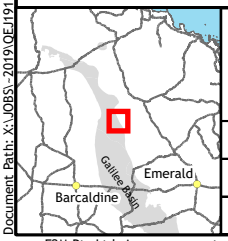
N

Scale 1:240,000 (A4)

0 10

Kilometers

Coordinate System: GDA 1994 MGA Zone 55
Projection: Transverse Mercator



Notes:
Aerial Imagery: © ESRI 2021
Cadastre: © DNRME 2021
Watercourse: © Geoscience Australia 2012
Road: © DNRME 2021

Rev	Description	Drawn	Approved	Date
2	Issued for Use	JY	JMC	30/10/2025
0	Issued for Use	GO	JMC	08/08/2025

FIGURE 3.3-1: STUDY AREA OF THE COLOUR BAND RESIGHT PROGRAM

BTFRP Final Report
Bravus

Map Number	Job Number	Rev
1 of 1	QEJ19152	2

Document Path: X:\JOBS\-2019\QEJ19152\GIS\Final Research Program Report\PRO\QEJ19152\FinalReaseachProgram.aprx\QEJ19152_03_3_1_ColourBand_Resight_Program

E2M Pty Ltd gives no warranty in relation to the data (including accuracy, reliability, completeness or suitability) and accepts no liability for any loss, damage or costs (including consequential damage) relating to any use of the data in this map.

3.3.3.2 Ethics statement

Mist-netting and colour-banding were conducted under the animal ethics permit CA 2020/07/1392, issued by the Queensland Department of Agriculture and Fisheries; research permit WA0025814, issued by the Queensland Department of Environment, Science and Innovation; and Australian Bird and Bat Banding Scheme authority 2832–01, issued by the Australian Government Department of Climate Change, Energy, the Environment and Water.

3.3.3.3 Mist-netting and band program

Nine mist-netting and colour-band resighting surveys were conducted between August 2020 and May 2023, with approximately three per year to align with the dry season (May to November), early wet season (December to February) and wet season (February to April). An additional two water-source watch surveys were undertaken for a SBTF monitoring program (EcoLogical Australia, 2023) in October 2023 and May 2024, which allowed for additional colour-band resighting opportunities. Mist-netting was conducted at 10 locations where SBTF were observed, which included permanent and ephemeral water sources and a single foraging location remote from water. All SBTF captured (n = 551) were fitted with a uniquely identifiable coded Australian Bird and Bat Banding Scheme alloy band. On average 61.2 SBTF were fitted with an alloy band per survey period (range = 22–93) from 5.5 independent locations (range = 4–7) per survey. A subset of 264 SBTF, were also fitted with colour-bands that were uniquely identifiable over the first eight survey periods. On average 33 SBTF were fitted with colour bands per survey period (range = 5–65) from 4.4 independent locations per survey (range = 3–7).

3.3.3.4 Dispersal analysis

We collated all mist-netting and colour-band resighting data for which the identity of the individual SBTF was known. We used only photo-backed or high reliability colour-band resight data to minimise incorrect movement records. When multiple resightings or recaptures of a single bird were recorded on the same day, a single recapture was randomly selected. We calculated the maximum movement distance of each SBTF during the study period was calculated using the maximum Euclidean distance between the initial capture location and all subsequent resightings for that individual. We tested for a difference in the maximum dispersal distance between adult and juvenile SBTF using a Kruskal-Wallis rank sum test using R statistical language version 4.3.0 (R Core Team, 2023).

3.3.4 Results

A total of 177 SBTF were recaptured or resighted during the study period, with an average of 2.1 resightings per bird (range = 1–7). The average recapture or resighting distance from initial capture was 1.05 km, with a maximum dispersal distance of 11.2 km (Figure 3.3-2). A total of 6.2% of all recaptures and resightings (n = 11) were greater than 5 km from the location of first capture, with all of these movements occurring among well connected habitat in the northern portion of the study area (Figure 3.3-3). There were no movements between the SBTF populations north of Moray Carmichael Boundary Road and south of the Carmichael River (Figure 3.3-3).

SBTF recapture and resighting results showed a slight trend towards dispersal through time (Figure 3.3-4), with the trend being largely attributable to movements of SBTF within areas of interconnected habitat in the SBTF population north of Moray Carmichael Boundary Road (Figure 3.3-3). Maximum distances from a bird's initially capture location was 48.8% larger for juvenile SBTF ($\bar{x}$ = 2.37 km, range = 0–11.2 km, n = 11) than adult SBTF ($\bar{x}$ = 1.44 km, range = 0–10.8 km, n = 157); however, the difference was not significant (chi-squared = 1.42, p = 0.23, Figure 3.3-5).

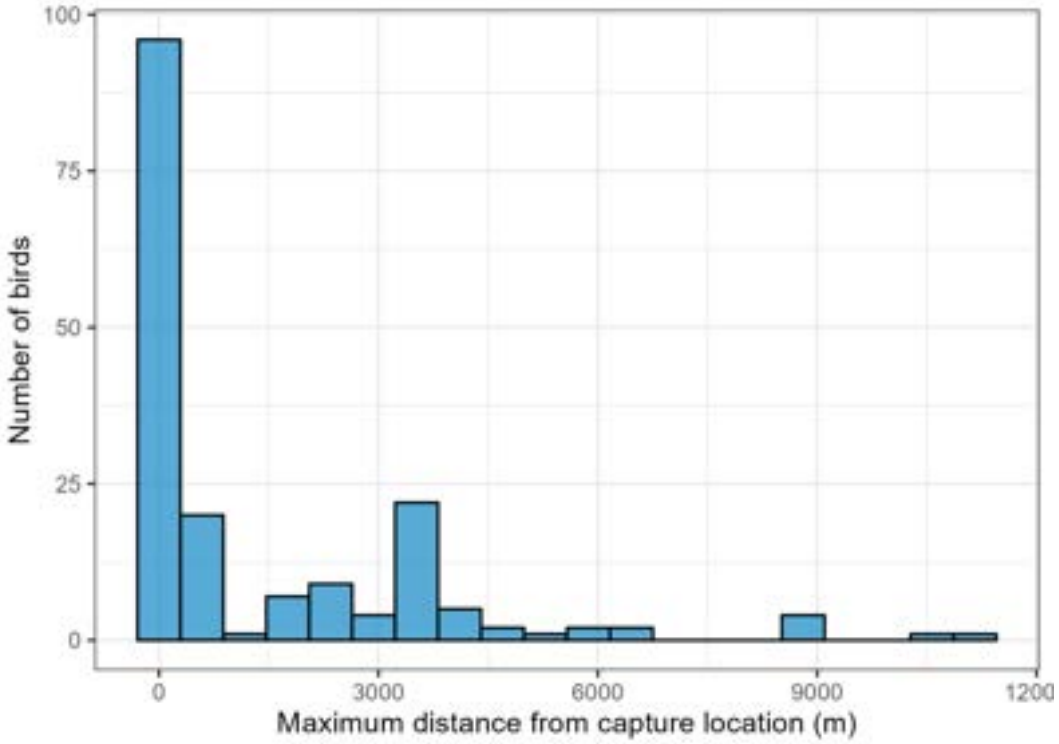
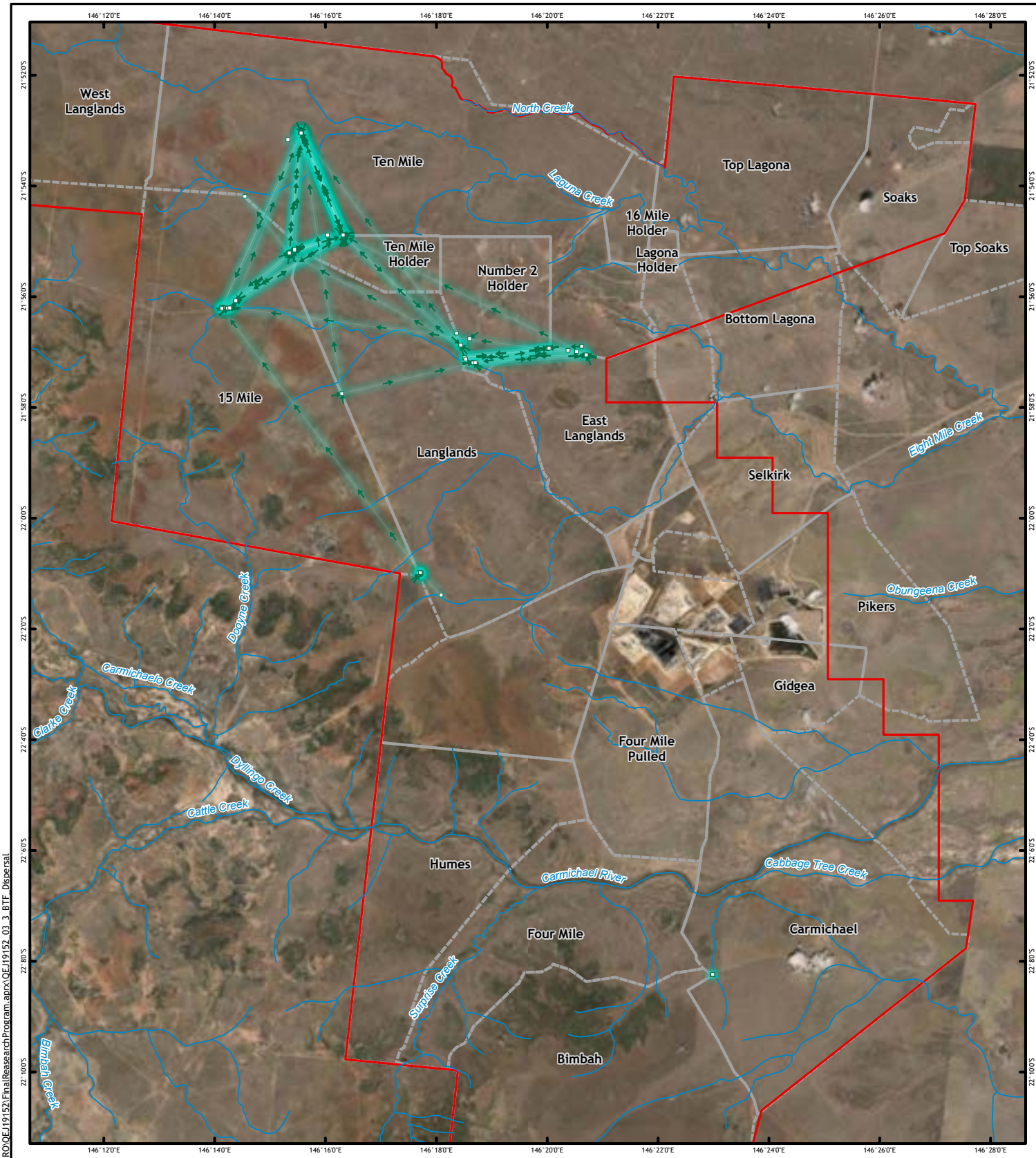


Figure 3.3-2: Distribution of maximum distance from the capture location for southern-black throated finch with one or more resighting or recapture.



Legend

- Dispersal Location
- Dispersal Direction
- Dispersal Density
- Watercourse
- Moray Downs Paddock

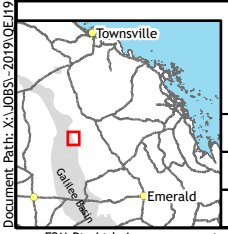
N

Scale 1:175,000 (A4)

0 5

Kilometres

Coordinate System: GCS GDA 1994



Notes:
Aerial Imagery: © ESRI 2024
Watercourse: © Geoscience Australia 2012

Rev	Description	Drawn	Approved	Date
2	Issued for Use	JY	JMC	30/10/2025
0	Issued for Use	GO	JMC	08/08/2025

FIGURE 3.3-3: MIST-NETTING RECAPTURES AND COLOUR BAND RESIGHTINGS OF BLACK THROATED FINCH

BTFRP Final Report
Bravus

Map Number	Job Number	Rev
1 of 1	QEJ19152	2

Document Path: X:\JOBS\-2019\QEJ19152\GIS\Final Research Program Report\PRO\QEJ19152\FinalReasearchProgram.aprx\QEJ19152_03_3_BTF_Dispersal

E2M Pty Ltd gives no warranty in relation to the data (including accuracy, reliability, completeness or suitability) and accepts no liability for any loss, damage or costs (including consequential damage) relating to any use of the data in this map.

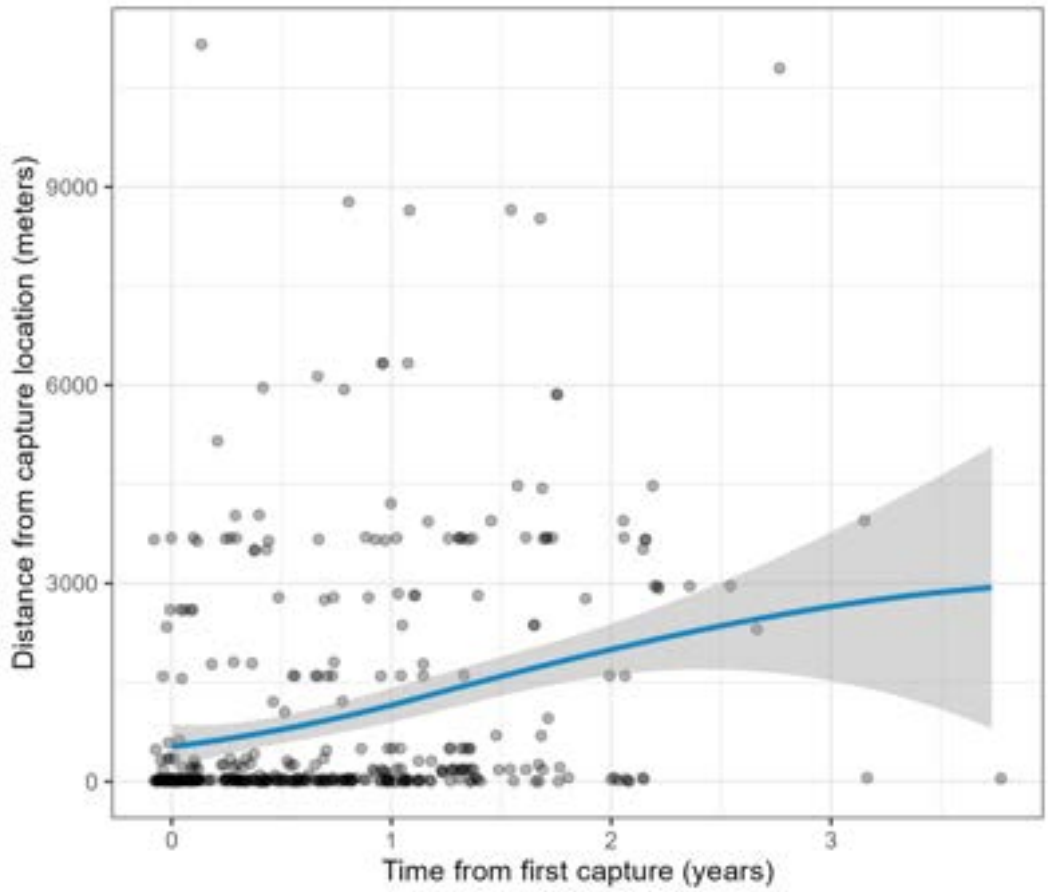


Figure 3.3-4: Relationship between the distance from the initial capture location compared to the time from first capture for mist-netting and colour band resightings of southern-black throated finch. A loess regression (blue line) was fitted, with 95% confidence intervals (grey).

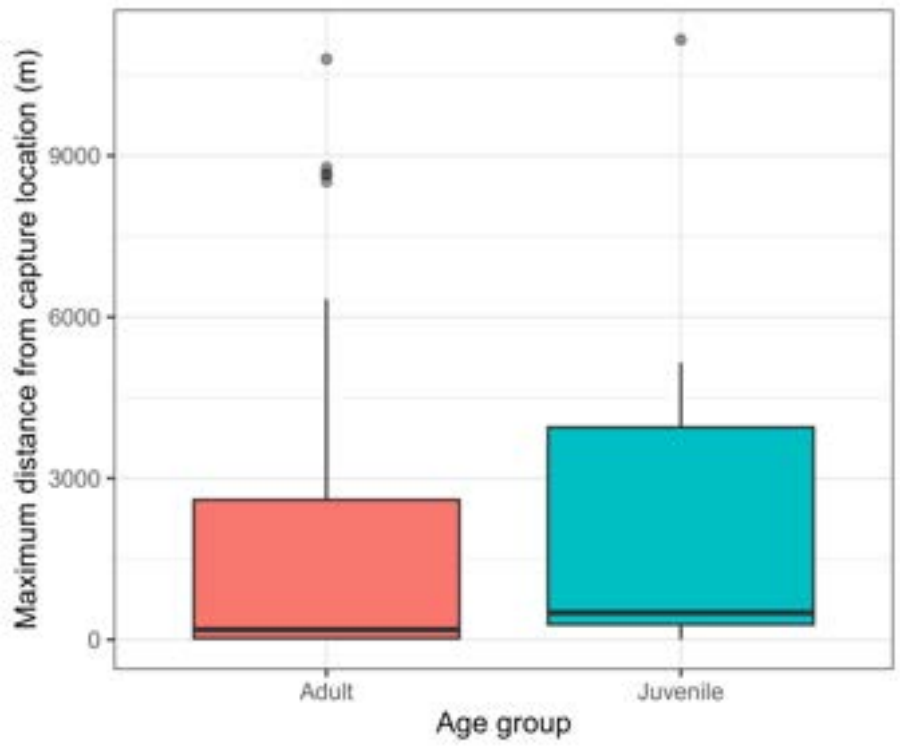


Figure 3.3-5: Distribution of the maximum distance from the initial capture location for adult (n = 157) and immature (n = 11) southern black-throated finch, recorded at the time of their first capture.

The age structure of the SBTF population is difficult to determine as the age of a SBTF cannot be determined through morphological traits or genetic studies. In addition, mortality cannot be differentiated from emigration, which is a problem that has been encountered my multiple banding studies on estrildid finches (Woinarski and Tidemann, 1992; Tidemann, 1996; Franklin, Dostine and Tidemann, 1998).

However, plumage and morphological differences can give an experienced observer an indication of whether the individual is a juvenile or adult. This qualitative data, which is most reliably recorded during mist-netting, can provide a ratio of juvenile to adult individuals captured that can be used as a baseline for ongoing monitoring (Figure 3.3-6). Juvenile to adult ratios in SBTF captured varies seasonally with the highest proportion of juveniles in the wet season (0.207) when breeding occurs (Zann, 1976; Mitchell, 1996b; Higgins and Cowling, 2007). In the dry and early wet seasons, the juvenile to adult ratio decreased to 0.053 and 0.072, respectively. However, when interpreting these values, it is important to simultaneously consider the juvenile to adult ratio and the population estimate (Section 3.5). A high juvenile to adult ratio maybe be a positive indicator for the population if it represents a successful recruitment event. However, if the high juvenile to adult ratio occurs in conjunction with a population decrease, it can represent an increased adult mortality rate.

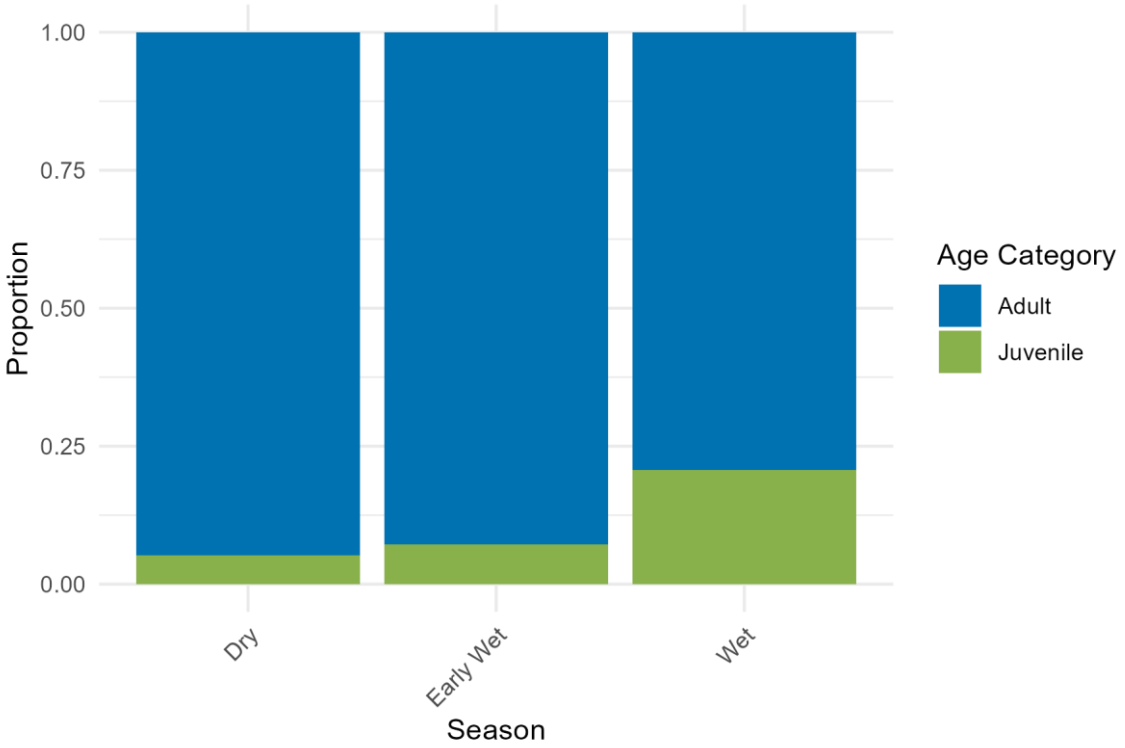


Figure 3.3-6: Ratio of juvenile to adult SBTF captured during mist-netting split by season

3.3.5 Discussion

This study assessed the long-term movements of SBTF captured and resighted over a three-year mist-netting and colour banding program. We found that SBTF had high site philopatry over the three-year study period, with the majority (93.8%, n = 166) of SBTF resighted or recaptured within 5 km of their initial capture location. The 6.2% (n = 11) of SBTF that were resighted or recaptured at distances greater than 5 km provide evidence for dispersal of some SBTF over the three-year period. The rate of long-distance (>5 km) dispersal recorded was substantially higher over this three-year tracking period than compared to the short term movements of SBTF recorded from short-term radio tracking studies, which have been nearly entirely within 5 km of their capture location (Section 3.1; Rechetelo *et al.*, 2016; van Osta *et al.*, 2024). The results indicate that on short-timeframes, SBTF are highly philopatric, with daily movements among foraging, drinking, and nesting areas within their home ranges. On longer timeframes, in the order of years, some SBTF shift their home ranges, potentially in response to changes in food and water availability (Franklin *et al.*, 2017; Collett *et al.*, 2022b) or dispersal from their natal home ranges for mate selection (Vegvari *et al.*, 2018; Li and Kokko, 2019).

Longer dispersal distances in juveniles compared to adults are common among bird species (Paradis *et al.*, 1998; Sutherland *et al.*, 2000; Paruk, 2018), including for estrildid finches (Zann and Runciman, 1994; Van Rooij and Griffith, 2011). While we found that juvenile SBTF had average maximum movement distances that were close to 50% larger than adult SBTF, the difference was not statistically significant, which was likely due to the small sample size of juvenile birds within our study (n = 11). In addition, some natal dispersal may have been missed in our study for birds that were captured between the time of attaining adult plumage (approximately 3 months) and sexual maturity (approximately 9–12 months) (Van Rooij and Griffith, 2011). However, given the juvenile biased dispersal trend observed in our study and observations of juvenile biased dispersal recorded in the closely related long-tailed finch (Van Rooij and Griffith, 2011)

and well-studied zebra finch (Zann and Runciman, 1994), it is probable that natal movement is a factor contributing to dispersal of SBTF.

All long-distance movement within our study occurred within the northern portion of the study area, which comprised an approximately 15,000 ha area of broadly suitable habitat for SBTF with multiple water sources. Despite substantial colour-banding and survey effort, we observed no SBTF movements between the northern and southern portions of our study area, which suggests that barriers to movement may exist between these populations. Vegetation in the east and the middle of the study area has largely been cleared for agriculture, and these non-remnant areas have no known SBTF populations (Author pers. obs.; GHD, 2012; Eco Logical Australia, 2023). The properties to the west contain significant tracts of remnant vegetation, but preferred SBTF habitat on these properties is fragmented, and heavily grazed, potentially reducing habitat suitability. Recent surveys on properties to the west also found small numbers of SBTF compared to the study area, which limits the potential for gene flow through small-scale movements across adjacent habitats (Zann and Runciman, 1994; Schidelko, Stiels and Rödder, 2011). These findings align with the landscape genetics results that suggest there is little geneflow between the SBTF populations within the north and south of our study area (Section 3.2).

The conservation implications of these findings are twofold. First, the SBTF population within the study area should be managed and monitored as two subpopulations, requiring monitoring efforts to detect changes in each population separately. Improvement of habitat for SBTF between these populations could improve gene flow between the northern and southern subpopulations. Second, this study highlights the impact of biogeographic barriers and contemporary land-use changes on the isolation of SBTF populations. Isolated populations have limited opportunities for gene flow, which may lead to differentiation and population sub-structuring (Frankham et al. 2002). Reduced genetic diversity decreases the species' ability to adapt to rapid environmental changes, such as habitat clearing and degradation (Frankham, 2010b). Dispersal of SBTF is promoted by large areas of interconnected habitat and existing SBTF populations, through which geneflow can occur. Clearing or degradation of habitat that results in fragmentation of populations may therefore result in a long-term reduction in gene flow for a species such as SBTF that is likely to have limited dispersal capacity (Schidelko, Stiels and Rödder, 2011; Rollins, Svedin and Griffith, 2012).

3.4 Sex-related variation in the morphometrics, movement ecology and diet of the southern black-throated finch

This section has been submitted to the journal *PLOS One*. Bibliographic details of this scientific paper are:

Williams, B., van Osta, J. M., Griffith S. C., Davis, S. & Dreis, B. Movement ecology and diet are consistent across sexes in the Endangered black-throated finch (*Poephila cincta cincta*). Submitted.

3.4.1 Abstract

Understanding sex-related differences in ecology is essential for effective conservation strategies, especially species that exhibit subtle to no sexual dimorphism. This study examines the Desert Uplands population southern black-throated finch (*Poephila cincta cincta*); an endangered species endemic to north-eastern Australia, focusing on key ecological aspects, including sex ratio, movement ecology, and diet. Although previously considered sexually monomorphic, we found subtle sexual dimorphism in throat patch size (17.1% larger in males) and wing length (7.48% larger in males), with no significant differences in other morphological traits. We found a balanced adult sex ratio (52.3% females, 47.7% males), indicating no significant sex bias; a key indicator for ensuring mating opportunities and breeding success. Despite these differences, only 67.6% of individuals could be sexed accurately based on these traits. We also explored the movement ecology of the species, finding no significant differences between sexes in home range sizes or movement distances, consistent with the species' tendency to live in mixed-sex flocks. Diet analysis, based on DNA metabarcoding of crop and faecal samples, indicated no significant sex-based dietary divergence, with both sexes consuming a similar variety of grass seeds. These findings highlight the importance of understanding the ecological dynamics of the southern black-throated finch for conservation efforts, suggesting that further studies on vocalisations, juvenile dispersal, and advanced morphometric techniques may improve understanding of this species.

3.4.2 Introduction

Conservation relies on understanding the ecology and behaviours of a threatened species to best deliver effective management strategies (Laguna, 2020). However, even within a species, there can be differences in ecological characteristics including physiology or social behaviours observed across sexes (Shine, 1989; Janicke and Fromonteil, 2021). Most previous studies on sexual dimorphism in birds have focused on dimorphic species, whereby striking differences in morphological features, including size, ornamentation and colouration, between males and females are evident (Hedrick and Temeles, 1989; Abraham, 1998). For example, Qvarnström et al. (2003) and (2006) demonstrated that breeding success and male ornamentation are positively correlated in the highly ornamented and sexually dimorphic collared flycatcher (F*icedula albicollis*). However, recognising expression of sexual variation can be difficult for roughly half of the world's avian species are sexually monomorphic, with similar appearing males and females appear (Griffiths *et al.*, 1998). The inability to recognise morphological differences between sexes also makes determining differences in a species ecology such as dispersal movements, home ranges, parental care and diet even more challenging to assess, in such cases, sex differentiation must rely on DNA methods (Griffiths, 2000; Cerit and Avanus, 2007).

In many species perceived as sexually monomorphic, detailed investigations of morphological traits have revealed subtle sexual dimorphism (Eaton, 2005). Despite the subtlety of these differences, they can be significant for the species' ecology (Dias, 2019; Owens & Hartley, 1998). The long-tailed finch (*Poephila acuticauda)*, is one such species that was generally considered sexually monomorphic, and indeed, used to provide empirical evidence for the sexual indistinguishability hypothesis; an argument that sexual monomorphism reduces sex-specific competition between dominant and subdominant individuals in a group-living monogamous bird (Langmore and Bennett, 1999; Dechaume-Moncharmont, Monceau and

Cezilly, 2011). However, subsequent research by van Rooji and Griffith (2010) found that long-tailed finches had significant differences among sexes in seven out of ten morphological traits that they assessed, and that 92% of individuals could be reliably sexed by integrating data on these traits (Van Rooij and Griffith, 2010).

Sexual dimorphism can be driven by divergent selection on males and females that is often linked to different sex roles in reproduction or the acquisition of resources (Owens & Bennett, 1994). As such, these pressures can have important sex-specific impacts on fitness and mortality (Promislow, Robert and Thomas, 1992; Székely *et al.*, 2014; Janicke and Fromonteil, 2021). For example, in families such as raptors and procellariforms (albatrosses, petrels and shearwaters), differences in body size impose higher energetic demands on the larger sex, making them more vulnerable to risks of injury or mortality (Bugoni, Griffiths and Furness, 2011; Yang *et al.*, 2022). Sex-specific behaviours can also drive mortality bias between sexes, for example, Grüebler et al. (2008) found that female meadow-nesting birds are more likely to killed by mowing whilst incubating or brooding. These skews in sex ratio in local populations can have adverse effects on population health and productivity (Székely *et al.*, 2014; Vegvari *et al.*, 2018). Declining species are prone to skewed sex ratios, which can further accelerate population declines and local extinctions (Donald, 2007; Morrison *et al.*, 2016; Heinsohn *et al.*, 2019). As a result, whilst adult sex ratios typically vary within populations, and are often skewed in birds, unbalanced sex ratios in small, fragmented populations can be a worrying sign of unbalanced sex mortality and risk of population collapse (Donald, 2007; Serrano-Davies *et al.*, 2022).

Birds may also have sex-related differences in their exploitation of dietary resources, due to differing nutritional or energetic requirements, such as caregiving to offspring or competition for mates, or to reduce resource competition between the sexes or sympatric species (Shine, 1989; Catry, Phillips and Croxall, 2006). Such is the case of the great bustard (*Otis tarda*), in which adults have sex-specific dietary differences that are linked to the distinct reproductive roles and body size difference between the males and females (Bravo *et al.*, 2016). Likewise, sexual differences in species movement ecology may also result in differences in home range size, seasonal patterns of movement, and habitat use (Nathan *et al.*, 2008; Allen and Singh, 2016). For example, in the king rail (*Rallus elegans*) males were found to have smaller home ranges than females, however their home ranges contained a higher percentage of higher quality habitat synonymous with their breeding territories (Kolts and McRae, 2017). As foraging methods, diets, movement patterns, and home ranges, can all impose different costs and benefits, their influence on an individual's survival is critical to understand (Durell *et al.*, 2008; Papageorgiou *et al.*, 2021). It is also important to understand the dynamics that these differences may have over temporal and spatial scales, such as breeding and non-breeding seasons, or across periods of resource limitations, which may further impact an individuals ability to survive and reproduce (Liker and Székely, 2005; Durell, 2000; Molokwu, Olsson and Ottosson, 2007; Promislow, Robert and Thomas, 1992). Therefore, knowledge about these potential differences in the requirements of the two sexes is important to inform management strategies for the species.

Finches within the family Estrildidae, also known as grass finches, are widespread across Australia, spanning from the tropics to arid zones (Higgins, Peter and Cowling, 2006). There is substantial variation of sexual morphology, diet and movements observed across species of these estrildid finches. Species within this group range from the zebra finch which is distinctly sexually dimorphic to species that are more subtly sexually dimorphic such as the plum-headed finch (*Emblema modesta*) to those that are near sexually monomorphic, such as the long-tailed finch, black-throated finch (*Poephila cincta*) and double-barred finch (*Taeniopygia bichenovii*) (Zann, 1972; Shephard, Pridham and Forshaw, 2012). Estrildid finches also have a high variation in movement ecology and diet among the species, ranging from low-mobility dietary generalists, such as crimson finch (*Neochmia phaeton*) to high mobility dietary specialists (e.g. Gouldian finch) that are nomadic across regional scales (Franklin *et al.*, 2017; Sydney J Collett *et al.*, 2022b). However, there has been little research on sex-related differences in the ecology of these

species. This is particularly important for estrildid finches that are threatened, which in Australia include Gouldian finch, eastern star finch (*Neochmia ruficauda ruficauda*), southern black-throated finch (*Poephila cincta cincta*), diamond firetail (*Stagonopleura guttata*) and western beautiful firetail (*Stagonopleura bella samueli*) (Department of Climate Change, Energy, the Environment and Water, 2024). Understanding the sex-related differences in morphology, diet and movement patters of these species will help to inform conservation actions (Catry, Phillips and Croxall, 2006).

The southern black-throated finch is an estrildid finch, endemic to north-eastern Australia. This sub-species has suffered an estimated 80% contraction in its historical range and its contemporary population is centred on two population 'strongholds'; Townsville and Desert Uplands (Laguna *et al.*, 2019). As such the southern black-throated finch is listed as Endangered both under the Queensland *Nature Conservation Act 1992*, and nationally under the Commonwealth *Environment Protection and Biodiversity Conservation Act 1999* (Commonwealth of Australia, 2023). The southern black-throated finch is a clade species of the previously mentioned long-tailed finch that shares many morphological and ecological characteristics, but its sex-related biology remains poorly studied. The long-tailed finch and southern black-throated finch diverged around 1.75 million years ago (Lopez *et al.*, 2021) and they are ecologically similar and allopatric, both inhabiting grassy open savanna woodlands and with similar home ranges, drinking patterns and mobility (Sheard *et al.*, 2020; Sydney J Collett *et al.*, 2022a, 2022b; van Osta *et al.*, 2024b). Like the long-tailed finch, southern black-throated finch are believed to show subtle sexual dimorphism, while retaining many similarities in the ornamentation of both sexes (Zann, 1972). However, morphological and ecological differences between sexes of southern black-throated finch have yet to be quantified, and to date the sex ratio of adult populations, movement patterns and diet have not been studied.

In this study we investigated the sexual differences in morphology, movements, diet and adult sex ratio of the southern black-throated finch. Broadly, our work aimed to inform ecological knowledge of this endangered species that supports our understanding of sex-related ecological risks for the species and guides conservation actions (Laguna *et al.*, 2019).

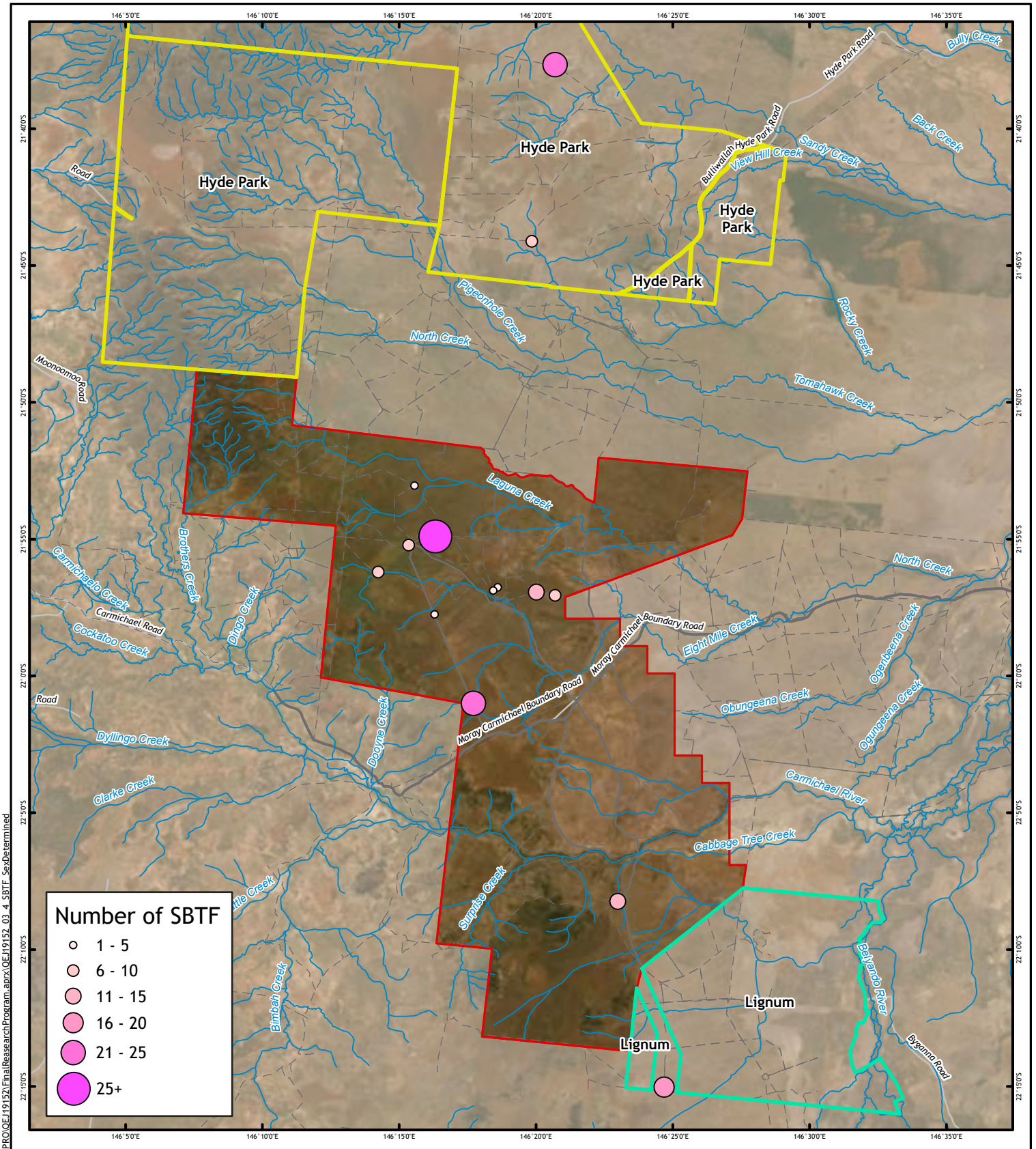
3.4.3 Methods

3.4.3.1 Study area and sampling

This study was conducted on the Moray Downs property in the Desert Uplands Bioregion of Queensland, Australia (Figure 3.4-1). The property is managed by Bravus Mining and Resources and includes areas designated as environmental offset for the southern black-throated finch. The study area is within the tropical savanna biome of northern Australia, which is broadly characterised by long annual dry seasons with summer dominated rainfall (Woinarski, Andersen and Murphy, 2020). The majority of vegetation communities within the study area are eucalypt woodlands with a grassy ground layer, including those dominated by *Eucalyptus melanophloia, E. similis* and *E. brownii* that occur on deep red earths (Department of Resources, 2025). Rainfall within the study area averages 541 mm per year, with the majority of this (approximately 65%) falling within the wet season, which is typically from December to March (Bureau of Meteorology, 2023, station number 036050, 25 km from the study area).

Mist-netting was undertaken for multiple purposes, which included population monitoring, morphometric measurements, colour banding for long-term movement research, tagging for radio tracking, blood samples to assess the genetic traits of the southern black-throated finch, including sex as well as crop and faecal samples to assess diet. As a result of these multiple purposes and animal welfare and ethics considerations, each treatment could not be applied to each bird. Given the difficulty of determining gender using morphological measures, blood samples were taken from a subset of birds for molecular sexing. Blood samples were taken using a 23-gauge sterilised needle to puncture the brachial vein. Blood was collected on a Whatman classic Flinders Technology Associates collection card paper and stored with 70% denatured ethanol. Blood samples were collected from a total of 155 southern black-throated finch

across five survey periods approximately evenly distributed between September 2021 and May 2023. On average 31 ± 19.4 birds were blood sampled during each survey period ($\bar{x}$ ± 1 SD, range = 9–62). Samples were taken across 13 locations, with the number of samples per location ranging from 1–32 southern black-throated finch.



Legend

- Watercourse
- Major Road
- Minor Road
- Track
- Project Area
- Hyde Park Station
- Lignum Station

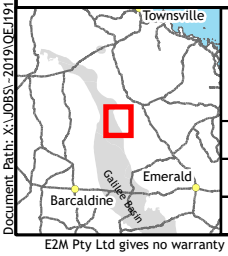
N

Scale 1:350,000 (A4)

0 5 10 15

Kilometers

Coordinate System: GDA2020



Notes:
Aerial Imagery: © ESRI 2021
Cadastre: © DNRME 2024
Watercourse: © Geoscience Australia 2012
Road: © DNRME 2024

Rev	Description	Drawn	Approved	Date
2	Issued for Use	JY	JMC	30/10/2025
0	Issued for Use	GO	JMC	08/08/2025

FIGURE 3.4-1: LOCATIONS OF SBTF FOR WHICH SEX WAS DETERMINED

BTFRP Final Report
Bravus

Map Number	Job Number	Rev
1 of 1	QEJ19152	2

Document Path: X:\JOBS\-2019\QEJ19152\GIS\Final Research Program Report\PRO\QEJ19152\FinalReasearchProgram.aprx\QEJ19152_03_4_SBTF_SexDetermined

E2M Pty Ltd gives no warranty in relation to the data (including accuracy, reliability, completeness or suitability) and accepts no liability for any loss, damage or costs (including consequential damage) relating to any use of the data in this map.

3.4.3.2 Sex determination

The 155 blood samples taken from the southern black-throated finches were sent to Diversity Arrays Technology (http://www.diversityarrays.com/) for DArTseq genotyping-by-sequencing (GBS). After quality control, 14,808 SNPs were retained for 155 samples (Davis et al., *in prep*) and the sex of each individual was inferred from genetic data using a method adapted from *seGMM* (Liu et al., 2022) and *seXY* (Qian et al., 2017). In brief, the sex of each sample was inferred by first mapping unfiltered loci to the Z and W chromosomes of the zebra finch (*Taeniopygia guttata*), RefSeq assembly GCF_003957565.2. Mapped loci were filtered to remove those with percent identity < 80%, E-value > $1e^{-10}$, reproducibility < 0.98, loci located in psuedoautosomal regions or on both chromosomes, and physically linked loci (retaining only one SNP per RAD tag). Three sex-associated metrics (Z heterozygosity, Z/W ratio and W missing rate) for each individual were used to infer sex using a model-based clustering approach (Figure 3.4-2).

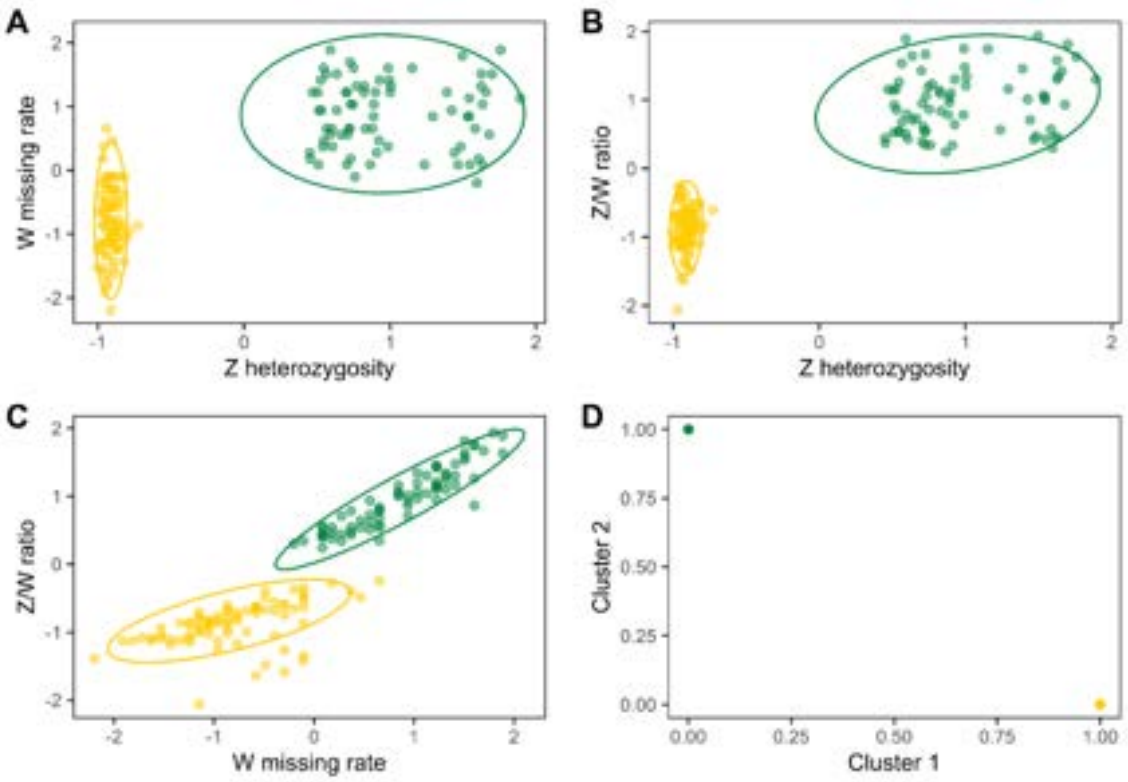


Figure 3.4-2: Sex inference results for black-throated finches. A-C) correlations between pairs of sex-associated features used in the analysis, coloured by sex inferred with mclust clustering method (green = males, yellow = females). D) Assignment probability of samples to each cluster (cluster 1 = female, cluster 2 = male). Males are ZZ in birds, and thus are expected to have higher Z heterozygosity, a higher W missing rate and a higher Z/W ratio.

3.4.3.3 Morphology measurements

Morphology measurements were taken from 111 southern black-throated finch for which sex was molecularly determined (as above). A standard suite of morphology characteristics was measured following specifications by Lowe (1989) and Dr Jon Coleman (2020), which included tarsus length, wing length, bill length, tail length and weight. In addition, we recorded the width and length of the black throat-patch of each bird, which has been identified as a key sexually related character of the species (Zann, 1972; Higgins, Peter and Cowling, 2006), and was found to be one of the more sexually dimorphic

traits in the closely related long-tailed finch (Van Rooij and Griffith, 2010) as depicted in Figure 3.4-3. The size of this throat patch was then estimated by multiplying its length by its width.

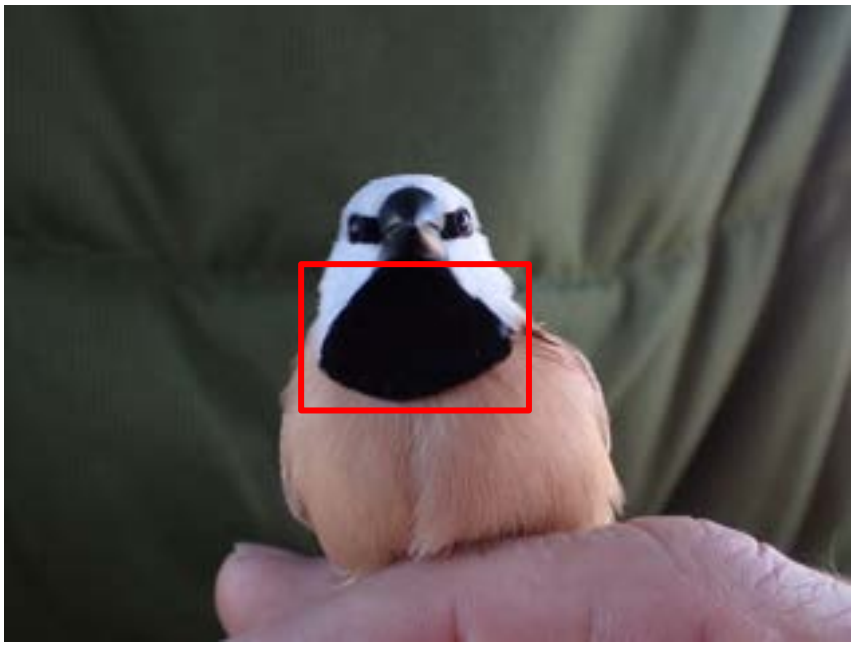


Figure 3.4-3: Photo of SBTF throat patch

We tested for sex-related differences among the morphometric measurements using a multiple analysis of variance (MANOVA) in R statistical language version 4.3.2 (R Core Team, 2023). We also tested the predictive power of the morphology variables to classify sex using a linear discriminant analysis with the MASS package version 7.6-61 (Venables and Ripley, 2013). We computed standardised coefficients of the discriminant analysis using the candisc package version 0.9.0 (Friendly and Fox, 2024).

3.4.3.4 Movement ecology analyses

All southern black-throated finch within this study (n = 155) were fitted with a uniquely coded Australian Bird and Bat Banding Scheme supplied alloy band (size 02), which were used to record mist-netting recaptures over the approximate 20-month study period. Colour bands were also fitted to 52 of the southern black-throated finch, which provided resighting information during water source watches that were undertaken as part of a long term southern black-throated finch monitoring program within the study area. 434 water source watches were undertaken three times per year (early wet, wet and dry seasons), with an average of 29 water source watches per survey event. Incidental colour band sightings were also recorded where there was a supporting photograph to identify the colour band combination. Radio transmitters were attached to 54 southern black-throated finch for which sex was later determined (25 female and 29 male).

Methods to assess the home range sizes and habitat use of these SBTF are provided in detail by van Osta et al. (2024b). In brief, the methods included a combination of manual and automated radio tracking to record individuals for an average of 21.8 ± 22.5 days (range = 0.827–120 days). Home range sizes were calculated using an autocorrelated kernel density estimate (AKDE) home ranges using the ctmm package version 1.2.0 (Calabrese, Fleming and Gurarie, 2016; Silva *et al.*, 2022). We tested for an effect of sex and season on the 95% autocorrelated kernel density estimate home range of SBTF using a generalised linear model with a negative binomial distribution in the glmmTMB package version 1.1.7 (Magnusson *et al.*, 2023). An interaction effect between sex and season was included in the model.

We calculated the maximum distance that each banded southern black-throated finch was recaptured or resighted from its initial capture location. Only mist-netting recaptures and photo-backed resightings were used for this analysis to ensure accuracy of movement records. We tested for sex-related variation in this maximum movement distance using a Kruskal-Wallis test. We also calculated the number of individuals that moved more than 3 km from their initial point of capture and tested for sex-related differences using a chi-squared test.

3.4.3.5 DNA metabarcoding of crop and faecal samples

Crop (n = 6) and faecal (n = 53) samples were analysed from a subset of southern black-throated finch captured between October 2022 and October 2023. These samples were spread approximately evenly among four surveys and included 30 dry season samples and 29 wet season samples.

Analysis of the biological content in southern black-throated finch faecal and crop samples was undertaken by the Griffith University DNA Sequencing Facility through metabarcoding of the ITS2 gene region. Subsequent identification of taxa from the DNA sequences was undertaken using the QIIME2 pipeline by Blue Mountain Analytics. Publicly available genomic databases were supplemented through Sanger sequencing of locally sourced plant material. DNA was extracted from this plant material using a DNeasy Pro Plant kit (Qiagen). Confirmation of DNA quality and quantity was conducted by Nanodrop quantification.

We tested for an effect of sex on the diet of the southern black-throated finch using a generalised linear model with an ordered beta distribution (Douma and Weedon, 2019) in the glmmTMB package version 1.1.7 (Magnusson *et al.*, 2023). Sex-related differences were tested using an interaction effect between the taxonomic grouping identified by the DNA metabarcoding (generally to a genus level) and sex. All model fits and the distribution of residuals were tested using the DHARMa package version 0.4.6 (Hartig, 2022).

3.4.4 Results

3.4.4.1 Adult sex-ratio

Overall, there was no indication of a sex bias among adult southern black-throated finch, with 52.3% (n = 81) being female and 47.7% (n = 74) being male (binomial test, p = 0.63). Furthermore, there was no significant evidence of sex bias among the three areas of southern black-throated finch sampled (chi-squared 3.74, p = 0.154).

3.4.4.2 Morphometrics

Females had significantly smaller sizes of their throat patch (17.1%, p < 0.001) and wing lengths (7.48%, p = 0.011) than male southern black-throated finch (Table 3.4-1). No significant differences were identified in the other morphological characteristics assessed (tail length, tarsus length, bill length and weight), with the average values of these characters all being less than 3% difference between the sexes.

Linear discriminant analysis was 67.6% accurate in predicting the sex of southern black-throated finch given the morphological measurements recorded (Table 3.4-1). Patch size followed by wing length and tarsus length contributed the largest discriminant power, with standardised coefficients of 0.849, 0.384 and 0.344, respectively (Table 3.4-1).

Table 3.4-1: Assessment of morphological variation between male (n = 58) and female (n = 53) southern black-throated finch

Measurement	Mean ± SE		Difference[1]			Standardised coefficient of discriminant analysis[2]
	Female	Male	%	*F*	p	
Patch size (mm^2)	240 ± 6.48	285 ± 7.95	17.1	18.8	< .001	0.849
Wing length (mm)	60.4 ± 0.169	61.1 ± 0.220	1.20	6.69	0.011	0.384
Tail length (mm)	37.1 ± 0.338	37.9 ± 0.334	2.04	2.58	0.111	-0.028
Tarsus length (mm)	15.4 ± 0.115	15.6 ± 0.093	1.56	2.72	0.102	0.344
Bill length (mm)	10.5 ± 0.102	10.5 ± 0.095	0.627	0.223	0.637	-0.066
Weight (g)	14.3 ± 0.167	14.6 ± 0.135	2.08	2.01	0.160	-0.111

[1] Differences between sexes were assessed using a MANOVA (R Core Team, 2023).
[2] Contribution of each morphological characteristic to a linear discriminant analysis (Venables and Ripley, 2013; Friendly and Fox, 2024).

3.4.4.3 Movement ecology

The mean 95% AKDE home range estimate was 229 ha (SE = 49.3 ha, range = 8.58–1170 ha) for female southern black-throated finch and 281 ha (SE = 37.1 ha, range = 28.4–862 ha) for male southern black-throated finch (Figure 3.4-4). While there was no statistically significant difference between the sexes, male home ranges showed a trend of being 22.9% larger than female home ranges (GLM: ß = 0.373, 95% CI: -0.0651–0.8112, Table 3.4-2). There was no statistically significant difference between the average home ranges between the dry season (258 ha, SE = 29.8 ha, range = 20.8–632 ha) and the wet season (255 ha, SE = 65.1 ha, range = 8.58–1170 ha) (GLM: ß = -0.232, 95% CI: -0.908–0.444, Table 3.4-2), nor interaction effects between sex and season (GLM: ß = -0.00970, 95% CI: -0.829–0.809, Table 3.4-2). However, large variation in home range sizes limited our ability to discern significant differences related to the effect of sex and season on home range size.

Table 3.4-2: Estimated regression parameters, standard errors, z-values and p values for a generalised linear model relating home range size to sex and season.

Predictor	Estimate	Std. error	z-value	p
(Intercept)	5.42	0.170	31.9	< .0001
Sex (Male)	0.373	0.224	1.67	0.0951
Season (Wet)	-0.232	0.345	-0.67	0.502
Sex (Male) x Season (Wet)	-0.00970	0.418	-0.02	0.982
Observations	54			
R^2 marginal	0.828			

The percent of southern black-throated finch that had a maximum distance from their initial point of capture of more than 3 km was similar between female (13.0%, n = 3/23) and male (13.8%, n = 4/29) southern black-throated finch (chi-squared = 0.00208, df = 1, p-value = 0.964). The average maximum

distance that southern black-throated finch were recorded from their initial capture location was also similar between females (1.08 km, SE = 0.303 km, range = 0–3.9 km) and males (1.16 km, SE = 0.283 km, range = 0.003–4.50 km) (Kruskal-Wallis chi-squared = 0.557, df = 1, p-value = 0.455, Figure 3.4-5).

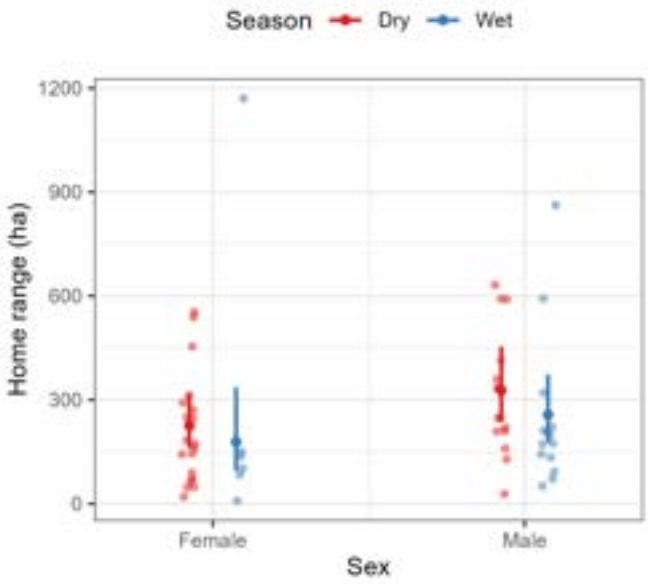


Figure 3.4-4: Home ranges of female (n = 25) and male (n = 29) southern black-throated finch separated in the dry (red, n = 34) and wet (blue, n = 20) seasons). Lines show a 95% confidence interval of the mean. Home ranges were calculated using a 95% autocorrelation corrected kernel density estimate (van Osta *et al.*, 2024). A random horizontal jitter has been added to aid visualisation of the distribution of data.

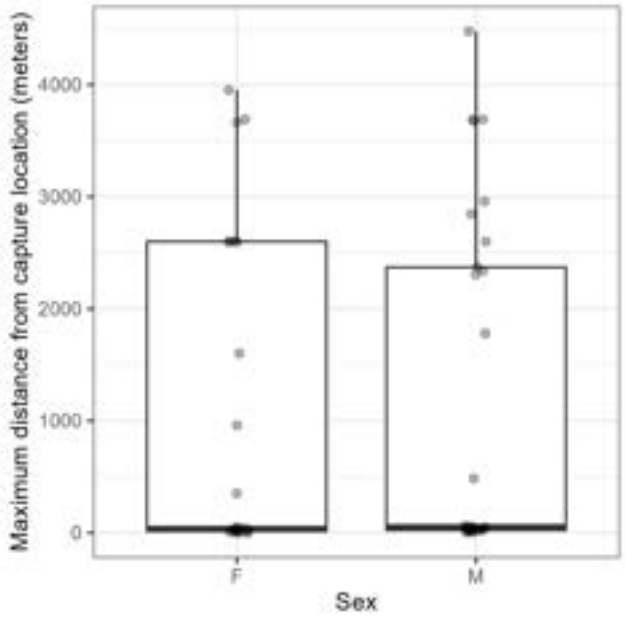


Figure 3.4-5: Maximum distance female (n = 23) and male (n = 29) southern black-throated finch were resighted from their initial capture location. A random horizontal jitter has been added to aid visualisation of the distribution of data.

3.4.4.4 Diet

The contribution of all forage taxa examined was similar between sexes (p values > 0.05, Figure 3.4-6, Table 3.4-3). In addition, there was no significant interaction between season and sex on the diet of southern black-throated finch (p values > 0.05, Figure 3.4-6, Table 3.4-3).

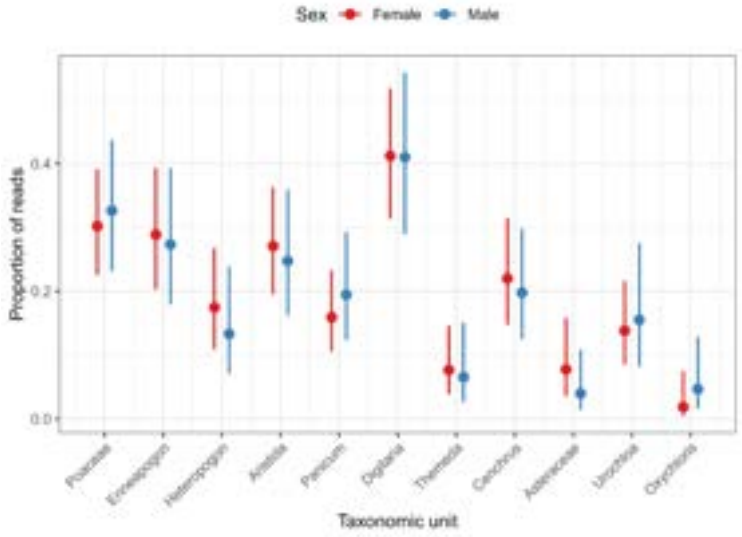


Figure 3.4-6: Interaction effects of taxon and sex on the proportion of DNA reads for crop and faecal samples of female (n = 34) and male (n = 25) southern black-throated finch. Means and interactions effects have been derived using a generalised linear model with an ordered beta distribution. Lines show a 95% confidence interval of the mean. Taxa that contributed less than 1% of DNA reads were excluded from the analysis.

Table 3.4-3: Relationships between proportion of taxa in diet, season and sex for a generalised linear model with an ordered beta distribution.

Predictor	Estimate (β)	95% CI	p
(Intercept)	-0.9899	-1.42 – -0.5598	< .001
Species (Asteraceae)	-1.489	-2.363 – -0.6141	0.0008493
Taxa (*Cenchrus*)	-0.2757	-0.8715 – 0.3202	0.3645
Taxa (*Digitaria*)	0.6357	0.07498 – 1.196	0.02628
Taxa (*Enneapogon*)	0.08888	-0.4948 – 0.6726	0.7654
Taxa (*Heteropogon*)	-0.5627	-1.206 – 0.08045	0.08638
Taxa (*Oxychloris*)	-2.972	-4.46 – -1.485	< .001
Taxa (*Panicum*)	-0.6746	-1.253 – -0.09581	0.02235
Taxa (Poaceae)	0.1526	-0.3734 – 0.6787	0.5696
Taxa (*Themeda*)	-1.501	-2.294 – -0.7076	< .001
Taxa (*Urochloa*)	-0.8396	-1.487 – -0.1919	0.01107
Season (Wet)	-0.1419	-0.433 – 0.1492	0.3393
Sex (Male)	-0.1221	-0.7772 – 0.5331	0.715

Predictor	Estimate (β)	95% CI	p
Taxa (Asteraceae) x Sex (Male)	-0.5838	-2.018 – 0.8508	0.4251
Taxa (*Cenchrus*) x Sex (Male)	-0.01227	-0.9221 – 0.8975	0.9789
Taxa (*Digitaria*) x Sex (Male)	0.1133	-0.7639 – 0.9904	0.8002
Taxa (*Enneapogon*) x Sex (Male)	0.04561	-0.8516 – 0.9428	0.9206
Taxa (*Heteropogon*) x Sex (Male)	-0.1992	-1.241 – 0.8426	0.7078
Taxa (*Oxychloris*) x Sex (Male)	1.068	-0.8094 – 2.946	0.2648
Taxa (*Panicum*) x Sex (Male)	0.3653	-0.5217 – 1.252	0.4196
Taxa (Poaceae) x Sex (Male)	0.2348	-0.5866 – 1.056	0.5752
Taxa (*Themeda*) x Sex (Male)	-0.04732	-1.321 – 1.227	0.942
Taxa (*Urochloa*) x Sex (Male)	0.2582	-0.7996 – 1.316	0.6324
Season (Wet) x Sex (Male)	0.1435	-0.3067 – 0.5937	0.5322
Observations	649		
AIC	185.6		

3.4.5 Discussion

3.4.5.1 Adult sex ratio

The population of southern black-throated finch sampled within this study had a balanced adult sex ratio. This population in the Desert Uplands Bioregion of Queensland is one of the two remaining stronghold populations for this Endangered species (Laguna *et al.*, 2019). Biased adult sex ratios can be a marker of populations under pressure or declining (Allentoft *et al.*, 2010; Serrano-Davies *et al.*, 2022). Any skews in sex ratio in local populations can have adverse effects on population health and productivity, particularly in socially monogamous birds where males and females maintain a stable pair bond over their reproductive period and the contribution of both sexes is typically required for the successful raising of offspring (Székely *et al.*, 2014; Vegvari *et al.*, 2018).

The equity in the number of males and females within the population will reflect mate choice and potential breeding success within the species.

In this population of the southern black-throated finch, the absence of a sex-biased skew indicates that both females and males should have sufficient access to mates. Nonetheless, this is the first study that provides the sex ratio of southern black-throated finch and provides a baseline for ongoing monitoring of the species and can be compared to other populations of the species.

3.4.5.2 Sexual dimorphism

Our results suggest that while male and female southern black-throated finch appear similar, they are not sexually monomorphic. We identified that southern black-throated finches are sexually dimorphic in two of the six assessed traits, notably that across individual traits, males possessed larger features than females with respect to throat patch area (17.1%) and wing length (7.48%). We found that tail length, tarsus length, bill length and weight did not differ between males and females of this species. Whilst we identified that there is distinguishability in individual traits between the sexes in the southern black-

throated finch, our analysis using six morphological traits, demonstrated that only 67.6% of individuals could be reliably sexed based on appearance alone.

Previously, studies on avian morphology in respect to visual signalling often focuses on conspicuous traits such as plumage or ornamentation (Gluckman, 2014; Owens & Hartley, 1998), however, other species have been found to possess more subtle attributes that are important in expressing qualities between males and females (Rowe and Skelhorn, 2004; Stevens and Cuthill, 2007; Lind *et al.*, 2014). While we found that sexual dimorphism in the southern black-throated finch was subtle, our study measured only six morphological traits. Sexual dimorphism may also be present in other attributes that were not measured in this study. Due to the sexual indistinguishability hypothesis first published by Burley (1981) we now know the importance of identifying the deficiencies of human vision relative to avian vision (Bennett and Cuthill, 1994). It is now known that birds possess tetrachromatic vision which allows them to distinguish reflectance in the UV spectrum, an ability human's trichromatic vision does not afford (Burley, 1981). Optical spectrometry has revealed often subtle but significant sexual dichromatism expressions in several species that were previously believed to be monomorphic, for example in the sympatric long tailed finch van Rooji and Griffith (2010) found that colour spectrometric measurements of the upper mandible and crown, had significant differences between sexes. Santos and Lumeji (2007) also studied the plumage and individual feathers on long-tailed finch and found that often different angle geometries of plumage were overlooked, as single unidirectional reflectance spectrometry was often the only method employed. As such, additional sexual discrimination of morphology using multi-optical spectrometry may help to increase the confidence of sexing the species using non-DNA methods, leading to a greater understanding of sexual dimorphism in the species, as well as potentially reveal markers for breeding success and mate selection dynamics within the species (Santos and Lumeij, 2007; Van Rooij and Griffith, 2010).Nonetheless, although it is possible that subtle sexually dimorphic colormetric variation may exist in the southern black-throated finch, in the closely related long-tailed finch, the most dimorphic feature was found to be the throat patch which differed by 15% (van Rooij & Griffith 2010) - very similar to our finding of 17%.

Southern black-throated finch may also discriminate the sex of conspecifics using behavioural cues over short and long-term interactions (Tomaszycki *et al.*, 2009). Differences in acoustic cues or vocalisations are evident in the sister species of the southern black-throated finch, namely zebra finches (*Taeniopygia gutatta*), who possess a song system which is highly sexually dimorphic (Tomaszycki *et al.*, 2009). Given the presence of vocalisation differences in such a closely related species, this type of dimorphism may be present in the southern black throated finch, a species known for its complex vocal repertoire of 12 known calls (Shephard, Pridham and Forshaw, 2012). The ability to sex a species by vocalisations presents a non-invasive alternative to classical sexing techniques such as morphometric and genetic analyses as these involve the capture and subjection of birds to procedures creating ethical constraints (Volodin *et al.*, 2015). In addition, van Osta et al. (2023), has already developed a dataset for an automated recogniser for the southern black throated finch. If future studies determine the presence of acoustic sexual dimorphism between male and female southern black-throated finch, the employment of low-cost acoustic sensors to feed into the model developed by van Osta et al. (2023) would enable an alternative detection and monitoring of threatened species and assist in estimating sex ratios across spatial and temporal scales (Hervás, Rosa M. Alsina-Pagès, *et al.*, 2017; Sugai, Silva, Ribeiro, *et al.*, 2019; S. Ross *et al.*, 2023).

Several hypotheses have been proposed to explain sexual dimorphism and monomorphism, including mutual mate choice, genetic correlation, sexual selection and sexual indistinguishability (Burley, 1986; Hedrick and Temeles, 1989; Abraham, 1998). Whilst Burley (1986) presented that often sexual dimorphism is influenced by either an individual's own desirability to attract a mate (differential-access hypothesis), or an individual's amount of parental investment influences their mate attractiveness (differential-allocation hypothesis), Burley (1986) also presented the sexual indistinguishability hypothesis which

suggests that sexual monomorphism is an adaption evolved to avoid competition in monogamous group-living species. As such, the sexual indistinguishability theory dictates that sexual differentiation is reduced in cooperative breeding birds (Burley, 1986). Southern black-throated finch are a socially monogamous and flock-living species in which parental care is shared between sexes (Zann, 1972; Shephard, Pridham and Forshaw, 2012). Given that we could distinguish the sex of only 67.6% of the southern black-throated finch using the traits that we measured, the sexes can be considered morphologically similar but not indistinguishable.

3.4.5.3 Movement ecology

Our study did not find significant evidence of sex-biased home ranges between the sexes of the southern black-throated finch. These findings are consistent with the movement ecology of southern black-throated finch, which van Osta (2024b) identified have largely sedentary home ranges characterised by local-scale movements over large home ranges and limited broader landscape movements. These findings align closely with similar findings in the long-tailed finch in which individuals (both males and females showed high levels of site fidelity across a number of years (van Rooij & Griffith 2011).

Southern black-throated finch are a flock living species which display similar movement requirements between sexes (Laguna *et al.*, 2019; van Osta *et al.*, 2024b). Group living or flocking is often related to the abundance and distribution of the foraging resource, as well as perceived predation risk while foraging, and is more prevalent in species which consume a plant diet (Beauchamp, 2022). Southern black-throated finch are therefore not likely to have movement ecology differences between sexes as males and females appear to move together as part of flocks, and this is similar to what has been more directly observed in the closely related zebra finch where a male and female will remain in a close pair bond throughout the year and they typically move together very closely (Tyson, Loning, *et al.*, 2024).

It must also be noted that often movement differences are more likely to be apparent in juvenile birds, including juvenile dispersal (Maness and Anderson, 2013; Weston *et al.*, 2013; Lequitte-Charransol, Robert and Jiguet, 2024). However, due to ethical constraints, as our study focussed primarily on adult birds and their movement habits, we did not assess juvenile sex-biased dispersal. This would be a valuable component to assess in future surveys.

3.4.5.4 Diet

Our results suggest that the diet of southern black-throated finch did not substantially vary between sexes, nor was there an impact of season on the sex-related variation of the species diet. Evidence shows that in species where there are distinctive reproductive roles, dietary divergence of feeding structures, including foraging behaviour and diet, between males and females often exists, as this is attributed to differences in energetic and nutritional requirements (Olsen et al., 2013). For example, where male-male competition and female choice is present, such as within polygynous species, this arrangement often favours larger males and hence male-larger dimorphism due to allometric scaling is often observed (Shaffer, Weimerskirch and Costa, 2001; Andersson and Simmons, 2006; Fairbairn, 2013). Such is the case of the Cory's shearwater (*Calonectris diomedea*), which exhibit differences in beak size between sexes, due to antagonistic competition (Navarro, Kaliontzopoulou and González-Solís, 2009), and the great bustard (*Otis tarda*) which differ between sexes due to the distinct reproductive roles and body size (Ruckstuhl and Neuhaus, 2006; Bravo *et al.*, 2016). Southern black-throated finch are known to share parental care and have similar movement ecologies between sexes, which likely contributes to similar dietary requirements (Higgins, Peter and Cowling, 2006; Laguna *et al.*, 2019).

Dietary dimorphism has been documented as a tactic to reduce resource competition and availability of resources on a spatial and temporal scale across the landscape between the sexes, known as foraging niche divergence (Ruckstuhl and Neuhaus, 2006; Bravo *et al.*, 2016). In the case with the sympatric

species, Collett et al. (2022b) found that Gouldian finch (*Chloebia gouldiae*), long-tailed finch (*Poephila acuticauda*) and masked finch (*Poephila personata*) showed plasticity in ecological niche overlap among the three finch species depending on grass seed availability within the landscape and diversified their diet in times of resource scarcity. The southern black-throated finch is a granivorous bird, with a diet which comprises primarily of a wide variety of grass seeds and occasionally insects where available (Laguna *et al.*, 2019). Furthermore, the species occurs in mixed-sexed flocks, which includes flock feeding (Shephard, Pridham and Forshaw, 2012; Laguna *et al.*, 2019). Joint selection of foraging sites by the mixed-sex flock also likely contributes to the similar diet observed between male and female southern black-throated finch (Goodale *et al.*, 2015; Richardson *et al.*, 2022). It is known that mixed flocks often comprise of species with distinct foraging niches, however most species exhibit foraging niche overlap (Jones, Walters and Robinson, 2020). This is consistent with the facilitation hypothesis, which indicates that similar foraging species were significantly more likely to associate in flocks (Jones, Walters and Robinson, 2020; Zhang *et al.*, 2020). Therefore, in the case of the southern black-throated finch, whereby it is a flock living bird where sexes forage at the same sites, it is evident that there is no effect of foraging site selection on dietary niche partitioning between sexes.

3.4.6 Conclusions

Our study found that the southern black-throated finch exhibits some degree of sexual dimorphism, however, does not show sex-related variation in their home ranges, dispersal movements and diet. Our study also identified that the southern black-throated finch population in the Desert Uplands, one of the two remaining stronghold populations for this endangered species, has a balanced adult sex ratio, which provides a baseline for ongoing monitoring of the species and can be compared to other populations of the species.

Our findings suggest that conservation actions for the southern black-throated finch do not need to consider sex-specific ecological requirements at this point in time, although future studies should examine sex-biased juvenile dispersal as a potential source of selection on males and females separately.

3.5 Southern black-throated finch population estimate

The BTFMP requires a population estimate to be developed at year 5 of the monitoring program to monitor ongoing impacts to the population and inform management actions. The BTFMP specifies that a population estimate for 10 Mile Bore must also be developed along with the population estimate for the broader study area. However, the results of this research program that have been undertaken after the BTFMP identify that the SBTF populations within the study area are more appropriately split into a northern subpopulation that occurs north of Moray-Carmichael Boundary Road and a southern subpopulation that occurs south of the Carmichael River, which is evidenced by little movement and genetic exchange between these subpopulations (Sections 3.2 and 3.3). Furthermore, substantial movement of SBTF occurs among Ten Mile Bore and water sources within 3–4 km surrounding Ten Mile Bore (CWAT301, CWAT302 and CWAT303), which constrains our ability to clearly define and assess a population size for the Ten Mile Bore Area in isolation. Population estimates provided in this section have therefore been presented using the northern and southern subpopulation groupings of SBTF.

Population estimates of SBTF have been produced using three methods, which are calculated from multiple, largely independent datasets. These include a:

- Mark-resight method that uses colour band resighting data
- Distance sampling method that uses SBTF observations on 1 km walked transects; and
- Effective population size estimation using SBTF genetic data. Calculation of effective population sizes are detailed in Section 3.2, and key results are reproduced in this section for ease of comparison.

3.5.1 Mark-resight estimate

A mark-resight model is used for study designs where animals are marked during capture events, which are followed by 'resighting' occasions, where marked individuals are identified but are not marked (McClintock, White and Pryde, 2019). Colour-banding during mist-netting events, followed by resighting of colour banded birds through walked survey, water source watches or incidentally, follow a mark-resight model.

Based on the model requirements and recommendations provided by Cooch and White (2019) and McClintock, White and Pryde (2019), a zero-inflated poisson-log normal mark-resight model was selected as the model allows for:

- Animals to immigrate/emigrate into/from the study area. Radio telemetry of SBTF found two occasions of SBTF moving outside of the study area boundary towards a water source on Carmichael Station to the west (John van Osta pers. obs.).
- Marks are uniquely distinguishable, while the model still allows input of records where a mark has been partially identified.
- Individuals to be sighted more than once per sampling event.
- The population of marked individuals to be unknown (as marked SBTF may emigrate from the population).
- Is suitable for low encounter rates.

3.5.1.1 Data collection

The BTFRP included a mist-netting program that allows for an analysis of SBTF population size through a mark-resight methodology. Approximately half of the SBTF were fitted with uniquely identifiable colour bands that allow resighting of known birds without recapture. The remaining SBTF were fitted with a radio transmitter and a uniquely identifiable metal tag that allows identification of known birds upon recapture. To date, 10 mist-netting events have been conducted where SBTF were captured via mist-netting. The number of SBTF colour-banded and resighted during each survey period are identified within Table 3.5-1.

All resightings of marked birds occurred within the northern subpopulation of SBTF that occurs north of Moray-Carmichael Boundary Road (Table 3.5-1). As such, the population estimates presented here are specific to the northern subpopulation of SBTF within the study area.

Table 3.5-1: Number of marked SBTF captured and resighted during survey periods

Survey	BTF capture and colour band application		BTF recapture and resighting			
	Number of SBTF captured (mist-netting)	Number of SBTF colour banded	SBTF recaptured		Colour bands resighted	
			Total	Different individuals	Total	Different individuals
2020 dry season (August–September)	45	22	1	1	4	3
2021 early wet season (February)	36	14	1	1	5	3
2021 wet season (March)	22	10	0	0	23	17
2021 dry season (September–October)	70	38	9	9	68	40
2022 early wet season (February–March)	80	63	8	7	21	17
2022 wet season (June–July)	93	65	17	17	112	73
2022 dry season (October)	103	51	13	13	37	32
2023 early wet season (March)	113	13	15	13	45	31
2023 wet season (May-June)	148	5	11	11	63	38
2023 dry season (October)	64	4	13	13	36	29
Total	**774**	**285**	**88**		**414**	**283**

3.5.1.2 Data analysis

Program MARK (White and Burnham, 1999) Version 9.0 was used to construct a zero-inflated poisson-log normal mark-resight model, in accordance with the model requirements identified within Cooch and White (2019) and McClintock, White and Pryde (2019). Data inputs into the model included the capture and resight histories of each SBTF, the number of unbanded SBTF observed during each survey and the number of colour banded SBTF observed during a survey whose colour band combination could not be identified with confidence. Program MARK was used to run models with all parameter variants and use model averaging in accordance with Doherty, White and Burnham (2010). Candidate models were ordered based on the goodness of fit as identified by Akaike's Information Criterion (AIC) (Doherty, White and Burnham, 2010; McClintock, White and Pryde, 2019). Parameter estimates, including the population estimates, from the models were averaged using Program MARK's inbuilt model weighting function, which weights the best-fitting models highest (Cooch and White, 2019).

3.5.1.3 Results

The mark-resight population estimate for the northern sub-population within the study area ranged from 519 (95% confidence interval [CI] = 378–663) in February 2021 to 604 (95% CI = 477–730) in March 2023 (Table 3.5-2). The population estimates of SBTF remained broadly stable over the four year study period, with the 95% confidence interval ranges overlapping among the estimates for all survey periods (Table 3.5-2).

Table 3.5-2: Population size estimates for the northern subpopulation of SBTF within the study area calculated using the mark-resight method

Survey period	Estimated population size	Lower 95% confidence interval	Upper 95% confidence interval
2020 dry season (August–September)	520	378	663
2021 early wet season (February)	519	317	720
2021 wet season (March)	521	429	614
2021 dry season (September–October)	572	459	685
2022 early wet season (February–March)	561	349	774
2022 wet season (June–July)	599	504	693
2022 dry season (October)	586	407	765
2023 early wet season (March)	604	477	730
2023 wet season (May–June)	552	461	644
2023 dry season (October)	567	469	665

3.5.2 Distance sampling estimate

Distance sampling is a method for estimating the density and number of wildlife (Buckland *et al.*, 2015). Distance sampling involves systematically recording the distance of detected objects or animals from a designated line or point (typically referred to as line or point transects). For line transect sampling,

observers travel along a predetermined path, recording the perpendicular distance from the transect line to each detected individual or group (Thomas *et al.*, 2010). The population density can then be estimated using a function of the likelihood of detection decreasing with distance from the observer (Buckland *et al.*, 2015).

3.5.2.1 Data collection

The distance transect program comprised 216 unique 1 km transects that were included in repeat monitoring events across 11 survey periods between September 2020 and October 2023 (Table 3.5-3). Distance transects included those undertaken for both the BTFRP (n = 679 km of transect) and the SBTF Monitoring Program (n = 285 km of transect). An average of 74 km of transects were walked each survey (range = 28–168 km), within each an average of 13 flocks of SBTF were recorded (range = 1–16). Data were summarised into yearly totals for the purposes of population size analysis, to smooth survey-to-survey stochasticity in detection rates given the low numbers of detections per survey (Table 3.5-3).

Distance transects were also undertaken in 2024 under the BTFMP; however, these transects have been excluded from this analysis as no BTFRP transects were undertaken in 2024, which resulted in a substantially different distribution of 1 km transects within the study area than compared to previous years.

Table 3.5-3: Number of 1 km transects undertaken and southern black-throated finch recorded during the study period

Survey	Number of 1 km transects	Number of SBTF flocks	Number of SBTF
2020 wet season (May-June)	28	1	1
2020 dry season (August-September)	112	9	33
2020 Total	**140**	**10**	**34**
2021 early wet season (February)	111	8	22
2021 wet season (March)	117	16	46
2021 dry season (September-October)	168	8	42
2021 Total	**396**	**32**	**110**
2022 early wet season (February-March)	104	11	24
2022 wet season (June-July)	31	1	10
2022 dry season (October)	76	9	20
2022 Total	**211**	**21**	**54**
2023 early wet season (March)	42	9	33
2023 wet season (May-June)	83	1	4
2023 dry season (October)	29	2	24
2023 Total	**154**	**12**	**61**

3.5.2.2 Data analysis

The perpendicular distance of each SBTF record to the 1 km transect was calculated using ArcGIS Pro. A conventional distance sampling analysis engine was constructed within the Distance software (version 7.5), which models the probability of a SBTF flock being detected relative to the flock's perpendicular distance to the transect (Thomas *et al.*, 2010). A global detection function was modelled within the Distance software, which assumed that the detectability of SBTF was constant among years (Thomas *et al.*, 2010). All records greater than 150 m (n = 3) from the transect line were truncated, with these records presumed to be erroneous and potentially having a disproportionate effect on the SBTF density estimates (Buckland *et al.*, 2015). The encounter rate was set as a global variable, which assumed that SBTF had a constant detectability among years.

The distance sampling method produces a density estimate of SBTF within the area that the transects represent. For our study area, the transects have been concentrated in SBTF habitat, with the majority of the transect length occurring in the regional ecosystem 10.5.5a (*Eucalyptus melanophloia* woodland), which was the dominant regional ecosystem mapped as SBTF habitat by the Species Distribution Model that was used to stratify distance transect locations (Section 2.1). The area of habitat containing regional ecosystem 10.5.5a, totalling 34,750 ha, was therefore used as the area of potential SBTF habitat that the distance transects represent. This approach allowed a single SBTF density estimate across the study area, which minimised the number of habitat categories within which SBTF density was estimated to mitigate stochastic variation in SBTF density estimates from small numbers of SBTF flocks recorded during 1 km transects.

3.5.2.3 Results

The distance sampling population estimate for the study area ranged from 1100 (95% confidence interval [CI] = 457–2643) in 2020 to 1517 (95% CI = 864–2664) 2022 (Table 3.5-4). The population estimates of SBTF remained broadly stable over the four year study period, with the 95% confidence interval ranges overlapping among the estimates for all years (Table 3.5-4).

Table 3.5-4: Population size estimates derived from distance sampling

Year	Estimated population size	Lower 95% confidence interval	Upper 95% confidence interval
2020	1100	457	2643
2021	1221	758	1967
2022	1517	864	2664
2023	1188	578	2445

3.5.3 Water source watch census estimate

3.5.3.1 Data collection

Water source watches are a key survey method recommended in the *Significant Impact Guidelines for the Endangered Black-throated Finch (southern subspecies; Poephila cincta cincta)* (DEWHA, 2009b). This method has proven to be the most effective in consistently detecting SBTF during the surveys conducted to date with low observer variation. However, using water source watch results to track population changes has been made difficult due to an increase in in the number of permanent and ephemeral water sources within the study area since 2014. These additional water sources have the effect of spreading

SBTF over more water sources, thereby reducing SBTF sightings at the routinely monitored waterbodies. To address this, a 'census' style water source watch survey was introduced in 2024 with the aim of addressing these issues and providing a rough estimate of SBTF population that could be compared year to year.

To maximize the proportion of the SBTF population recorded, the census was carried out in the late dry season when SBTF rely most on permanent water sources and ephemeral water sources are not available. In addition, data for the census was captured in the heat of the day when drinking rates are likely to be highest (van Osta *et al.*, 2024). All accessible water sources were monitored, with large water sources such as dams monitored by ecologists, supplemented by remote cameras placed on small water sources such as troughs and ephemeral scrapes. Water source watches were carried out over two days to monitor the northern and southern subpopulations as per the findings of the genetic studies (Section 3.2). All water source watches were undertaken simultaneously over a period of six hours from 9am to 3pm. Over the two days, a total of 38 water sources were surveyed, including 18 surveyed by ecologists and 20 surveyed with remote cameras (Table 3.5-5).

Table 3.5-5: Survey effort during water source census per methodology and subpopulation

Survey method	Number of water sources (north)	Number of water sources (south)	Total water sources
Water source watches	12	6	**18**
Remote cameras	15	5	**20**
Total water sources	**27**	**11**	**38**

3.5.3.2 Data analysis

Although the use of remote cameras allowed for all water sources to be monitored simultaneously with significantly less survey effort, remote cameras often capture multiple images of the same individuals during a detection event. This limitation meant that the number of photos of SBTF recorded by the cameras cannot be directly compared to the number of records from human observers during water source watches. To address this, a remote camera detection event was defined as the maximum number of SBTF observed within any single photo within a set time period (Green *et al.*, 2023). A calibration exercise was undertaken during the 2024 dry season survey to determine to optimal length of the time period over which sequential photos should be grouped, which involved simultaneous remote camera and human water source watches. Remote cameras results calibrated using a range of time periods were compared to results of the water source watches (Figure 3.5-1). The most similar results between the two methods were produced when a time period of 10 seconds was used to calibrate the remote camera results, so this was used as the defined detection event for data analysis. Grouping periods of less than 10 seconds over estimated SBTF numbers, while grouping periods greater than 10 seconds under-estimated SBTF numbers because they merged counts of multiple flocks or drinking events by different individuals.

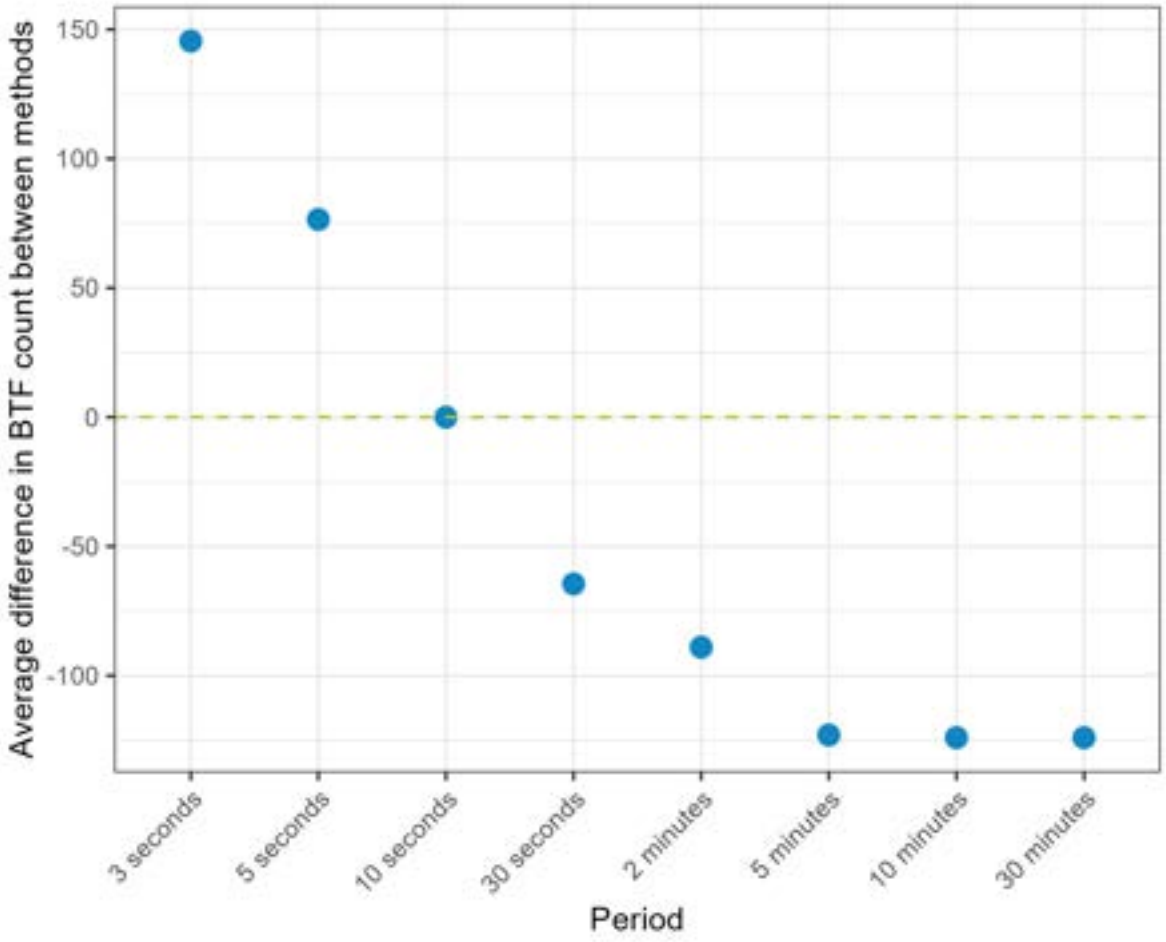


Figure 3.5-1: A calibration curve showing the average difference in SBTF count between a water source watch and remote camera results grouped by various observation periods, ranging from 3 seconds to 30 minutes.

Cameras were deployed for a minimum of five days and an average SBTF use over the five days was calculated to account for potential variation in detection rates. This average, in combination with the total SBTF recorded during the water source watches, produced a population estimate of the SBTF utilising water sources at that point in time.

3.5.3.3 Results

The water source watch census population estimate for October 2024 was 652.8, consisting of 554.8 records from the northern subpopulation and 98 from the southern subpopulation (Table 3.5-6). All remote camera records were from two water sources in the north of the study area (CWAT302 and CWAT301) whereas the water source watches produced a similar number of records from the north and south of the study area. The lack of remote camera records in the south of the study area is likely due the differences in water sources utilised most by SBTF. The water source most used by SBTF in the south is Lignum dam (CWAT111) which is a large dam with an extensive boundary that is not suitable for remote camera monitoring. In comparison, the most used water sources in the north are troughs that are easily monitored using remote cameras.

Table 3.5-6: Number of SBTF recorded during water source census per methodology and subpopulation

Survey method	Number of SBTF (north)	Number of SBTF (south)	Total SBTF
Water source watches	87	98	**185**
Remote cameras	467.8	0	**467.8**
Total SBTF	**554.8**	**98**	**652.8**

During the 2024 dry season survey, when the census occurred, remote cameras recorded a more diverse drinking pattern compared to the previous wet season survey, as shown in Figure 3.5-2. Daily temperatures during the 2024 dry season water source census period were extremely hot, with site temperatures recorded at 38ºC and 41ºC on days one and two of the census, respectively. In comparison, temperatures during the 2024 wet season survey were on average 26ºC (range 19–29ºC). The remote camera results suggest that under these high temperature scenarios, some SBTF may switch to a bi-daily drinking pattern, which had the effect of distributing the remote camera records throughout the day, rather than exhibiting a peak in activity around the heat of the day, as seen in the 2024 wet season (Figure 3.5-2). Another potentially confounding factor was that the majority of 2024 dry season remote camera records were from CWAT302 (Spider Alley), at which SBTF were observed foraging within 1 km of the trough. The close foraging area may reduce the cost of travel for SBTF and potentially promote more regular drinking. This variation in drinking patterns may have influenced the results of the census as a key assumption of the population estimate is that a majority of the population visited the water source during the survey period. Capturing less of the daily activity during the survey period may have resulted in a lower population estimate; however, it is difficult to determine whether this is the case without data to compare to.

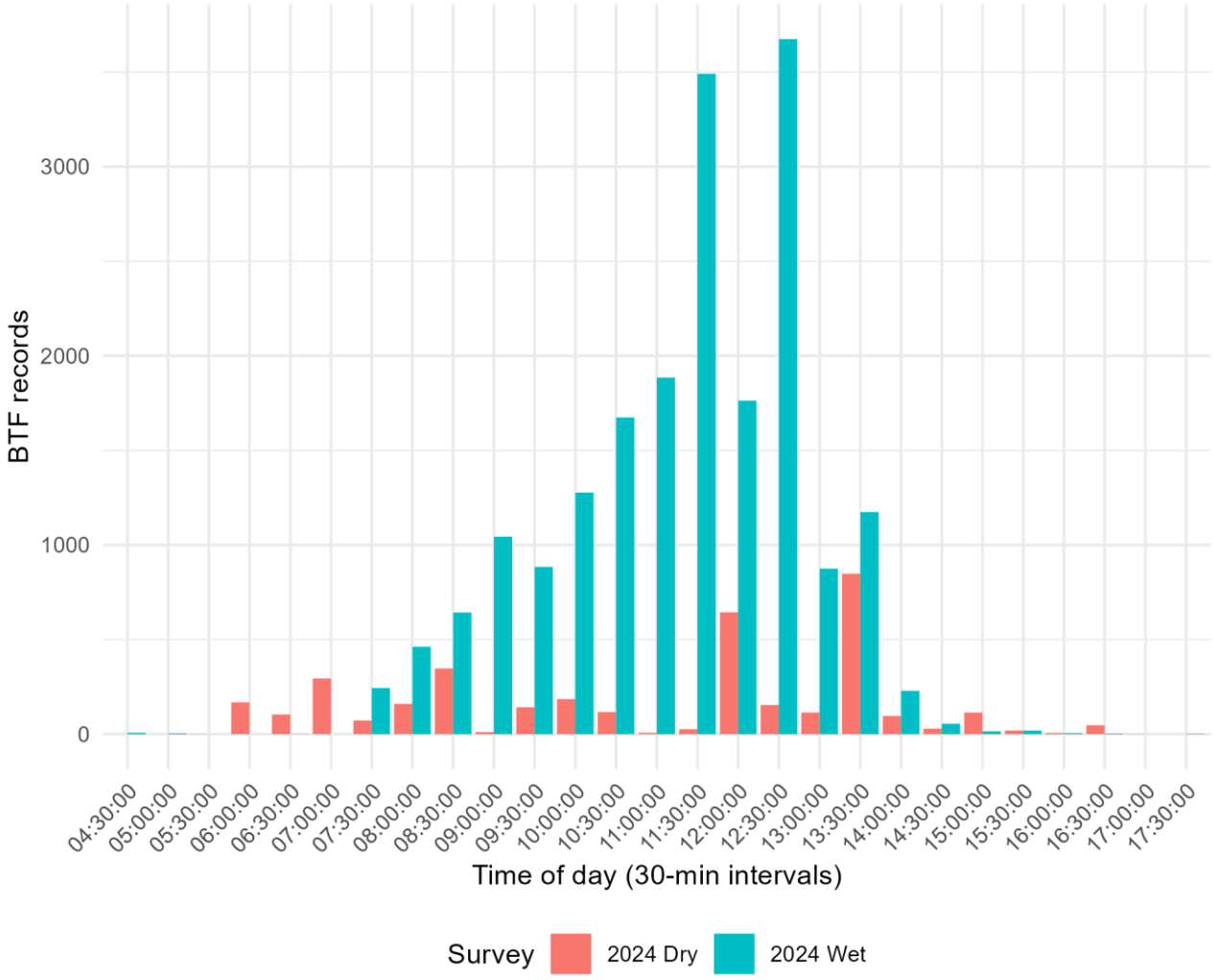


Figure 3.5-2: SBTF records throughout the day at water source monitoring sites during the 2024 dry and wet season surveys. The number of BTF records are not directly comparable between the surveys due to differences in the number of cameras deployed and sites at which they were deployed.

3.5.4 Comparison of southern black-throated finch population estimates

Population estimates of SBTF have been derived from four sources, which include:

- Genetic analysis to estimate the effective population size (Section 3.2)
- Mark-resight population estimate (Section 3.5.1)
- Distance sampling population estimate (Section 3.5.2)
- Water source watch census estimate (Section 3.5.3)

These estimates have been produced through independent datasets, which afford multiple lines of evidence to determine the population size of SBTF. However, there are fundamental differences in the methodologies that result on the population estimate representing different spatial extents and interpretation of the 'population', which must be considered when comparing the methods.

Estimation of effective population sizes (N_e) using DNA-based approaches are well established (Luikart *et al.*, 2010). N_e is a measure of the genetic diversity of the species population (Luikart *et al.*, 2010). While N_e is correlated with the species actual population size (referred to as the 'census population size', N_c), the two values may be substantially different (Luikart *et al.*, 2010; Palstra and Fraser, 2012). Frankham (1995) found that the average ratio of N_e to N_c was 0.1 among many lifeforms, and on average 0.33 among three species of finches examined (*Geospiza* spp.). Applying a 0.33 ratio to the N_e estimate (Table 3.5-7), suggests an approximate N_c of 3,839 (95% CI: 3,148–4,909); which includes SBTF populations within the Project Area, Lignum Station, Hyde Park Station as well as adjoining SBTF populations that are genetically connected. Considering this larger area, the population estimate yields a similar density (0.05 SBTF per ha (95% CI: 0.04–0.06)) to most other methods. However, defining the exact spatial area covered by this population estimate may not be feasible without further regional genetic analysis of the species (Section 3.2).

The spatial coverage of distance sampling is well defined compared to genetic and to a lesser extent mark-resight estimates. Distance transects were distributed throughout SBTF habitat within the Project Area and sampled SBTF primarily in their foraging and nesting habitats (away from water sources). As such, distance transects account for SBTF that may forage within the Project Area but drink on adjacent properties. The distance sampling method produced a Nc estimate of 1,188 (95% CI: 578–2,445), which covers the spatial extent of all SBTF habitat within the Project Area (Table 3.5-7). Despite sampling the entire Project Area independently of water sources, the distance sampling estimate produced a similar to other methods, with 0.03 SBTF per ha (95% CI: 0.02–0.07). However, the focus of monitoring is detecting change in the population, which distance sampling estimates are unlikely to effectively show due to a broad 95% confidence interval.

Mark-resight population estimates are focused on water sources, due to the practicalities of capturing and resighting SBTF. The extremely small number of colour-band resightings south of the Carmichael River prevented a mark-resight population estimate of SBTF from the southern sub-population. Furthermore, resighting of SBTF was focused on water sources included within the SBTF monitoring program, which may result in some of the lesser used water sources within the Project Area being under-represented in the mark-resight population estimate. The mark-resight method produced a N_c estimate of 567 (95% CI: 469–665), which covers all SBTF habitat within 4 km of all resight locations within the Project Area (Table 3.5-7). Furthermore, this estimate excludes SBTF that may forage and/or nest within the Project Area but drink on adjacent properties.

The mark-resight study encompassed all water sources where SBTF were known to drink in the Project Area. However, no resights of colour bands were recorded south of the Carmichael River and therefore this population estimate is constrained to the northern sub-population of SBTF within the Project Area. Furthermore, small numbers of SBTF that drink on adjacent properties but forage within the study area will not be included within this population estimate.

The water source watch census method produced an estimate of 653.8 over a spatial extent that includes all interconnected patches of habitat within the Project Area that contain a surveyed water source (Table 3.5-7). This spatial extent is determined based on the assumption that all individuals within the habitat will need to drink from a water source within the survey window as the survey was undertaken in the late dry season when SBTF rely most on permanent water sources and in the heat of the day when drinking rates are likely to be highest (van Osta *et al.*, 2024). Similarly to the mark-resight population estimate, SBTF that drink on adjacent properties but forage within the Project Area will not be included within this population estimate. In addition, as this estimate is purely a count of the number of SBTF visiting the water sources, a confidence interval cannot be produced. Similarly to the mark-resight population estimate, the census estimate is also focused on water sources; however, each water source is sampled equally.

Table 3.5-7: Comparison of southern-black throated finch population estimates

Method	Spatial scale of estimate	Population estimate (95% confidence interval [CI])	Density per ha (95% CI)	Trend in population 2020-2023
Genetics	Habitat within the Project Area and adjoining properties, totalling ~81,500 ha.	N_e: 1,267 (95% CI: 1,039–1,620) N_c (estimate assuming 0.33 N_e/N_c ratio): 3,839 (95% CI: 3,148–4,909)	N_c: 0.05 (95% CI: 0.04–0.06)	NA
Mark-resight	Habitat within 4 km of all resighting locations within the Project Area and adjoining properties, totalling ~53,600 ha.	N_c: 567 (95% CI: 469–665), in October 2023	0.011 (95% CI: 0.009–0.012)	Stable
Distance sampling	Habitat within the Project Area, totalling ~34,750 ha.	N_c: 1188 (95% CI: 578–2445)	0.03 (95% CI: 0.02–0.07)	Stable
Water source watch census	Habitat within the Project Area that contains a surveyed water source, totalling ~34,750 ha.	N_c: 653.8	0.02	NA

4 Aim 2: Determine the foraging and breeding habitat requirements and the relationship with vegetation structure and composition and water

4.1 Nest site and habitat selection

4.1.1 Abstract

Knowledge of a species' habitat requirements can lead to improved conservation and management strategies, particularly for threatened species with disjunct populations. This study examines the nesting habitat and nest site selection of the Endangered southern black-throated finch in the Desert Uplands bioregion in Queensland, Australia. Surveys identified 175 black-throated finch nests, primarily in *Eucalyptus melanophloia* and *Eucalyptus similis* savanna woodlands with sparse canopy and shrub layers. The majority of nests (58%) were located in *Acacia sericophylla*, with *Eucalyptus melanophloia*, *Melaleuca nervosa* and *Bursaria incana* also commonly used. Nests were constructed from locally sourced grasses and forbs and were typically well concealed amongst the foliage of the host tree, potentially as nest predation risk strategy. The factors influencing nest site location were determined using generalized linear models, with the best predictive model identifying that NDVI, canopy height, shrub cover, grass cover and bare ground cover were the key variables affecting black-throated finch selection of nest sites. The model determined that areas of high NDVI, low shrub cover and a mix of grass and bare ground cover were favoured. This suggests that the black-throated finch have a preference for productive vegetation in heterogeneous landscapes with accessible resources for nest construction and foraging. Nest sites were typically located relatively far from water sources, , potentially due to the sparser availability of water in the Desert Uplands landscape. These findings highlight the necessity for targeted conservation strategies that address the requirements of disjunct populations of the black-throated finch. Future research should focus on the impacts of predation, temporal nest site selection, and the potential effects of climate change on nesting habitat availability.

4.1.2 Introduction

The conservation and effective management of threatened species requires knowledge of their biology and threats, particularly focusing on filling knowledge gaps that impede management (Scheele *et al.*, 2018). In particular, knowledge of their breeding biology is critical to facilitate the protection and management of habitat features that affect their breeding productivity and survival (Martin, 1992; Frankham, 2010a; Xiao *et al.*, 2017; Hale *et al.*, 2020). However, rare or little-known species can be problematic to manage as they can be difficult to detect, and studies may be both costly and time consuming (Raphael and Molina, 2007). Current methods to identify critical habitat for threatened species may fail to support their recovery because identifying necessary habitats takes time, during which the species continue to decline (Martin *et al.*, 2017; Palm *et al.*, 2020).

The selection of a nest site within a habitat is a key component for successful reproduction and ongoing survival of bird species and may be influenced by numerous features including food and water availability, vegetation characteristics as well as risks and protection from predators (Martin, 1995; Clark and Shutler, 1999; Weidinger, 2002). The selection of nest sites may be influenced by factors at multiple spatial scales including characteristics of the host tree or shrub, habitat surrounding the nest and habitat features

within the broader landscape (Johnson, 1980; Morris, 1987; Martin and Roper, 1988; Block and Brennan, 1993; Latif *et al.*, 2020; Wang *et al.*, 2020). At a broader scale, birds are likely to initially select the type of vegetation cover in which to breed, i.e. nesting habitat, and then select specific nest sites on the basis of individual vegetation structure and species (Johnson, 1980; Martin, 1998; Rotenberry and Wiens, 1998). Therefore it is important to consider the influence of spatial variation in vegetation structure from the site to landscape scale when studying habitat use by birds (Morris, 1987; Wiens, Rotenberry and Van Horne, 1987; Moen and Gutiérrez, 1997; Otis, 1997; Sica *et al.*, 2020; Nugent *et al.*, 2022; Davison *et al.*, 2023).

Vegetation structure and species composition have been identified as important factors when determining the distribution, abundance, habitat use and nest site selection by birds (Block and Brennan, 1993; Moen and Gutiérrez, 1997; Rotenberry and Wiens, 1998; Polak, 2007). The structural characteristics of vegetation provide birds with important resources such as perches, nest and roost sites, foraging substrates and shelter from predators (Misenhelter and Rotenberry, 2000). The composition of plant species has been demonstrated to strongly influence the distribution of birds, with some having strong associations and preferences for particular habitat types or certain flora species (Martin, 1993, 1998; Rotenberry and Wiens, 1998). Therefore, it is important to examine both vegetation structure and floristics at multiple scales when studying the relationship between birds and habitat selection.

Nesting habitat for birds may require specific resources and environmental conditions that differ from other activities such as foraging and these resources may occur in spatially distinct locations from other resource requirements (Wiens, 1986; Southwood, 1988; Block and Brennan, 1993; Martin, 1995; Mansouri *et al.*, 2022; Berlusconi *et al.*, 2024). Nesting habitat may consist of a higher density of important food and water sources to improve breeding success (Holway, 1991; Misenhelter and Rotenberry, 2000; Hosseini-Moosavi *et al.*, 2017; Mansouri *et al.*, 2022). Nest sites may also be selected by birds to increase breeding success by reducing the risk of predation through increased concealment and reduced audio and olfactory cues as well as providing protection from rain and direct radiation heat and to regulate temperature (Martin and Roper, 1988; Holway, 1991; Martin, 1993; Endo, 2012; Hosseini-Moosavi *et al.*, 2017; Salema, Gale and Bumrungsri, 2018).

Land use and management practices, such as pastoralism and altered fire regimes, can have a profound effect on the availability and quality of resources in woodland habitats (Liedloff *et al.*, 2001; Kutt and Woinarski, 2007; Woinarski and Legge, 2013). This impact to critical resources has contributed to the decline of granivorous birds, which are overrepresented among Australia's threatened bird species (Franklin, 1999; Franklin, Woinarski and Noske, 2000; Franklin *et al.*, 2005; Garnett, Baker and ProQuest, 2021). Despite savanna woodlands in northern Australia being sparsely settled, with relatively intact vegetation and subject to low-intensity pastoralism, around 25% of granivorous birds are declining in abundance (Franklin, 1999; Franklin *et al.*, 2005; Woinarski *et al.*, 2011; Edwards, Russell-Smith and Meyer, 2015).

Effective management of threatened species requires knowledge of the target species' biology and threats, particularly focusing on filling knowledge gaps that inhibit effective management and recovery (Scheele *et al.*, 2018). Understanding the requirements of nesting habitat therefore enables the protection, restoration and management of these areas for threatened species. While identifying critical habitat is an essential step in the threatened species recovery process, the absence of effective protection, management and regulation means the recovery of many threatened species is unlikely to be successful (Camaclang *et al.*, 2015; Martin *et al.*, 2017; Woinarski *et al.*, 2017). Information from studies into threatened species can be used to develop more effective management strategies and regulations to improve conservation efforts.

The black-throated finch (*Poephila cincta*) is a granivorous bird with two subspecies. The southern subspecies of the black-throated finch (*Poephila cincta cincta*) is listed as Endangered under

Commonwealth and Queensland legislation due to a large population decline and a range contraction over the past 30 years (BTFRT, 2007; DEWHA, 2009). The clearing and fragmentation of habitat and changes in land use, particularly pastoralism and agriculture, have rendered other extant black-throated finch habitat unsuitable due to habitat degradation associated with grazing pressures, the introduction of exotic pasture grasses and altered fire regimes (Franklin *et al.*, 2005; Vanderduys *et al.*, 2016b; Mula Laguna *et al.*, 2019). Remaining black-throated finch populations appear to be limited to the Townsville Plains and the Desert Uplands regions in Queensland. There is also anecdotal evidence to suggest that the Desert Uplands bioregion, and the Alice Tablelands subregion in particular, may support a larger black-throated finch population that that of the Townsville Plains (Buosi *et al.*, 2021; BTFRP, unpublished).

There has been comparatively little research completed on the Desert Uplands population of black-throated finch to date, and little is known of the species' nesting habitat requirements in this region (Rechetelo, 2015; Laguna *et al.*, 2019). Studies from the Townsville Plains population are unlikely to be meaningful for understanding the breeding ecology of black-throated finch in the Desert Uplands due to differences in habitats, vegetation composition, climate and availability of water between the two regions. The Desert Uplands bioregion has a semi-arid climate with predominantly low fertility soils with poor structure formed on sand plains and sandstone ranges in contrast to the more fertile soils and higher rainfall of the Brigalow Belt bioregion, which includes the Townsville Plains subregion (Sattler, 1999; Bastin *et al.*, 2009). The Alice Tablelands subregion of the Desert Uplands is dominated by savanna woodland vegetation communities characterised by *Eucalyptus similis*, *Acacia shirleyi*, *E. melanophloia* and *E. brownii* and contains extensive salt lakes and fringing dunes, whereas the Townsville Plains consists of *E. platyphylla*, *E. crebra* and *Corymbia dallachyana* dominated woodlands as well as salt marsh, beach scrub and mangrove communities (Sattler, 1999).

Although the decline of granivorous birds in Australia are well documented (Franklin, 1999; Franklin *et al.*, 2005), a lack of knowledge of the ecological requirements for many of these species hinders the ability to implement effective conservation and management measures (Garnett, Baker and ProQuest, 2021). To effectively manage and conserve the habitat and populations of black-throated finch in the Desert Uplands bioregion, it is critical to understand their nesting habitat requirements. We investigated the nest site selection of black-throated finch at multiple spatial scales to determine the factors influencing nesting habitat.

4.1.3 Methods

4.1.3.1 Study area

The study was undertaken over approximately 75,770 ha centered on 21° 59' S, 146° 21' E and located approximately 150 km north-west of Clermont and 190 km south of Charters Towers (Figure 4.1-1), within the Alice Tablelands subregion of the Desert Uplands Bioregion in central Queensland. The Desert Uplands Bioregion coincides with the southern extent of the current known range of the black-throated finch as well as the Galilee Basin which is subject to numerous proposed and approved mining projects with impacts to >20,000 ha of potential black-throated finch habitat approved to date (Vanderduys *et al.*, 2016b; Reside *et al.*, 2019).

The Desert Uplands occurs within a hot semi-arid Köppen-Geiger climate zone and the Townsville Plains is within the tropical savanna with a dry winter climate zone (Peel, Finlayson and McMahon, 2007). Vegetation in the Desert Uplands is characterised by eucalypt and acacia woodlands with a spinifex understorey occurring on sandstone ranges and sand plains with low fertility soils while the Townsville Plains consists of alluvial plains and marine deposits surrounded by sedimentary and igneous hills that support eucalypt and melaleuca woodlands (Sattler and Williams, 1999; Bastin *et al.*, 2009, 2009). Land use in the Desert Uplands is dominated by pastoralism whereas the Townsville Plains has mixed land use

including urban and industrial areas resulting in more fragmented habitats (Bastin *et al.*, 2009; DES, 2018b).

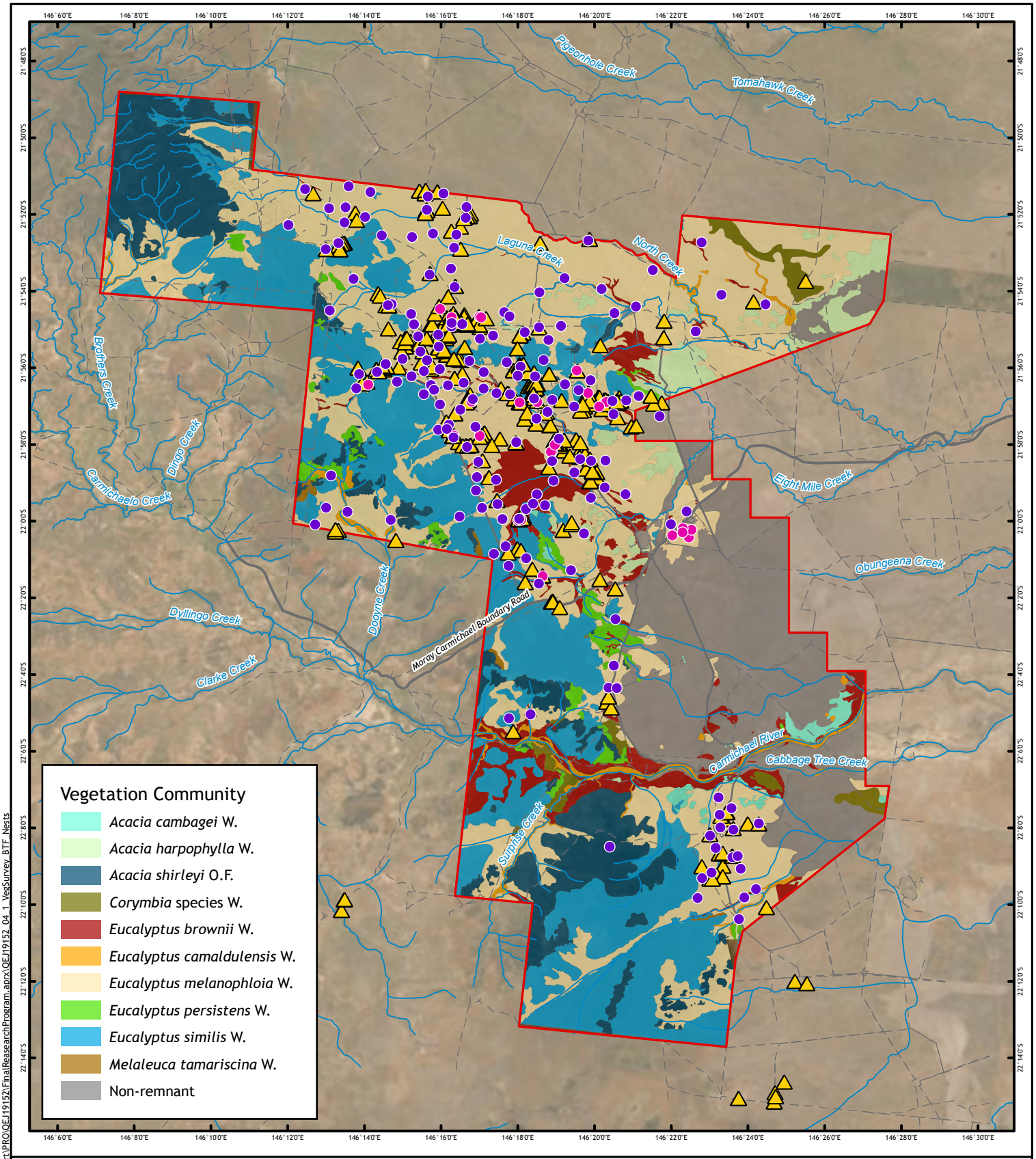
Table 4.1-1 Climatic and vegetation characteristics of the Townsville Plains and Desert Uplands which host the remaining black-throated finch 'stronghold' populations

Environmental characteristic	Townsville Plains	Desert Uplands*
Mean annual rainfall (DES, 2013a)	968.2 mm	588.7 mm
Woody cover (DES, 2018b)	60.9%	80.7%
Wetlands as a percent of total subregion area, excluding estuarine wetlands	8.5%	3%

* Values reported for the Alice Tableland subregion of the Desert Uplands, which contains the majority of black-throated finch records and which the study area occurs within (Mula Laguna *et al.*, 2019; BTFRP, unpublished).

Land use within the study area consists of cattle grazing and native vegetation management within a ~33,400 ha environmental offset set aside under the Commonwealth *Environment Protection and Biodiversity Protection Act 1999*, and approximately 33,660 ha consists of a portion of the Carmichael Coal Mine mining lease. Areas of currently active mining approximately 5,000 ha) have been excluded from the study area.

Approximately 79% of the study area consists of remnant vegetation, comprising 24 Regional Ecosystems (REs) (Neldner *et al.*, 2022). Regional Ecosystems are vegetation communities that are consistently associated with a specific combination of geology, landform and soil in a bioregion (Sattler and Williams, 1999) and these have been mapped for the entire state of Queensland (Neldner *et al.*, 2022). The vegetation is dominated by eucalypt woodlands, particularly *Eucalyptus melanophloia* woodland on plains (RE 10.5.5a) and *Eucalyptus similis* and/or *Corymbia brachycarpa* woodland on sand plains (RE 10.5.1a) that account for 73.5% of the remnant vegetation within the study area. Other habitats include acacia woodlands and shrublands with approximately 10,207 ha of the study area cleared for grazing pastures, roads and mining infrastructure. Darkies Range borders the western extent of the study area with one major watercourse, the Carmichael River traversing the study area in the south, while the smaller North Creek flows through the north-eastern region.



Legend

- Vegetation Survey Site
- Vegetation Survey Site with 1km Transect Undertaken
- BTF Nest
- Watercourse
- Major Road
- Minor Road
- Track
- Project Area

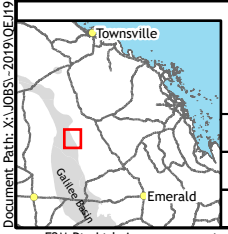
N

Scale 1:250,000 (A4)

0 5 10

Kilometres

Coordinate System: GDA2020



Notes:
Aerial Imagery: © ESRI 2021
Cadastre: © DNRME 2021
Watercourse: © Geoscience Australia 2012
Road: © DNRME 2021

Rev	Description	Drawn	Approved	Date
2	Issued for Use	JY	JMC	30/10/2025
0	Issued for Use	GO	JMC	08/08/2025

FIGURE 4.1-1: VEGETATION SURVEY SITES AND BTF NEST ACTIVE SEARCHES

BTFRP Final Report
Bravus

Map Number	Job Number	Rev
1 of 1	QEJ19152	2

Document Path: X:\JOBS\-2019\QEJ19152\GIS\Final Research Program Report\PRO\QEJ19152\FinalReasearchProgram.aprx\QEJ19152_04_1_VegSurvey_BTF_Nests

E2M Pty Ltd gives no warranty in relation to the data (including accuracy, reliability, completeness or suitability) and accepts no liability for any loss, damage or costs (including consequential damage) relating to any use of the data in this map.

4.1.3.2 Study species

Black-throated finch inhabit eucalypt woodlands consisting of a grassy understorey with sparse shrubs, suitable grass species and bare ground for accessing grass seeds and near reliable water sources (BTFRT, 2007; Higgins and Cowling, 2007; Rechetelo, 2015; Zann, 1976). Remaining black-throated finch populations appear to be limited to the Townsville Plains and the Desert Uplands regions (Buosi *et al.*, 2021). While nests are often located in similar habitats to where black-throated finch forage, little is known of the nesting habitat requirements for black-throated finch in the Desert Uplands (Rechetelo, 2015; Laguna *et al.*, 2019).

Studies of black-throated finch undertaken to date have focused on populations in the Townsville Plain and Mareeba regions in north Queensland (Zann, 1976; Rechetelo, 2015; Laguna *et al.*, 2019; Eco Logical Australia, 2023). However, a subpopulation of black-throated finch in the Desert Uplands Bioregion is considered to be particularly important for the long-term survival of black-throated finch and the ecological requirements of this subpopulation are currently poorly understood . The Desert Uplands and Townsville Plains have distinctly different climates, vegetation communities, habitat resources and threatening processes and therefore a more detailed understanding of their nesting habitat is required to effectively conserve and manage critical habitat for the black-throated finch in the Desert Uplands.

The black-throated finch breeding season is thought to typically occur between October to May with the timing and duration contingent on the availability of food resources (Zann, 1976; Mitchell, 1996b; Higgins and Cowling, 2007). Consequently, breeding varies annually and geographically in response to rainfall patterns, temperature and habitat condition. Breeding, and associated nest site selection, is likely to be triggered by the onset of rainfall during the wet season and when the abundance of grass seed is the greatest (D. F. Mitchell, 1996; Higgins and Cowling, 2007). In Townsville, black-throated finch have been recorded breeding between January to July (D. F. Mitchell, 1996; NRA, 2007) while on the Atherton Tableland they have been recorded breeding from September to January (Bravery, 1970). In the Desert Uplands, breeding and juvenile birds have been recorded between February and September (B. Dreis, pers obs).

Nests in the Townsville Plains have been found in grassy eucalypt woodlands with fewer trees and less coarse woody debris cover compared to surrounding available habitat (Rechetelo, 2015). Several studies in the Townsville area also located the majority of active nests within 400 m of permanent water (Isles, 2007; NRA, 2007; Buosi, 2011a; Rechetelo, 2015). In contrast, active nests have typically been located much further from permanent water (mean ~2 km and maximum of 4 km) in the Desert Uplands (E2M, 2020). Nests are typically dome shaped and constructed from grass stems with a feather or plant down lining (Zann, 1976; Mitchell, 1996b; Higgins and Cowling, 2007). Both male and female birds collect material for nests, usually within close proximity to the nest (Zann, 1976; Mitchell, 1996b). Nests have been recorded in a variety of host tree species including *Melaleuca* spp., *Grevillea* spp., *Acacia* spp. and *Eucalyptus* spp. (Zann, 1976; Rechetelo, 2015; E2M, 2020). The nests are typically built in the upper quarter to half and outer branches of trees and tall shrubs, however, they have also been recorded in tree-hollows, in mistletoe and in the base of raptor nests (BTFRT, 2007; Higgins and Cowling, 2007; Rechetelo, 2015; Zann, 1976). Nests have been recorded at heights between 1.5 m and 12.2 m (Higgins and Cowling, 2007)with no notable preference for the aspect of the nest entrance (Zann, 1976).

Black-throated finch often nest in colonies, with multiple nests constructed in a single tree and / or nests located in neighbouring trees and shrubs, often less than 50 m from other nest trees and up to 250 m apart (Zann, 1976; Higgins and Cowling, 2007; Isles, 2007; NRA, 2007). Black-throated finch have been recorded nesting in the same area in consecutive seasons, within hundreds of metres, though actual nesting location tends to change from year to year (Rechetelo, 2015). Nests are used for breeding and roosting purposes, with birds returning each night to roost. During the non-breeding season, some nests may be used as roost nests housing multiple adult pairs and young (Zann, 1976).

4.1.3.3 Conceptual Model

We developed a multi-scale conceptual model based on vegetation and environmental variables, and their interactions, that influence black-throated finch habitat selection in the Desert Uplands. We considered the following four candidate models to determine the main factors in nesting habitat selection for the black-throated finch.

4.1.3.3.1 Landscape scale model

The landscape model predicts that black-throated finch select nesting habitat based on landscape scale features such as vegetation community, proximity to water, elevation and vegetation growth.

Granivorous birds, such as the black-throated finch, require access to water sources to drink from daily (Zann, 1976; Evans *et al.*, 1985). Therefore reliable water sources are considered to be a key element in black-throated finch habitat and previous studies have suggested that proximity to water is critical in the selection of nest sites (BTFRT, 2007; DEWHA, 2009). Permanent and ephemeral water sources have been mapped throughout the study area and surrounding habitats based on remote imagery and field observations.

The Normalized Difference Vegetation Index (NDVI) is a measure of vegetation greenness and growth, with areas of high NDVI potentially increasing site occupancy by ground foraging birds (White *et al.*, 2022). Based on an assumption of a positive relationship between primary and secondary production in terrestrial habitats, NDVI is often used to predict food availability (Fernández-Tizón *et al.*, 2020). Studies have demonstrated that NDVI can be a successful at predicting the presence and species richness of birds with higher NDVI increasing the probability of occurrence (Debataraja, Pradana and Winarni, 2021; Leveau and Isla, 2021).

4.1.3.3.2 Site scale model

This model predicts that black-throated finch select nest sites based on site-scale vegetation characteristics and resources including vegetation structure and availability of dietary resources. The proximity and abundance of food sources can be an important factor in nest site selection and breeding success (Holway, 1991; Misenhelter and Rotenberry, 2000; Hosseini-Moosavi *et al.*, 2017). Site scale vegetation structure may be important for birds to select nest sites that provide suitable tree characteristics for perches and roosts, constructing nests and shelter from predators (Misenhelter and Rotenberry, 2000). Furthermore, predation risk may be reduced in the nest patch with increasing foliage cover as it inhibits the visual, auditory and chemical cues for predators to locate nests (Martin, 1993).

4.1.3.3.3 Composite model

The composite model includes all landscape and site-scale variables. This model predicts that black-throated finch selects nest sites based on a combination of landscape and site-scale characteristics.

4.1.3.4 Survey design

A total of 220 sites were randomly selected using the Create Spatial Sampling Locations tool in ArcGIS within areas of woodland habitat based on a MaxEnt model developed for black-throated finch habitat suitability within the study area (Dreis, unpublished, 2024). Spatial autocorrelation in the nest presence data was assessed using the ape 5.0 R script (Paradis and Schliep, 2019) to calculate Moran's Similarity Index, which confirmed that the data was spatially independent (Moran's I = 0.06, p = 7.56e-09). Areas of obviously unsuitable black-throated finch habitat, such as cleared pasture and vegetation communities that do not consist of eucalypt dominated woodlands, were excluded from the sampling design.

Seasonal surveys were undertaken over ten periods between March 2020 and June 2023 across the wet and dry seasons to detect any seasonal differences in habitat use and resource availability. Two surveys were conducted during the wet season (January to May) and one during the dry season (July to October) each year, with an additional survey undertaken in the 2020 wet season to trial several survey methods. Surveys consisted of vegetation plots and 1 km transects searching for black-throated finch and their nests.

4.1.3.5 Nest searches

Numerous methods were employed to locate black-throated finch nests and to assess the characteristics of nesting habitat within the study area. Searches for black-throated finch nests were conducted along 1 km transects at 220 sites with a total of 552 transects surveyed over 3 years. Searches for nests were also undertaken during radio-tracking of tagged birds (van Osta *et al.*, 2024b) including following birds back to the nest as well as radio-tracking birds that were roosting during the night. Nests were also recorded incidentally during other research activities including vegetation assessments, while observing foraging black-throated finch and while travelling throughout the study area.

We calculated the number of black-throated finch nests recorded within 500 m of each vegetation site to determine nest presence within proximity to each site.

4.1.3.6 Host tree and nest characteristics

Birds may select nest locations based on the characteristics of potential host trees, such as foliage cover that provides concealment and reduced audio and olfactory cues to reduce risk predation as well as providing protection from rain and direct radiation heat and to regulate temperature (Martin and Roper, 1988; Holway, 1991; Martin, 1993; Endo, 2012; Hosseini-Moosavi *et al.*, 2017; Salema, Gale and Bumrungsri, 2018). For each black-throated finch nest that was located, characteristics of the host tree were recorded including the species, the vegetation stratum of the host species and the crown diameter. The height of the nest above the ground, the dimensions of the nest, the percent visibility and any nest materials were also recorded. The vegetation stratum for habitat utilised by black-throated finch in the study area were defined in accordance with the relevant Regional Ecosystem Technical Descriptions (DES, 2018a) and consisted of a canopy layer (8 to 14 m), a sub-canopy tree layer (4 to 7 m) and a shrub layer (1 to 3 m).

4.1.3.7 Local-scale habitat characteristics

Vegetation surveys were undertaken at each of the 220 sites to characterise black-throated finch habitat throughout the study area. Sites were randomly selected and surveys were undertaken during the wet and dry seasons. Surveys were completed on a 100 x 50 m plot with ten 1 m^2 quadrats along the centre transect and data were recorded using standardised vegetation sampling protocols (Neldner *et al.*, 2022) (Figure 4.1-2).

Within each plot the species composition of the tree, shrub and ground layers, including weed species, was recorded. The height of the tree and shrub layers and the projected foliage cover of weed species was also recorded. The projected foliage cover of the tree and shrub layers was calculated using the crown intercept along the centre 100 m transect. Individual grass species cover as well as the cover of organic litter, bare ground, rock and coarse woody debris were recorded within each quadrat.

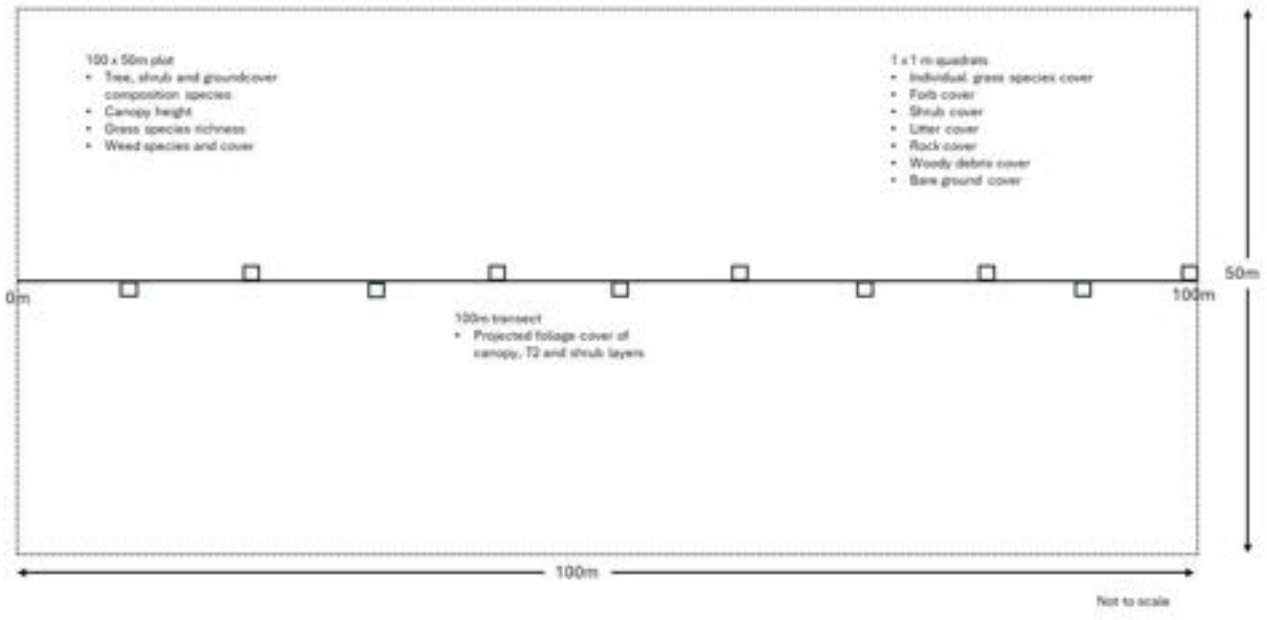


Figure 4.1-2: Layout of 100 x 50 m vegetation plots that were surveyed to characterise black-throated finch habitat throughout the study area.

4.1.3.8 Landscape-scale habitat factors

Landscape-scale characteristics of black-throated finch habitat and nest locations were analysed using GIS, including the vegetation community the nest was located in, NDVI, elevation, ruggedness and the distance of nests to permanent and ephemeral water sources.

Ephemeral water sources were mapped based on field observations and using NDVI from Sentinel II satellite imagery captured during the wet season, which has shown particular use to map ephemeral surface water bodies (Fan *et al.*, 2020). These water sources typically consisted of artificially created shallow depressions along tracks and fence lines, but also included natural wetlands, gilgai and drainage lines that would hold water for up to several weeks.

Table 4.1-2: Summary of environmental variables included in candidate models assessing habitat characteristics influencing nest-site selection by black-throated finch

Environmental variable	Description
Landscape scale	
Regional Ecosystem	Vegetation community as defined by Neldner et al., 2023.
Distance to permanent water	Distance (m) to nearest permanent water source (cattle troughs and large dams)
Distance to ephemeral water	Distance (m) to nearest seasonal water source recorded during wet season field surveys
Normalised Difference Vegetation Index (NDVI)	Measure of vegetation growth and greenness. Calculated from Sentinel II satellite imagery captured from the available data closest to the survey date and each pixel was averaged over each 100 x 100 m cell.

Environmental variable	Description
Ruggedness	Measure of the slope and change in elevation across a terrain surface.
Slope	The steepness of the terrain surface measured in degrees and calculated at each nest location.
Aspect	The bearing that a topographic slope measured in degrees from north calculated at each nest location.
Elevation	The height above sea level measured in metres calculated at each nest location.
Site scale	
Canopy height	The mean height of the ecologically dominant vegetation layer.
Tree richness	The number of tree species recorded within the 100 x 50 m plot.
Subcanopy cover	The foliage projection cover of the mid-storey tree layer measured along a 100 m transect.
Shrub cover	The foliage projection cover of the shrub tree layer measured along a 100 m transect.
Shrub richness	The number of shrub species recorded within the 100 x 50 m plot.
Ground cover	The percentage cover of ground layer vegetation cover measured within ten 1 m2 quadrats along a 100 m transect and averaged for each vegetation survey site.
Grass cover	The percentage cover of grass measured within ten 1 m2 quadrats along a 100 m transect and averaged for each vegetation survey site.
Grass richness	The number of grass species recorded within the 100 x 50 m plot.
Bare ground cover	The percentage cover of bare ground measured within ten 1 m2 quadrats along a 100 m transect and averaged for each vegetation survey site.
Weed cover	The percentage cover of exotic species measured within ten 1 m2 quadrats along a 100 m transect and averaged for each vegetation survey site.
Native grass seed availability	The percentage of seed held by each species measured within ten 1 m2 quadrats along a 100 m transect or 2 x 1m quadrats where a bird was observed foraging and averaged for each vegetation survey site.
Forb cover	The percentage cover of forb species measured within ten 1 m2 quadrats along a 100 m transect and averaged for each vegetation survey site.
Tree scale	
Host species	The tree or shrub species containing a black-throated finch nest.
Vegetation stratum	The vegetation layer (canopy, T2 or shrub) that the nest host tree is located within.
Crown diameter	The diameter of the crown of the tree or shrub species containing a black-throated finch nest.

Environmental variable	Description
Nest height	The height of the nest above the ground.
Nest length	The length of the nest measured from the nest spout.
Nest width	The width of the nest measured perpendicular the length.
Concealment	The averaged percent visibility of the nest from cardinal directions and above and below nest
Nest materials	The vegetation species used to construct a black-throated finch nest, where able to be identified.

4.1.3.9 Exploratory Analysis

Prior to undertaking statistical modelling, habitat variable data were tested for collinearity using Pearson correlation coefficient. Variables with a correlation coefficient > 0.7 were considered to be correlated and were included in models independently to determine their relative importance. Ground cover and grass cover were highly correlated (R = 0.74) so only grass cover was included in the final models as it had a stronger effect. Explanatory variables were verified and were standardised to a mean of 0 and standard deviation of 1 to allow for comparison of model parameter estimates. All exploratory and statistical analyses were conducted using R statistical software (R Core Team, 2023).

4.1.3.10 Analysis of factors influencing nest site selection

To determine which vegetation and landscape characteristics were most important for black-throated finch nest selection, we developed generalised linear models (GLMs) for the relevant candidate models: Model 1: only landscape-scale variables; Model 2: only site-scale variables; Model 3: a composite of landscape and site-scale variables. Models were developed with the 'lme4' R package (Bates *et al.*, 2015) and using the binomial family. A response variable of nest presence or absence for the GLM's where presence was attributed to sites where a nest/s had been recorded within 500 m of the centre of the 100 x 50 plot.

We then implemented the 'drop term' function from the MASS R package (Venables and Ripley, 2002) to create more parsimonious models by excluding habitat variables that did not provide a positive contribution to each model's performance. Consequently we ran a GLM for each of the candidate models using only the variables with a positive delta AIC: Model 4: landscape-scale variables reduced; Model 5: site-scale reduced and Model 6: composite reduced.

We compared the six candidate models using the GLMs and selected the best performing model based on the Akaike Information Criterion (AIC) using 'model select' function in the MuMIn R package (Barton, 2023). The AIC is based on the log-likelihood of the model and the number of explanatory variables in the model (Burnham and Anderson, 2004). Models with a smaller AIC demonstrate a higher model fit, taking into account the number of variables. The variables for the top-ranking models were standardised to a mean of zero and standard deviation of 1 to allow for comparison of the effect size for each variable. We then used the MuMIn R package (Barton, 2023) to perform model-averaging for the top three models to compare the effect size of the model averaged coefficients (effect size and standard error) to determine which habitat variables were providing the greatest influence of nesting habitat selection for the black-throated finch based on the p-value.

4.1.4 Results

4.1.4.1 Black-throated finch nest characteristics

Surveys identified a total of 357 potential finch nests within the study area based on the size, shape and construction material used. Due to the presence of the zebra finch (*Taeniopygia guttata*) in the study area, which constructs a similar nest, only confirmed nests or those with a high confidence of belonging to black-throated finch (due to the presence of nearby activity) were included in the analysis. Consequently, 175 nests were recorded as verified or very likely to be black-throated finch nests within study area and used in the data analysis. The majority of nests (n = 85) were recorded incidentally while doing other survey tasks or travelling through the site. A further 53 nests were recorded during radio-tracking and 37 recorded during active searches and 1 km transect surveys. Nests in the study area were an average of 1.76 km (± 71.7 m) from permanent water and 908.7 m (± 51.0 m) from ephemeral water (Table 4.1-3).

The size of the nests ranged from 10 to 35 cm (21.2 ± 0.5 cm) in length and 7 and 25 cm (13.3 ± 0.3 cm) wide (Table 4.1-3). Nests consisted of an outer layer constructed from the long and stiff stems of common grass species, including *Enneapogon* spp., *Aristida* spp., *Heteropogon contortus* and *Eriachne mucronata* as well as the forb species *Paranotis mitrasacmoides* subsp. *nigricans*, *Evolvulus alsinoides* and *Fimbristylis dichotoma*. Field observations suggest that material for the outer nest layer is sourced from locally common species with black-throated finch observed flying back and forth from the nest to collect material within 10 m of the nest site. An inner layer is constructed from finer and softer inflorescences of grasses such as *Panicum effusum*, *Sporobolus caroli* and *Perotis rara* which may be sourced from further than the direct line of sight to the nest.

Table 4.1-3: Nest and host tree characteristics of black-throated finch nests recorded in the Desert Uplands

Variable	Mean	Median	Minimum	Maximum	SE
Crown diameter (m)	3.36	2.50	0.50	12.00	0.19
Nest visibility (%)	36.60	37.50	0.00	85.00	1.73
Nest height (m)	3.50	3.00	0.25	12.00	0.16
Ephemeral Water Distance (m)	908.68	693.19	44.85	2,818.72	51.01
Permanent Water Distance (m)	1,762.06	1,850.81	168.73	4,133.65	71.70
Nest Width (cm)	13.28	13.00	7.00	25.00	0.25
Nest Length (cm)	21.16	20.00	10.00	35.00	0.51

4.1.4.2 Nest tree characteristics

The majority of black-throated finch nests were recorded in *Eucalyptus melanophloia* woodland on plains (RE 10.5.5a) (n=130; 72.6%) and *Eucalyptus similis* woodland on sand plains (RE 10.5.1a) (n=38; 21.2%). These vegetation communities cover 27,996 ha (46.7%) and 15,488 ha (25.8%) respectively. Ten host species were identified as containing nests, with a strong preference shown for *Acacia sericophylla* (n=102) which accounted for 58% of nests, with *Eucalyptus melanophloia* (n=30), *Melaleuca nervosa* (n=16) and *Bursaria incana* (n=10) were also commonly used (Table 4.1-4). Nests were also located in the hollows of a dead stag (n=1) and a senescent acacia (n=1). The crown diameter of the host tree ranged from 0.5 to 12 m (mean = 3.4 ± 0.2 m).

Table 4.1-4: Host species containing black-throated finch nests in the Desert Uplands

Host species	n	Percentage of nests
Acacia sericophylla	102	58.3
Archidendropis basaltica	2	1.14
Bursaria incana	10	5.71
Carissa lanceolata	2	1.14
Corymbia dallachiana	6	3.43
Corymbia setosa	3	1.71
Eucalyptus brownii	1	0.57
Eucalyptus melanophloia	30	17.1
Melaleuca nervosa	16	9.14
Melaleuca tamariscina	1	0.57
Dead stag	1	0.57
Senescent *Acacia* sp.	1	0.57

The majority of nests were recorded in the subcanopy tree layer (n= 87; 48.6%) and shrub layer (n= 50; 27.9%) at heights ranging from 0.25 to 12 m (mean = 3.5 ± 2.1 m) above the ground (Table 4.1-3). There was a variation recorded in nest height between different species that occur in the same vegetation stratum (Figure 4.1-3). Nest heights varied between the sub-canopy species, *Acacia sericophylla* (2.7 ± 0.1 m) and *Melaleuca nervosa* 4.1 ± 0.5 m), with a similar difference between canopy species *Eucalyptus melanophloia* (6.0 ± 0.5 m) and *Corymbia dallachiana* (6.3 ± 1.1 m).

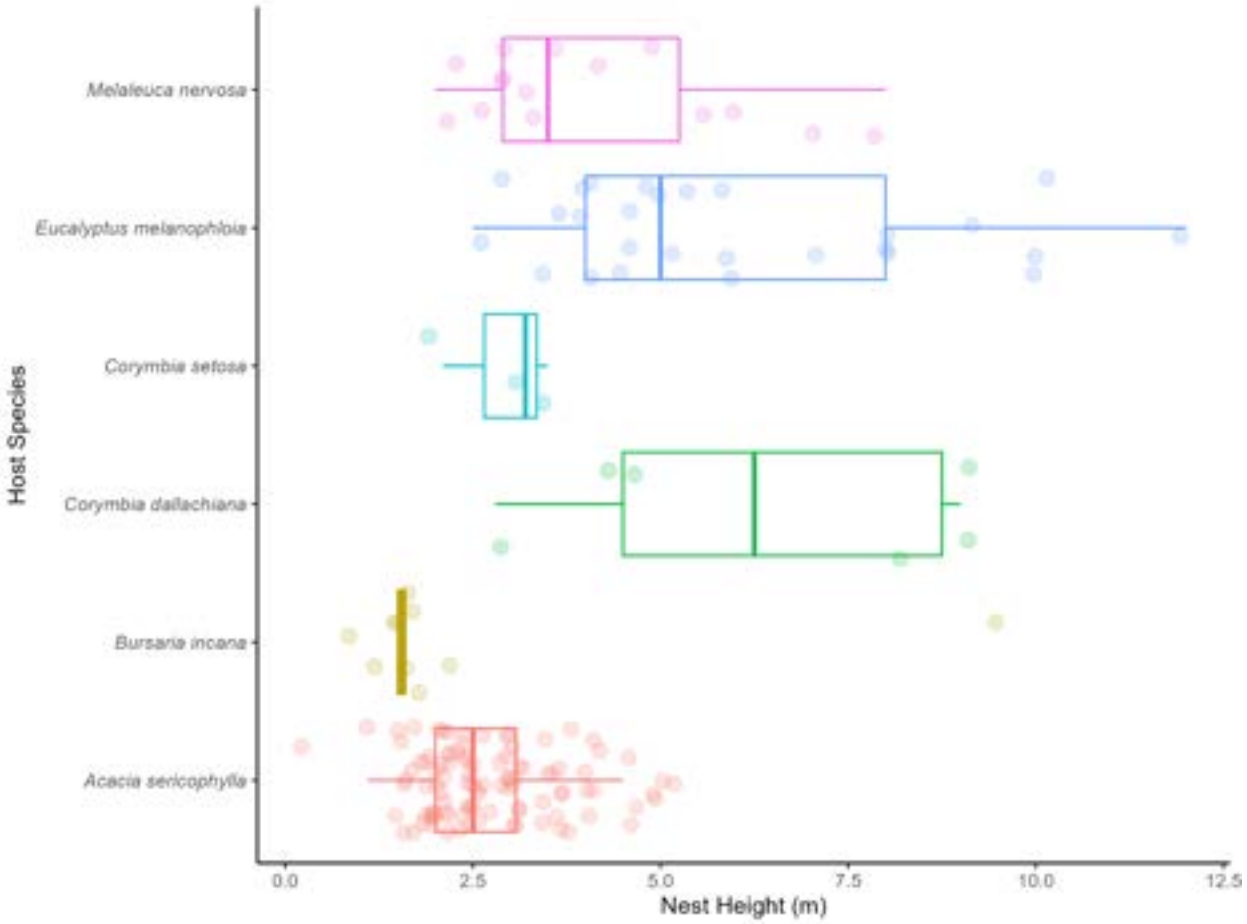


Figure 4.1-3: Box plots of nest heights for the most common black-throated finch nest host species in the Desert Uplands

Nests were generally well concealed amongst the foliage of the host tree, with visibility ranging from 0 to 85% (mean = 36.6 ± 1.7%) (Table 4.1-3). Four nests (2.3%) were recorded within hollow tree limbs and therefore had 0% visibility. Nests occurring in canopy trees tended to be more concealed than those within lower vegetation stratum, with nests in *Eucalyptus melanophloia* (24.6 ± 3.2 %) and *Corymbia dallachiana* (18.3 ± 4.0 %) typically having lower visibility than those in *Acacia sericophylla* (42.4 ± 1.8 %) and *Melaleuca nervosa* (44.2 ± 3.2 %).

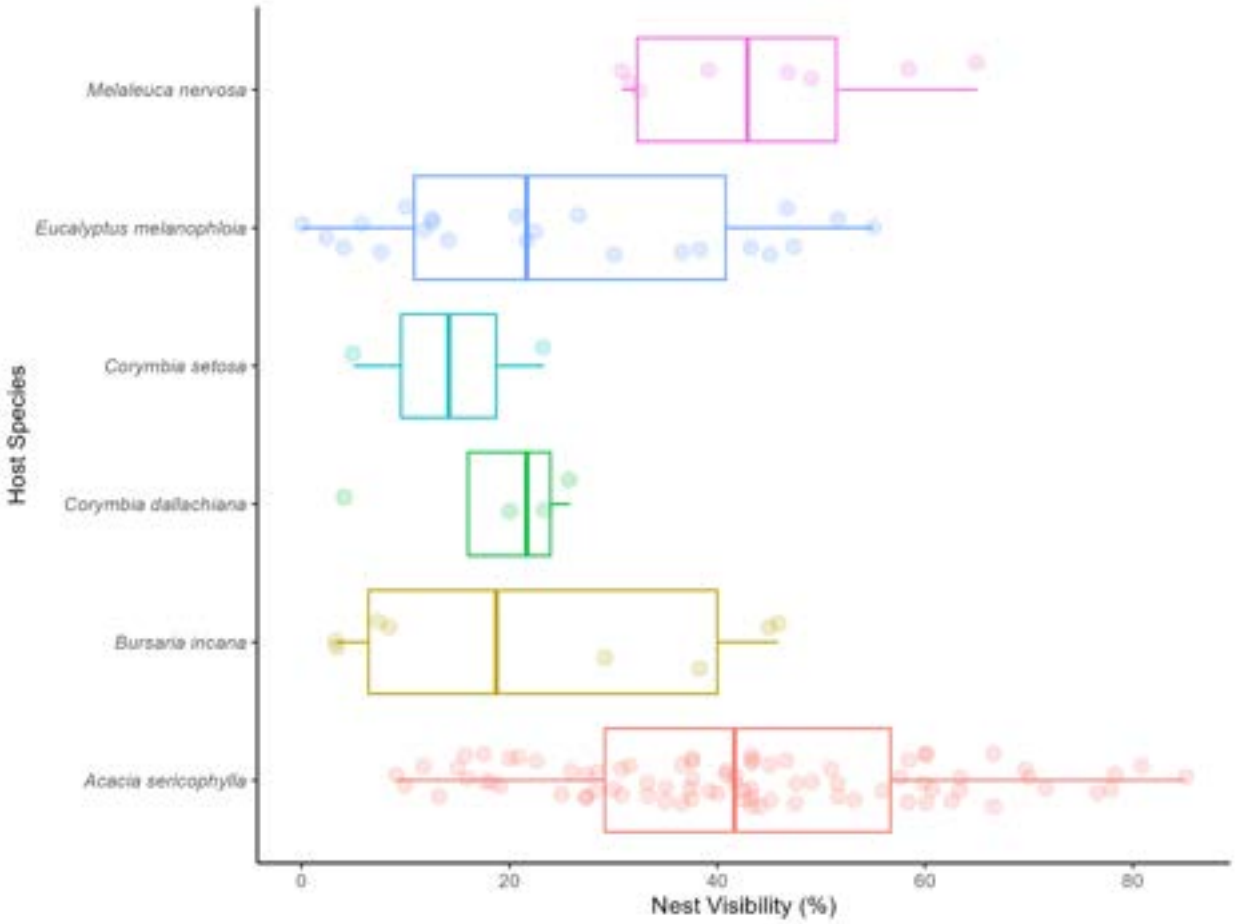


Figure 4.1-4: Visibility of black-throated finch nests in the most commonly recorded host species

4.1.4.3 Site-scale vegetation characteristics

Black-throated finch nest sites in the study area are characterised by a woodland with a sparse canopy cover and very sparse subcanopy and shrub layers (Table 3). Tree and shrub richness is relatively low, although this is generally typical of the vegetation communities that dominate the study area. The groundcover is moderately dense and dominated by grass with moderate cover of bare ground (Table 4.1-5).

Table 4.1-5: Vegetation characteristics of black-throated finch nesting habitat in the Desert Uplands

Variable	mean	median	min	max	SE
Canopy cover (%)	19.33	17.40	0.00	55.00	0.98
Canopy height (m)	10.52	10.50	3.00	15.00	0.21
Tree richness	3.08	3.00	1.00	7.00	0.15
Subcanopy cover (%)	2.52	0.000	0.00	43.50	0.50
Shrub cover (%)	6.48	4.80	0.00	23.20	0.58
Shrub richness	5.79	6.00	0.00	12.00	0.20

Variable	mean	median	min	max	SE
Grass cover (%)	41.34	41.45	0.00	81.70	1.50
Grass richness	8.53	9.00	0.00	18.00	0.28
Forb cover (%)	0.51	0.00	0.00	13.00	0.16
Bare ground cover (%)	23.36	22.50	0.20	65.40	1.06
Weed cover (%)	8.35	5.00	0.00	65.00	1.11

4.1.4.4 Nest site selection models

The results of the model ranking according to the Akaike weights revealed that the reduced composite model (Model 6) was the most parsimonious model for determining black-throated finch nest site selection (Table 4.1-6). This model included five variables: NDVI, canopy height, shrub cover, grass cover and bare ground cover. Proximity to water was a notable omission from the variables in the best performing model. The site scale (reduced) and landscape scale (reduced) models also produced a good fit with respective AIC of 224.9 and 226.6 (Table 4.1-6). site scale (reduced) included five variables: canopy height, shrub cover, shrub richness, grass cover and bare ground cover and the landscape scale (reduced) only included NDVI and elevation.

Table 4.1-6: Results of AIC-based model selection for black-throated finch nest site selection. Number of model parameters (k), maximised log-likelihood values (logLik), AIC values (AIC), AIC differences (delta) and Akaike weights (weight) are shown for each of the models.

Models	K	logLik	AIC	ΔAIC	Weight
Composite (reduced)	5	-105.292	222.6	0.00	0.685
Site scale (reduced)	5	-106.441	224.9	2.30	0.217
Landscape scale (reduced)	2	-110.302	226.6	4.02	0.092
Site scale	12	-103.096	232.2	9.61	0.006
Landscape scale	7	-108.470	238.9	16.36	0.000
Composite	16	-101.689	243.4	20.79	0.000

The GLM indicates that the black-throated finch selects nest sites based on habitat variables at multiple scales. At the landscape scale, NDVI (effect size=5.355; $p<0.05$; SE=2.49) was identified as a significant factor in nest site selection and had the highest effect size of all variables (Figure 4.1-5). The strong positive effect of NDVI in the model indicates that black-throated finch preferentially select areas with higher vegetation productivity and growth to construct nests. At the site scale, areas with lower canopy height (-0.118; $p=0.120$; SE=0.76) and shrub cover (-0.059; $p<0.05$; SE=0.027) as well as more extensive grass cover (0.024; $p=0.059$; SE=0.013) and bare ground cover (0.024; $p<0.05$; SE=0.020) were strong predictors of black-throated finch nest sites (Figure 4.1-5). Although canopy height was not a significant variable, it had the highest effect size of the site scale variables. Shrub cover was a significant indicator of black-throated finch nest site selection, however the effect size was smaller than canopy cover.

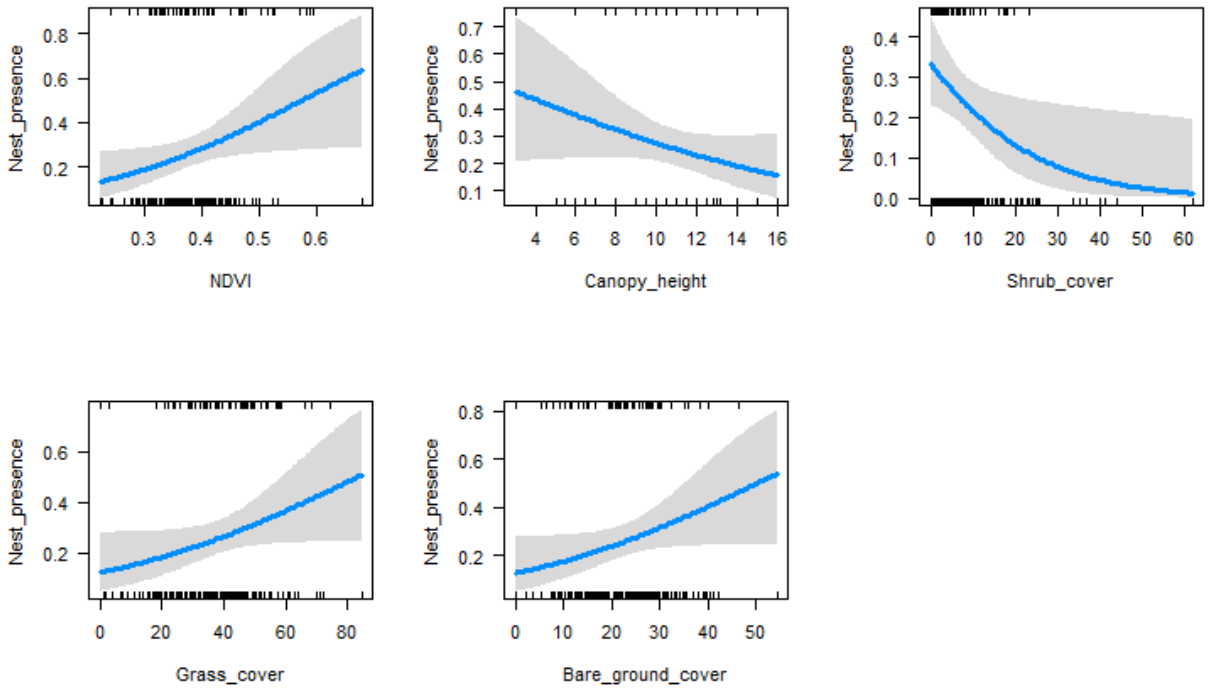


Figure 4.1-5: Predicted probabilities (solid line) with 95% confidence intervals (grey area) of linear mixed model for black-throated finch nest presence with respect to: (a) NDVI; (b) canopy height; (c) shrub cover; (d) grass cover; and (e) bare ground cover.

Because both grass cover and bare ground had a positive relationship with black-throated finch nest site selection, we tested for an interaction effect between both variables. We considered three categories of bare ground, low (9%), moderate (22%) and high (35%) and assessed the effect of grass cover. This demonstrated that there is a consistent positive relationship between grass cover and bare ground across all three categories of bare ground (Figure 4.1-6).

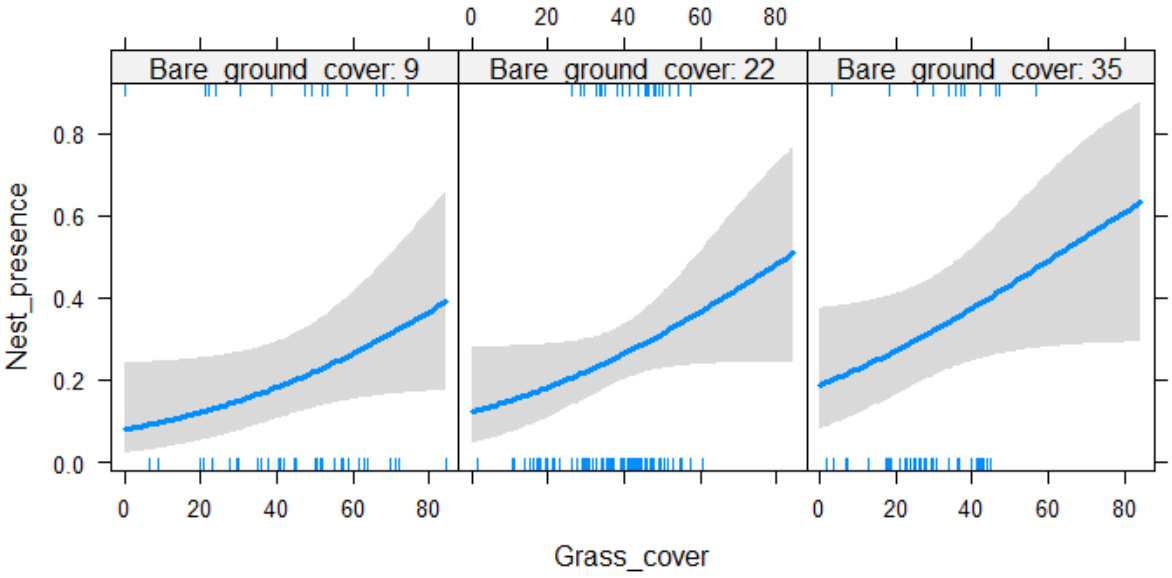


Figure 4.1-6: Predicted probabilities (solid line) with 95% confidence intervals (grey area) of black-throated finch nest presence with respect to grass cover at three different bare ground cover classes: (a) 9% bare ground cover; (b) 22% bare ground cover; and (c) 35% bare ground cover.

4.1.5 Discussion

Our study provides an assessment of the nesting habitat and nest site selection of the Endangered black-throated finch in the Desert Uplands region of central Queensland. Models demonstrated that the selection of nest sites is influenced by habitat factors across multiple scales. Habitat characteristics can vary across the different scales that birds occupy (Steele, 1992) and between areas used for foraging, breeding, or roosting activities (Chase, 2002; Davis, 2005; Nugent *et al.*, 2022). Areas that are rich in food sources may lack the fine-scale resources and habitat features required for breeding and therefore birds may demonstrate partitioning between these habitats (Wiens, 1986; Holway, 1991; Martin, 1995; Misenhelter and Rotenberry, 2000; Hosseini-Moosavi *et al.*, 2017; Remešová *et al.*, 2020; Mansouri *et al.*, 2022; Berlusconi *et al.*, 2024). Accordingly, black-throated finch demonstrated a strong preference to nest within two woodland vegetation communities dominated by *Eucalyptus melanophloia* (RE 10.5.5a) or *Eucalyptus similis* and *Corymbia setosa* (RE 10.5.1a), and also favoured areas with higher NDVI and grass cover as well as habitat with low shrub cover.

Similar to other studies which have identified NDVI as a good predictor of abundance, species richness and habitat quality for birds (Nieto, Flombaum and Garbulsky, 2015; Bonthoux *et al.*, 2018), our models highlighted the importance of NDVI in nest site selection for the black-throated finch. NDVI is an reliable indicator of landscape-level patterns of net primary productivity, including biomass production and grass seed yield (Wang *et al.*, 2005; Flowers *et al.*, 2008) and increased rates of vegetation growth are often associated with increased site occupancy by ground foraging birds due to greater food availability and resources for nest construction (Fernández-Tizón *et al.*, 2020; Debataraja, Pradana and Winarni, 2021; Leveau and Isla, 2021; White *et al.*, 2022). In semi-arid environments with highly variable rainfall such as the Desert Uplands, the availability and quality of resources for food and breeding in areas with higher NDVI are likely to be more pronounced than the broader landscape (Funghi *et al.*, 2020; White *et al.*, 2022).

In contrast to studies in the Townsville Plains, where nests were located close (typically within 400 m) to permanent water (Isles, 2007; NRA, 2007; Buosi, 2011a; Rechetelo, 2015), we found black-throated finch select nest sites in the Desert Uplands to be typically much further from water. This was an unexpected result as permanent water is known to be an important resource for the black-throated finch due to their requirement to drink daily (Zann, 1976; Evans *et al.*, 1985; BTFRT, Department of Environment and Climate Change (NSW) and Queensland Parks and Wildlife Service, 2007; DEWHA, 2009b). This regional difference could potentially be due to the greater concentration and availability of water sources in the Townsville region, or it may indicate that other resources, particularly preferred grasses, for food and nest construction are a higher priority.

While our models identified that black-throated finch select areas of higher vegetation productivity at the landscape scale, nest site selection was also strongly influenced by habitat features at the site scale. Fine-scaled associations with habitat selection has been well documented for numerous finches and granivorous birds (Peacock, Macleod, Davies, *et al.*, 2024) and we found that local vegetation characteristics to be important for black-throated finch nest site selection. Habitat characterised by a low shrub cover was preferred for nest sites, which is consistent with studies from the Townsville Plains (Rechetelo, 2015).

Areas with more extensive grass and bare ground cover are also favoured, when selecting nesting locations in the Desert Uplands. This is in contrast to studies in Townsville that found no difference in ground cover between breeding and non-breeding habitat (Rechetelo, 2015). Areas of more extensive grass cover are likely to provide more food for granivores (Crowley and Garnett, 1999; Moorcroft *et al.*, 2002) and the availability and proximity to foraging resources can be an important consideration in nest site selection for some birds (Barea, 2012; Renton *et al.*, 2015; Segura and Acevedo, 2021). Black-throated finch nests are mainly made from grass material (Zann, 1976; Mitchell, 1996b; Higgins and Cowling, 2007) and therefore areas with more extensive grass cover may be preferable due to the availability of nest construction material. However, adequate inter-tussock spacing and bare ground is important in foraging habitat as it allows the birds to access grass seeds on the ground (Zann, 1976; Mitchell, 1996b; Rechetelo, 2015). The interaction and positive relationship between grass cover and bare ground from our models demonstrates that habitat with a heterogeneous ground layer consisting of a mosaic of bare ground and adequate grass cover is important for black-throated finch when selecting nest sites.

Although previous studies have identified that black-throated finch prefer to forage in areas with a diversity of grass species (D. F. Mitchell, 1996), our models did not indicate that grass species richness was an important factor in the selection of nest sites. Furthermore, grass diversity was found to be negatively associated with nest site selection within the Townsville Plains (Rechetelo, 2015). These results suggest that black-throated finch may have specific habitat requirements for both nesting and foraging.

Black-throated finch in the Desert Uplands exhibit a preference for several host trees when selecting nest sites. These species are substantially different from the host species identified in other black-throated finch populations (Zann, 1976; Rechetelo, 2015), which is likely a reflection of the different vegetation communities that occur between the Desert Uplands, Atherton and Townsville regions. While the majority of nests in our study were identified in small to medium sized trees (e.g. *Acacia sericophylla* and *Eucalyptus melanophloia*) at heights up to 12 m, there was also a large number recorded within shrubs (e.g. *Melaleuca nervosa* and *Bursaria incana*). This differs to what Rechetelo (2015) and Zann (1976) reported previously for the Townsville Plains and Atherton areas, where trees were predominantly used and few nests recorded in shrubs.

Our study identifies the importance of habitat heterogeneity for the black-throated finch across the landscape, including specific resource requirements for breeding habitat that may differ from those used for foraging. Therefore, it is important that conservation management strategies include the identification of areas that support these diverse habitats and resources across the Desert Uplands region. The management of these areas should consider the specific requirements for breeding and foraging when

planning fire regimes and grazing activities (Lunt *et al.*, 2007; Woinarski and Legge, 2013; Lebbink, Dwyer and Fensham, 2021).

Our study identified a similar preference for nesting in habitat with lower shrub cover to the Townsville Plains population. Appropriate fire regimes are therefore likely to be important for maintaining suitable black-throated finch nesting habitat as the absence of fire may lead to thickening of woody vegetation (Murphy *et al.*, 2014; Kirkpatrick and Jenkinson, 2022).

Our study found several differences in the requirements for breeding habitat between populations in the Desert Uplands and the Townsville Plains. This indicates the black-throated finch populations respond to the different resources that are available in each region and emphasizes the importance for specific regional conservation strategies for populations of the black-throated finch.

4.1.6 Conclusion

This study provides a comprehensive understanding of the net site selection of the Endangered black-throated finch in the Desert Uplands bioregion. We demonstrate that black-throated finch select nest sites based on habitat resources and features across multiple scales. At the landscape scale, black-throated finch favours *Eucalyptus melanophloia* woodlands and *Eucalyptus similis* woodlands with higher NDVI and vegetation productivity. These areas are likely to provide a greater abundance of breeding and foraging resources. Habitat consisting of sparse canopy and shrub layers and a ground layer comprising a moderate cover of grass with patches of bare ground were also favoured for nesting.

The study highlights the requirement for the identification of critical habitat and targeted management strategies in these areas that are necessary for the species' survival. Effective conservation strategies should consider the specific habitat requirements of back-throated finch in the Desert Uplands, emphasizing the need to manage habitat to maintain suitable vegetation structure and habitat resources for breeding. Future research should examine predation impacts, temporal variations in nest site selection, and the long-term effects of climate change on the finch's habitat, to facilitate the development of informed conservation and management practices.

Although this study represents the most comprehensive analysis of black-throated finch nest site selection to date, particularly within the Desert Uplands bioregion, further research is necessary to broaden our understanding of the species' ecological needs and enhance the conservation and management of their habitat. Future research on black-throated finch nesting should explore the effects of predation on nest site selection and breeding success. Studies should also investigate temporal patterns in nest site selection, such as seasonal and annual trends, as well as the role of conspecific attraction, to gain a more comprehensive understanding of their nesting behaviour. Genetic research could further reveal insights into breeding success, temporal use of nests and connectivity between different black-throated finch populations in the Desert Uplands region.

4.2 Foraging habitat selection

4.2.1 Abstract

Habitat selection is influenced by a complex relationship of factors across multiple scales, including the availability, quality and distribution of resources. Birds, in particular, are influenced by vegetation structure and composition, which provide essential resources for foraging, breeding and protection from predators. Optimal foraging theory suggests that animals maximise energy gain relative to effort and therefore select habitats and food sources that balance energy intake, accessibility and predation risk. Decreases in resource availability can cause a decline in populations and be a major threat to population viability. Understanding the foraging habitat selection preferences of at-risk species will inform management programs to allow optimal habitat to be protected and restored.

This study examines the foraging habitat selection of the Endangered southern black-throated finch (*Poephila cincta cincta*) within the Desert Uplands region of Queensland, Australia. We investigate the influence of habitat variables across multiple scales to determine the key drivers influencing their selection of foraging habitat. Black-throated finches were observed foraging 447 times across three vegetation communities, primarily in *Eucalyptus melanophloia* woodlands. Generalised linear models identified that preferred foraging habitat consists of areas closer to permanent water with high levels for all of; vegetation productivity (NDVI), native grass richness, bare ground and cover of several grass species; as well as low shrub and weed cover.

The results suggest that black-throated finch select foraging habitats that provide the greatest abundance and accessibility to grass seeds to maximise their energy consumption, while also conserving energy by selecting areas that meet these requirements that occur closer to permanent water. Conservation strategies should aim to manage black-throated finch habitat to enhance the availability of preferred food resources and maintain suitable habitat structure to ensure the long-term survival of black-throated finch populations in the Desert Uplands. Overall, our study demonstrates the importance of managing these habitats to maintain suitable availability and accessibility to native perennial grasses, especially species that are more profitable to consume. These findings highlight the importance of resource availability and accessibility, especially food, and energy trade-offs that granivorous birds such as the black-throated finch must consider when selecting foraging habitats.

4.2.2 Introduction

Habitat selection is influenced by a variety of factors, including the quantity, quality, distribution and combination of resources, which can fluctuate in terms of abundance and availability across space and time (Southwood, 1977; Johnson, 1980; Wiens, 1986). The availability of food, in particular, has significant impacts on the population dynamics and demography of bird species (Sutherland, 2004; Peacock, Macleod, Davies, *et al.*, 2024). Wildlife foraging behaviour is influenced not only by resource availability and habitat quality but also by competition and predation pressures (Wiens, 1986; Southwood, 1988; Block and Brennan, 1993; Kramer, 2001; Abdulwahab, Osinubi and Abalaka, 2019). As a result, how species utilise the landscape is a function of these interacting factors.

Habitat selection may occur at multiple spatial scales, including the broader landscape scale (e.g. extent of vegetation communities, distance to water), or site scale where local vegetation characteristics are important (Johnson, 1980; Morris, 1987; Martin and Roper, 1988; Block and Brennan, 1993). The process of multi-scale habitat selection is influenced by the behaviours and ecological requirements of animals as well as food availability, competition, predation risk and nesting requirements (Mayor *et al.*, 2009b; Mansouri *et al.*, 2022; Berlusconi *et al.*, 2024). Habitat heterogeneity is often important for bird populations through the provision of a diversity of resources, including multiple food sources, nesting sites

and shelter, required for breeding and survival during different life-history stages (Pickett and Siriwardena, 2011; Price *et al.*, 2013; Anderle *et al.*, 2023). Spatiotemporal habitat partitioning may be used by birds in response to food availability, competition with other species and specific foraging and breeding habitat requirements (Kent and Sherry, 2020; Collett *et al.*, 2022). Consequently, birds often spend a significant portion of time in habitats that are unsuitable for breeding due to physical constraints (e.g. offshore waters and tidal mudflats), seasonal availability of resources or nest predation risks (Alerstam and Högstedt, 1982). It is therefore critical for management and conservation strategies to understand the factors driving habitat use for all behavioural requirements across multiple spatial scales.

Vegetation structure and composition have been identified as important factors in the distribution, abundance, and habitat use and selection by birds (Block and Brennan, 1993; Moen and Gutiérrez, 1997; Shackelford and Conner, 1997; Rotenberry and Wiens, 1998; Polak, 2007; Ferger *et al.*, 2014; Nugent *et al.*, 2022). The structural characteristics of vegetation provide birds with important resources such as perches, nest and roost sites, foraging substrates and shelter from predators (Misenhelter and Rotenberry, 2000). Some bird species may also have strong associations and preferences for particular habitat types or certain flora species, which can strongly influenced their distribution (Martin, 1993, 1998; Rotenberry and Wiens, 1998). Numerous studies have therefore investigated these combined effects and multi-scaled habitat features when exploring the relationship between birds and habitat selection (Hall and Mannan, 1999; Mayor *et al.*, 2009b; García, Zamora and Amico, 2011; Nugent *et al.*, 2022). While the vegetation structure and composition can readily affect habitat selection, the distribution and availability of resources (e.g. roosting, nesting and foraging) within the landscape is also important shelter (DeWalt, Maliakal and Denslow, 2003; García, Zamora and Amico, 2011; Ferger *et al.*, 2014). Food availability has been shown to be particularly important in driving bird habitat selection (Korniluk *et al.*, 2021; Londe *et al.*, 2021; Nugent *et al.*, 2022; van Osta *et al.*, 2024b).

Optimal foraging theory predicts that animals seek to maximise their energy gain relative to the effort spent (MacArthur and Pianka, 1966). When food is unevenly distributed in patches, animals must consider travel time between patches, food type and food density in each patch, as well as nutritional intake and predation risk (Cowie, 1977; Benkman, 1991; Grundel, 1992; Alonso *et al.*, 1995; Frens, 2010). The availability and accessibility of food therefore strongly influences the behaviour of animals, including habitat selection and use (Wiens, 1989). Animals can maximise their energy intake by selecting food that provides the most calories or nutrients or by selecting foods they can process the fastest (Díaz, 1990, 1996; Luis Marone *et al.*, 2022).

Foraging strategies at multiple scales may include preferentially selecting food based on availability, accessibility, size and handling time (Sauter *et al.*, 2006). The remaining time may be spent on activities such as avoiding predators, defending territory, or seeking a mate, which represents a time-minimising strategy. In contrast, if an organism has limited time to forage and benefits most from maximising calorie intake, its fitness is enhanced by focusing on energy intake during the foraging period. This is referred to as an energy-maximising strategy (MacArthur and Pianka, 1966; Sauter *et al.*, 2006; Beumer *et al.*, 2020; King and Marshall, 2022). Considerable research has investigated foraging patterns and strategies in birds and how these influence habitat selection (e.g. Abdulwahab et al., 2019; Garabedian et al., 2020; Jones et al., 2006; Kohlmann and Risenhoover, 1996). One group of birds that has received much attention are granivores, where foraging strategies have been tested in natural systems (Garnett and Crowley, 1994; Moorcroft *et al.*, 2002; Milesi, De Casenave and Cueto, 2008; Perea, Venturas and Gil, 2014), but also through the use of experimental trials (Hayslette and Mirarchi, 2002; Sauter *et al.*, 2006; Molokwu, Nilsson and Olsson, 2011; Bedoya-Perez *et al.*, 2013; Camín *et al.*, 2015; Abdulwahab, Osinubi and Abalaka, 2019) to improve our understanding of their foraging habitat selection.

In northern Australian savannas, native annual grasses produce large quantities of dormant seeds during the wet season, providing abundant food resources for granivorous birds (Mott, 1978; Andrew and Mott, 1983; Crowley and Garnett, 1999a; Franklin *et al.*, 2005). These seeds remain the primary food source for

most granivorous birds during the dry season until the onset of summer rain that initiates germination of viable seeds (Andrew and Mott, 1983), creating a decline in seed availability by the early wet season (Woinarski and Tidemann, 1991; Crowley and Garnett, 1999; Dostine and Franklin, 2002). Consequently, birds may need to find alternative food sources or spend more time foraging around this time (Garnett and Crowley, 1994; Dostine and Franklin, 2002). Moreover, granivorous birds in northern Australian savannas have experienced a decline in their populations due to increased grazing pressure from cattle and altered fire regimes, which have significantly impacted ground cover and reduced resource availability (Franklin, 1999; Franklin *et al.*, 2005; Maute, 2011).

One such granivore is the black-throated finch (*Poephila cincta*), which occupies eucalypt woodland savannas in northern Australia and includes two distinct subspecies (Zann, 1976; Mitchell, 1996b; BTFRT, Department of Environment and Climate Change (NSW) and Queensland Parks and Wildlife Service, 2007; Rechetelo *et al.*, 2016b). The southern subspecies, *Poephila cincta cincta*, has undergone extensive declines in both population (potentially >50%) and geographic distribution (approximately 80% reduction) over the past 30 years. As a result, the species is listed as Endangered under Commonwealth and Queensland legislation and is subject to a national recovery plan (BTFRT, Department of Environment and Climate Change (NSW) and Queensland Parks and Wildlife Service, 2007; DEWHA 2009b). Contemporary black-throated finch populations appear to be confined to the Townsville Plains and Desert Uplands regions in Queensland. Anecdotal evidence suggests that the Desert Uplands, and particularly the Alice Tablelands subregion, may host the largest remaining population of black-throated finch (Buosi *et al.*, 2021).

The Desert Uplands Bioregion coincides with the southern extent of the current known range of the black-throated finch and this subpopulation is likely to be particularly important for the long-term survival of the species (Vanderduys *et al.*, 2016b; Reside *et al.*, 2019; DCCEEW, 2023). The semi-arid climate of the Desert Uplands provides distinctly different vegetation and habitat for the black-throated finch compared to the higher rainfall coastal areas of the Townsville Plains. More specifically, the vegetation on the Townsville Plains has been more extensively modified, with a greater proportion of introduced plant species and biomass than in the Desert Uplands region (Mula Laguna *et al.*, 2019).

Habitats occupied by the black-throated finch comprise an open tree layer and an understorey characterised by low vegetation density with a sparse shrub layer, diverse grass species and bare ground to allow access to the seed bank (Zann, 1976; Mitchell, 1996b; Rechetelo, 2015; Rechetelo *et al.*, 2016b; Vanderduys *et al.*, 2016b). Previous studies have indicated that they require a mosaic of different habitats with structural diversity to provide suitable habitat for foraging, roosting and breeding and are rarely observed in highly disturbed or modified habitats, although they have been recorded foraging alongside unsealed roads, beneath power lines and in rail corridors adjacent to woodland habitats (Mitchell, 1996b; NRA, 2007; Rechetelo, 2015; Vanderduys *et al.*, 2016b; E2M, 2022). Reliable water sources within an accessible distance from foraging and breeding habitat are essential for black-throated finch as they need to drink daily (NRA, 2007; van Osta *et al.*, 2024b). During the dry season, they rely on permanent water sources like springs, streams and waterholes or man-made structures such as dams and troughs. In the wet season, they use temporary water sources and shift to new feeding areas. Foraging and nesting habitats must be close to accessible water throughout the year (NRA, 2007; Buosi, 2011a).

Studies of the black-throated finch undertaken to date have focused on populations in the Townsville Plains and Atherton Tablelands regions (Zann, 1976; Rechetelo, 2015; Mula Laguna *et al.*, 2019). On the Townsville Plains the species requires a diversity of grasses available at different times, with accessible seeds on bare ground (Mitchell, 1996a; NRA, 2007; Rechetelo, 2015). Conversely, little is known about their foraging habitat requirements in the Desert Uplands and this knowledge gap has been identified as research priority for the species (Mula Laguna *et al.*, 2019; BTFRP, unpublished). Given the significant differences in the vegetation, climate, availability of water and threatening processes within both bioregions that these two populations inhabit, it is critical to enhance our understanding of their foraging

behaviour and habitat selection across their entire range in order to develop and implement effective management strategies to conserve the species. Therefore, this study aims to identify the factors influencing foraging habitat selection by the black-throated finch in the Desert Uplands Bioregion of Queensland, Australia. By determining the key drivers of foraging habitat selection in the study area, this research will enhance our understanding of the habitat needs of an important regional population. This knowledge will help guide habitat management strategies and inform efforts to preserve and improve black-throated finch habitat throughout its entire distribution.

4.2.3 Methods

4.2.3.1 Study area

The study was undertaken over approximately 75,770 ha centred on 21° 59' S, 146° 21' E and located approximately 150 km north-west of Clermont and 190 km south of Charters Towers (Figure 1), within the Alice Tablelands subregion of the Desert Uplands Bioregion in central Queensland. The Desert Uplands Bioregion coincides with the southern extent of the current known range of the black-throated finch and with the Galilee Basin, where more than 50% of potential black-throated finch habitat occurs under mining leases (Vanderduys *et al.*, 2016b; Reside *et al.*, 2019).

The Desert Uplands occurs within a hot semi-arid Köppen-Geiger climate zone, and the Townsville Plains is within the tropical savanna with a dry winter climate zone (Peel, Finlayson and McMahon, 2007). Vegetation in the Desert Uplands is characterised by eucalypt and acacia woodlands with a spinifex understorey occurring on sandstone ranges and sand plains with low fertility soils, while the Townsville Plains consists of alluvial plains and marine deposits surrounded by sedimentary and igneous hills that support eucalypt and melaleuca woodlands (Sattler and Williams, 1999; Bastin *et al.*, 2009, 2009). Land use in the Desert Uplands is dominated by pastoralism, whereas the Townsville Plains has mixed land use including urban and industrial areas, resulting in more fragmented habitats (Bastin *et al.*, 2009; DES, 2018b). A comparison of major climatic and vegetation characteristics between the Townsville Plains and Desert Uplands regions is presented in Table 4.2-1.

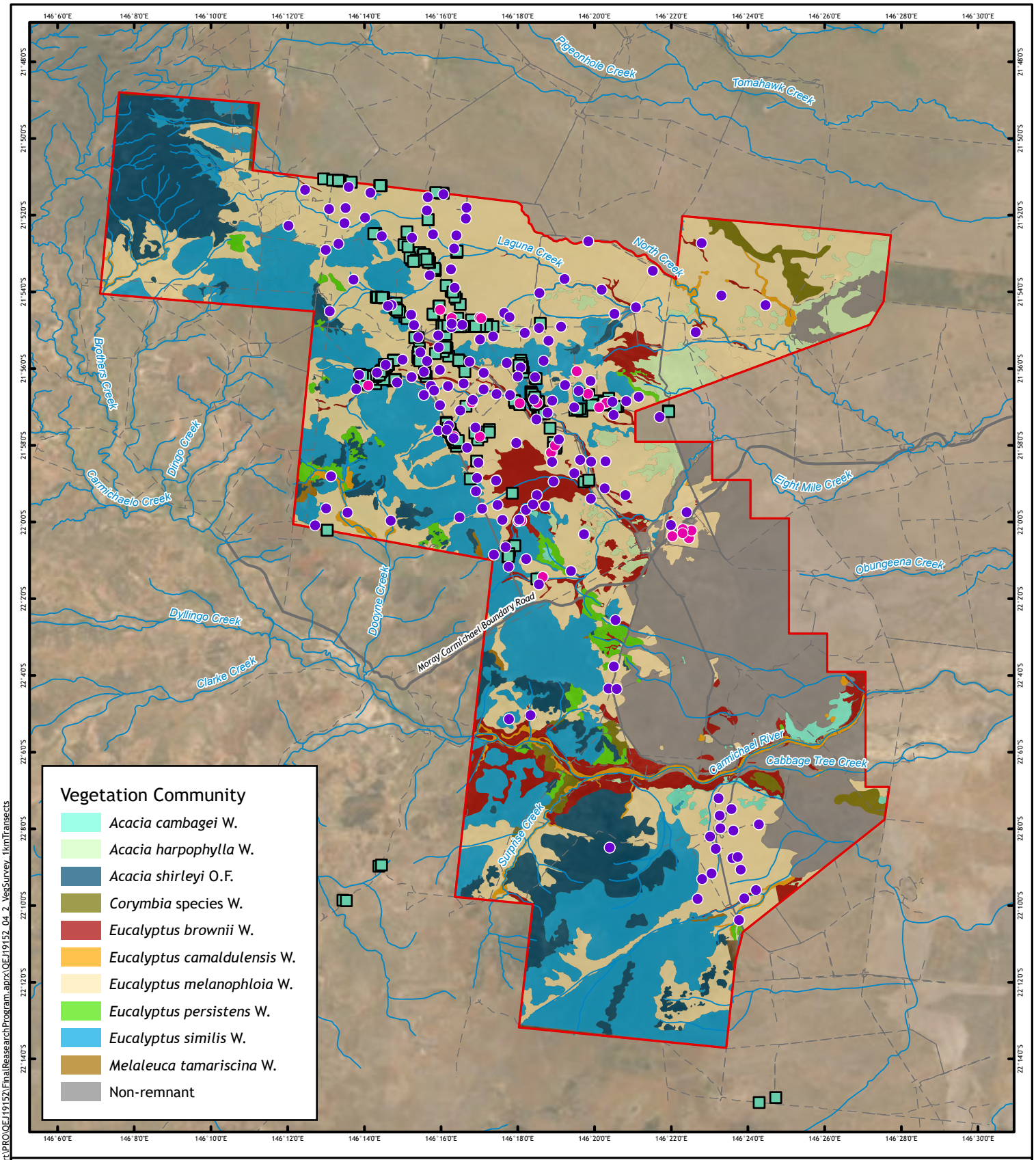
Table 4.2-1: Climatic and vegetation characteristics of the Townsville Plains and Desert Uplands which host the remaining black-throated finch 'stronghold' populations

Environmental characteristic	Townsville Plains	Desert Uplands*
Mean annual rainfall (DES, 2013a)	968.2 mm	588.7 mm
Woody cover (DES, 2018)	60.9%	80.7%
Wetlands as a percent of total subregion area, excluding estuarine wetlands	8.5%	3%

* Values reported for the Alice Tableland subregion of the Desert Uplands, which contains the majority of black-throated finch records and which the study area occurs within (BTFRP, unpublished; Mula Laguna *et al.*, 2019).

Land use within the study area consists of cattle grazing and native vegetation management within a ~33,400 ha environmental offset set aside under the Commonwealth *Environment Protection and Biodiversity Protection Act 1999*. Approximately 33,660 ha consists of a portion of the Carmichael Coal Mine mining lease, and areas of currently active mining (approximately 5,000 ha) have been excluded from the study area.

Approximately 79% of the study area consists of remnant vegetation, comprising 24 vegetation communities which are classified as Regional Ecosystems (REs) by the Queensland Government (Neldner *et al.*, 2022). Vegetation communities in the study region are dominated by eucalypt woodlands, and in particular *Eucalyptus melanophloia* woodland on plains (RE 10.5.5a) and *Eucalyptus similis* and/or *Corymbia brachycarpa* woodland on sand plains (RE 10.5.1), which account for 73.5% of the remnant vegetation within the study area. Other habitats include acacia woodlands and shrublands, with approximately 10, 207 ha of the study area cleared for grazing pastures, roads and mining infrastructure. Darkies Range borders the western extent of the study area, with one major watercourse, the Carmichael River, traversing the study area in the south, while the smaller North Creek flows through the north-eastern region.



Legend

- Vegetation Survey Site
- Vegetation Survey Site with 1km Transect Undertaken
- Foraging Plot
- Watercourse
- Major Road
- Minor Road
- Track
- Project Area

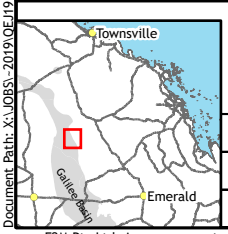
N

Scale 1:250,000 (A4)

0 5 10

Kilometres

Coordinate System: GDA2020



Notes:
Aerial Imagery: © ESRI 2021
Cadastre: © DNRME 2021
Watercourse: © Geoscience Australia 2012
Road: © DNRME 2021

Rev	Description	Drawn	Approved	Date
2	Issued for Use	JY	JMC	30/10/2025
0	Issued for Use	GO	JMC	08/08/2025

FIGURE 4.2-1: VEGETATION SURVEY SITES AND 1KM TRANSECTS

BTFRP Final Report
Bravus

Map Number	Job Number	Rev
1 of 1	QEJ19152	2

Document Path: X:\JOBS\-2019\QEJ19152\GIS\Final Research Program Report\PRO\QEJ19152\FinalReasearchProgram.aprx\QEJ19152_04_2_VegSurvey_1kmTransects

E2M Pty Ltd gives no warranty in relation to the data (including accuracy, reliability, completeness or suitability) and accepts no liability for any loss, damage or costs (including consequential damage) relating to any use of the data in this map.

4.2.3.2 Survey design

A total of 200 sites were randomly selected using the Create Spatial Sampling Locations tool in ArcGIS, within areas of eucalypt woodland habitat based on a MaxEnt model developed for black-throated finch habitat suitability within the study area (B. Dreis, unpublished data). Areas of unsuitable black-throated finch habitat, such as cleared pasture and areas with a low suitability (<0.25) MaxEnt score, were excluded from the sampling design.

Seasonal surveys were undertaken over ten periods between March 2020 and June 2023, across the wet and dry seasons to detect any seasonal differences in habitat use and resource availability. Each year, two surveys were conducted during the wet season (January to May) and one during the dry season (July to October), with an additional survey undertaken in the 2020 wet season to trial several survey methods. Surveys consisted of radio-tracking (van Osta *et al.*, 2024b) and 1 km transects to search for foraging black-throated finch, as well as vegetation plot surveys to assess the vegetation composition and structural characteristics of habitat across the study area.

4.2.3.3 Black-throated finch surveys

Several survey methods were employed to identify black-throated finch and collect foraging data in the study area. Searches for black-throated finch were conducted along 1 km transects at 187 sites in the wet and dry seasons for a total of 552 transects over the 3 years as well as active searches within areas of woodland habitat. Transects were undertaken in 9 REs; however, most were undertaken within 10.5.5a, 10.3.6 and 10.5.1 with 455, 46 and 34 transects, respectively (Table 4.2-2). Manual radio-tracking was also undertaken on 176 geotagged birds for 1,515 detections of birds (van Osta *et al.*, 2024b). We also included incidental observations of black-throated finch that were frequently seen on the side of access tracks while travelling through the study area. Where black-throated finch were observed we recorded, the location, flock size, behaviour (e.g. foraging, drinking, perching, flying, etc.) and other birds species present where they were observed in mixed flocks.

Table 4.2-2: Number of 1 km transects completed within each RE

RE	Number of transects completed
10.3.13a	2
10.3.6	46
10.4.3	1
10.4.5	2
10.5.1a	34
10.5.2	5
10.5.5a	455
10.7.3	3
10.7.3b/10.10.1a	4
Total	**552**

4.2.3.4 Ground cover assessment

Ground cover surveys were completed at each of the 200 vegetation survey sites, as well as 423 additional locations where black-throated finch were observed foraging. Ground cover assessments involved recording the composition and foliage projection cover of each grass species, as well as the cover of forbs, shrubs, leaf litter and bare ground. Vegetation survey sites were surveyed in both the wet and dry seasons. In addition, 361 foraging plots were surveyed during the wet season and 62 in the dry season.

At each vegetation site, ten 1 m^2 quadrats were sampled along a 100 m transect (Figure 4.2-2). Where birds were observed foraging, a 2 m x 1 m quadrat centred on the main area of foraging activity was surveyed, with the larger plot employed to account for multiple birds foraging in the same area as well as individual birds moving around whilst foraging. Despite birds often foraging in flocks, only a single quadrat was surveyed at each foraging location to avoid pseudo-replication.

Vegetation characteristics of foraging plots were compared across the three most commonly foraged vegetation community using the Kruskal–Wallis test. The analysed vegetation characteristics included perennial native grass cover, perennial native grass richness, bare ground cover, shrub cover, Stylosanthes scabra cover and Cenchrus ciliaris cover, as well as distance to permanent water. These characteristics were also compared between foraging plots and systematically collected vegetation survey plots using Mann–Whitney U tests.

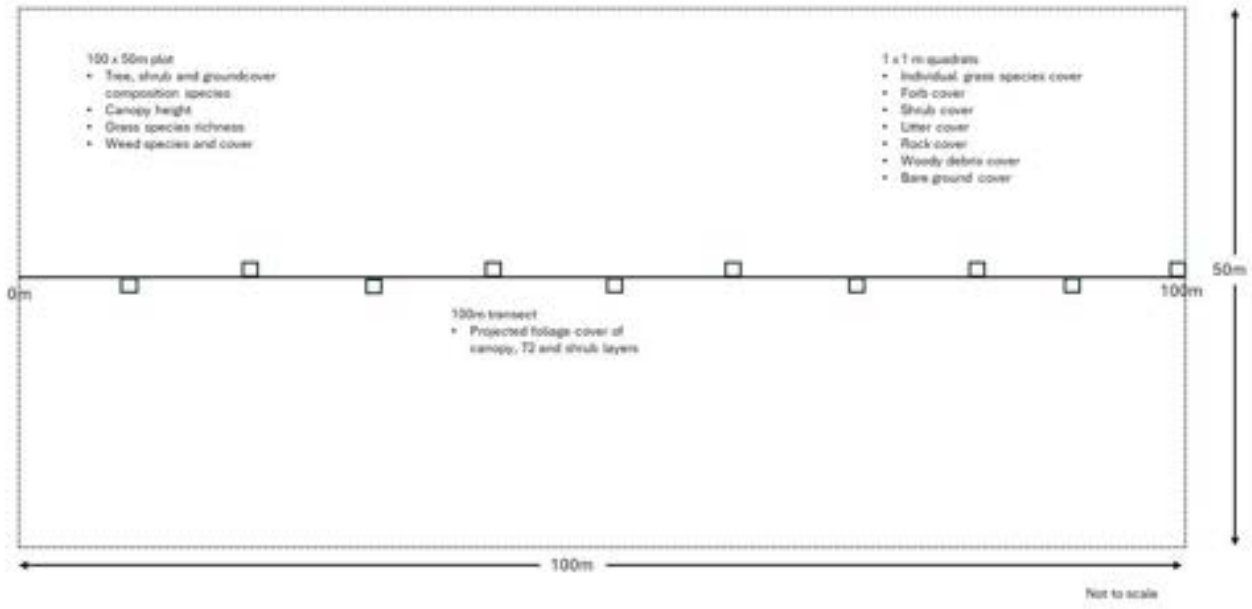


Figure 4.2-2 Layout of 100 x 50 m vegetation plots surveyed at 200 sites in order to characterise black-throated finch habitat throughout the study area.

4.2.3.5 Landscape-scale habitat factors

Landscape-scale characteristics of black-throated finch habitat (Table 4.2-3) were analysed using GIS, including the vegetation community the birds were recorded foraging in, the normalised difference vegetation index (NDVI) and the distance of foraging sites to permanent water sources. Permanent water sources were mapped based on field observations and using NDVI from Sentinel II satellite imagery captured during the dry season.

Table 4.2-3: Summary of landscape and site-scale environmental variables used to assess the habitat characteristics influencing forage-site selection of black-throated finch

Environmental variable	Description
Site scale	
Shrub cover	The foliage projection cover of the shrub tree layer measured within each 2 m^2 foraging quadrat and ten 1 m^2 quadrats along a 100 m transect and averaged for each vegetation survey site.
Native grass cover	The percentage cover of individual grass species was measured within each 2 m^2 foraging quadrat and ten 1 m^2 quadrats along a 100 m transect and averaged for each vegetation survey site. The total cover of native grass was calculated by summing the percentage cover of all individual native grass species.
Native grass richness	The number of grass species recorded within each 2 m^2 foraging quadrat and ten 1 m^2 quadrats along a 100 m transect and averaged for each vegetation survey site.
Bare ground cover	The percentage of bare ground measured within each 2 m^2 foraging quadrat and ten 1 m^2 quadrats along a 100 m transect and averaged for each vegetation survey site.
Weed cover	The percentage cover of exotic species measured within each 2 m2 foraging quadrat and ten 1 m^2 quadrats along a 100 m transect and averaged for each vegetation survey site. Individual cover was also recorded for each of the dominant weed species, Stylosanthes scabra and Cenchrus ciliaris.
Landscape scale	
Vegetation Community	Delineation of Regional Ecosystems as defined by Neldner et al. (2023). Spatial layer.
Distance to permanent water	Distance (m) to nearest permanent water source (cattle troughs and large dams) that reliably provide water year-round.
Normalised Difference Vegetation Index (NDVI)	Measure of vegetation growth and greenness. Calculated from Sentinel II satellite imagery captured from the available data closest to the survey date, with each pixel averaged over each 100 x 100 m cell.

4.2.3.6 Exploratory analysis

Prior to undertaking statistical modelling, habitat variables were tested for collinearity using Pearson correlation coefficient. Variables with a correlation coefficient > 0.7 were considered to be correlated and were included in models independently to determine their relative importance. Ground cover and grass cover were highly correlated (R = 0.74), but only grass cover was included in the final models as it had a stronger effect. All exploratory and statistical analyses were conducted using R statistical software (R Core Team, 2023).

4.2.3.7 Conceptual model

We developed a conceptual model based on resources and vegetation variables at the landscape and site scales, and their interactions, that influence black-throated finch habitat selection in the Desert Uplands.

Due to the large proportion of foraging observations recorded on or adjoining vehicle tracks, we categorised the foraging observation data and included distance to track as a random effect to account for potential sampling bias. Season was also included as a random effect in each model. We used the presence or absence of black-throated finch foraging at each site as the response variable for the models. Sites where birds had been recorded foraging were identified as a presence and vegetation survey sites across where no birds had been recorded were considered a pseudo-absence. We then developed five candidate generalised linear models (GLMs) with a binomial response variable of black-throated finch presence or absence to determine the main factors influencing foraging habitat selection for the black-throated finch.

4.2.3.7.1 Landscape scale models with (Model 1) and without (Model 2) interactions

These models predict that black-throated finch select foraging habitat at a landscape scale by responding to resources such as proximity to water and vegetation growth. Granivorous birds such as the black-throated finch require access to water sources to drink from daily (Zann, 1976; Evans *et al.*, 1985). Therefore, reliable water sources are considered to be a key element in black-throated finch habitat and previous studies have identified that proximity to water is an important consideration in habitat selection (BTFRT, 2007b; DEWHA, 2009a). Permanent water sources have been mapped throughout the study area and surrounding habitats based on remote imagery and field observations.

The NDVI is a measure of vegetation greenness and growth, with areas of high NDVI potentially increasing site occupancy by ground foraging birds (White *et al.*, 2022). Based on an assumption of a positive relationship between primary and secondary production in terrestrial habitats, NDVI is often used to predict food availability (Fernández-Tizón *et al.*, 2020). Studies have demonstrated that NDVI can be a successful at predicting the presence and species richness of birds with higher NDVI increasing the probability of occurrence (Debataraja, Pradana and Winarni, 2021; Leveau and Isla, 2021).

These models assume that black-throated finch may select foraging locations to reduce the costs associated with travelling between foraging and drinking sites. This would allow the birds additional time to spend on finding suitable breeding locations, finding a mate and nest construction to optimise reproductive success.

Model 1 considered the interactions between season and NDVI as well as season and distance to permanent water, whereas Model 2 did not include such interactions.

4.2.3.7.2 Grass cover model (Model 3)

This model predicts that black-throated finch select foraging sites based on the availability of food resources and suitable bare ground to allow birds to access seeds on the ground at the foraging site location. This model assumes that black-throated finch may select foraging patches that contain larger quantities of grass seeds that are more profitable due to a higher calorific and nutritional content and / or have a shorter handling period. This strategy would allow black-throated finch to maximise energy consumption during the available foraging time. This may require an additional investment in time spent searching for patches with higher densities of more profitable food sources.

4.2.3.7.3 Composite models with (Model 4) and without (Model 5) interactions

The composite model includes all landscape and **site scale** variables including the seasonal availability of water. These models predict that black-throated finch selects foraging areas based on a combination of resources at the landscape and site scale. This may include adopting a foraging strategy to maximise their energy consumption by reducing the time invested in searching for patches containing higher densities of profitable food sources. Model 4 also considers interactions between model parameters, specifically

season and NDVI as well as season and distance to permanent water, whereas Model 5 does not include such interactions.

4.2.3.8 Analysis of factors influencing foraging site selection

For each of the candidate models we explored generalised linear models (GLMs) using the binomial family in the 'lme4' R package (Bates *et al.*, 2015) to determine which vegetation and landscape characteristics were most important for black-throated finch foraging habitat selection.

We compared the five candidate GLMs and selected the best-performing model based on the Akaike Information Criterion (AIC) using the 'model select' function in the MuMIn R package (Barton, 2023). The AIC is based on the log-likelihood of the model and the number of explanatory variables in the model (Burnham and Anderson, 2004). Models with a smaller AIC demonstrate a better model fit, taking into account the number of variables. The habitat variables that were providing the strongest contribution on foraging habitat selection for the black-throated finch were determined based on their p-values and effect size.

4.2.4 Results

4.2.4.1 Characteristics of black-throated finch foraging locations

Black-throated finch were observed foraging on 423 occasions and within only three of the 24 vegetation communities occurring in the study area. Most of the foraging observations occurred within *Eucalyptus melanophloia* woodland on plains (RE 10.5.5a) (n=332; 78.5%), with 50 (11.8%) in *Eucalyptus brownii* woodland on alluvial plains (RE 10.3.6) and 41 (9.7%) in *Eucalyptus similis* woodland on sand plains (RE 10.5.1a). These vegetation communities cover 27,996 ha (46.7%), 3,132 ha (5.2%) and 15,488 ha (25.8%), respectively. Of the 552 1 km transects undertaken, 455 (82.43%) were in 10.5.5a, 46 (8.33%) were in 10.3.6 and 34 (6.16%) were in 10.5.1a. However, this survey effort does not include radio tracking or incidental observations during other surveys.

The foraging plots in these vegetation communities differed significantly in terms of perennial native grass cover, perennial native grass richness and bare ground cover. More specifically, foraging plots in *Eucalyptus melanophloia* woodlands had more extensive perennial native grass cover (21.15% ± 0.90%) than both other vegetation communities, including significantly higher cover than *Eucalyptus brownii* woodlands (15.00% ± 2.54%; Dunn test z = -3.59, p = 0.001). Foraging plots in *Eucalyptus melanophloia* woodlands also had higher perennial native grass species richness (3.12 ± 0.09), including significantly higher than *Eucalyptus brownii* woodlands (2.48 ± 0.26; z = -2.83, p = 0.01). In addition, *Eucalyptus melanophloia* woodlands had significantly lower bare ground cover (51.99% ± 1.14%) than *Eucalyptus similis* woodlands (64.32% ± 2.63%; z = 3.87, p < 0.001) (Table 4.2-4).

The mean distance to permanent water of foraging locations was 1,460 m (± 54.1 m) and there was variation between the three vegetation communities, with black-throated finch recorded foraging much closer to permanent water in RE 10.3.6 (mean = 571.0 ± 88.5 m) and *Eucalyptus similis* woodlands (mean = 830.8 ± 115.6 m) compared to *Eucalyptus melanophloia* woodlands (mean = 1,670.3 ± 61.1 m) (Kruskal–Wallis, x^2 = 74.7, p < 0.001; Table 4.2-4).

Table 4.2-4: Comparison of ground and shrub cover characteristics for three vegetation communities in the Desert Uplands where black-throated finch were observed foraging

	Eucalyptus brownii woodlands		*Eucalyptus similis* woodlands		*Eucalyptus melanophloia* woodlands		Kruskal-Wallis test	
Variable	*n*	*Mean (SE)*	*n*	*Mean (SE)*	*n*	*Mean (SE)*	χ^2	*p-value*
Perennial native grass cover (%)	50	15.00 (2.54)	41	16.54 (1.66)	332	21.15 (0.90)	13.5	**0.001**
Perennial native grass richness	50	2.48 (0.26)	41	2.88 (0.22)	332	3.12 (0.09)	8.4	**0.015**
Bare ground cover (%)	50	55.44 (3.05)	41	64.32 (2.63)	332	51.99 (1.14)	15.6	**<0.001**
Shrub cover (%)	50	0.00 (0.00)	41	0.85 (0.60)	332	0.21 (0.07)	2.2	0.339
Stylosanthes scabra p. cover (%)	50	1.86 (0.41)	41	1.88 (0.53)	332	2.23 (0.22)	0.4	0.814
Cenchrus ciliaris cover (%)	50	1.61 (0.60)	41	0.59 (0.25)	332	1.59 (0.31)	0.8	0.677
Distance to permanent water (m)	50	571.22 (88.47)	41	830.84 (115.56)	332	1670.26 (61.13)	74.7	**<0.001**

The ground and shrub cover characteristics of foraging plots also differed significantly from the pseudo-absence sites. Specifically, the cover of perennial native grasses (Mann–Whitney U = 47,790.0, p <0.001), shrubs (U = 52,061.5, p <0.001), *Stylosanthes scabra* (U = 72,565.5, p <0.001) and *Cenchrus ciliaris* (U = 73,537.5, p <0.001) was significantly lower at foraging sites, while perennial native grass richness (U = 150,596.5, p <0.001) and bare ground cover (U = 204,376.0, p <0.001) was significantly more extensive at foraging sites (Table 4.2-5).

Table 4.2-5:Comparison of ground and shrub cover characteristics for black-throated finch foraging sites and pseudo-absence sites in the Desert Uplands

	Foraging (n=423)		*Systematic (n=548)*		Mann-Whitney U	
Variable	*Mean*	*SE*	*Mean*	*SE*	*U-statistics*	*p-value*
Perennial native grass cover (%)	19.98	0.79	39.31	0.79	47,790.0	**<0.001**
Perennial native grass richness	3.02	0.08	2.24	0.03	150,596.5	**<0.001**

	Foraging (n=423)		Systematic (n=548)		Mann-Whitney U	
Variable	*Mean*	*SE*	*Mean*	*SE*	*U-statistics*	*p-value*
Bare ground cover (%)	53.59	1.01	22.20	0.55	204,376.0	**<0.001**
Shrub cover (%)	0.25	0.08	2.78	0.22	52,061.5	**<0.001**
Stylosanthes scabra p. cover (%)	2.15	0.19	4.26	0.23	72,565.5	**<0.001**
Cenchrus ciliaris cover (%)	1.49	0.25	4.31	0.31	73,537.5	**<0.001**
Distance to permanent water (m)	1,458.99	54.09	1,880.45	44.58	85,954.0	**<0.0001**

4.2.4.1.1 Seasonal patterns in foraging locations

There was minimal difference recorded and no significant difference for the distance to permanent water of foraging locations between the wet (mean = 1.47 km ± 60.08 m) and dry (mean = 1.42 km ± 118.38 m) seasons (Wilcoxon W = 11448, p = 0.77).

4.2.4.2 Generalised linear models of black-throated finch foraging habitat

The results of the model ranking according to the Akaike weights revealed that the composite model without interactions (Model 5) was the most parsimonious model for determining black-throated finch foraging habitat selection with an AIC of 341.25, but Model 4 also produced a good fit (AIC 341.58) (Table 4.2-6). Model 5 included 25 variables comprising: NDVI, vegetation community, perennial native grass richness, perennial native grass cover, shrub cover, bare ground cover, litter cover, distance to permanent water and the covers of 15 native grass taxa and two exotic species.

Table 4.2-6: Results of AIC-based model selection for black-throated finch foraging habitat selection using a mixed-effect with the presence or absence of foraging as the response variable. Number of model parameters (k), maximised log-likelihood values (logLik), AIC values (AIC), AIC differences (delta) and Akaike weights (weight) are shown for each of the models.

Models	k	logLik	AIC	ΔAIC	Weight
Model 4	25	-127.623	341.25	0.00	0.54
Model 5	27	-125.792	341.58	0.34	0.46
Model 3	17	-206.259	452.51	111.27	0.000
Model 1	5	-333.843	713.69	372.44	0.000
Model 2	3	-335.859	713.72	372.47	0.000

Model 5 indicates that the black-throated finch selects foraging habitat based on the availability and accessibility of key resources that occur at the landscape and the site scales.

At the landscape scale, NDVI (effect size = 15.85; p<0.0001; SE = 2.61) had the strongest effect on foraging habitat selection (Figure 4.2-3). The strong positive effect of NDVI in the model indicates that black-throated finch preferentially selected areas with higher vegetation productivity and growth in which

to forage. Proximity to permanent water (ES = -0.0006; p<0.005; SE = 0.0002) also had a significant influence on the selection of foraging locations, with birds preferring to forage at locations closer to water, regardless of season (Figure 4.2-4).

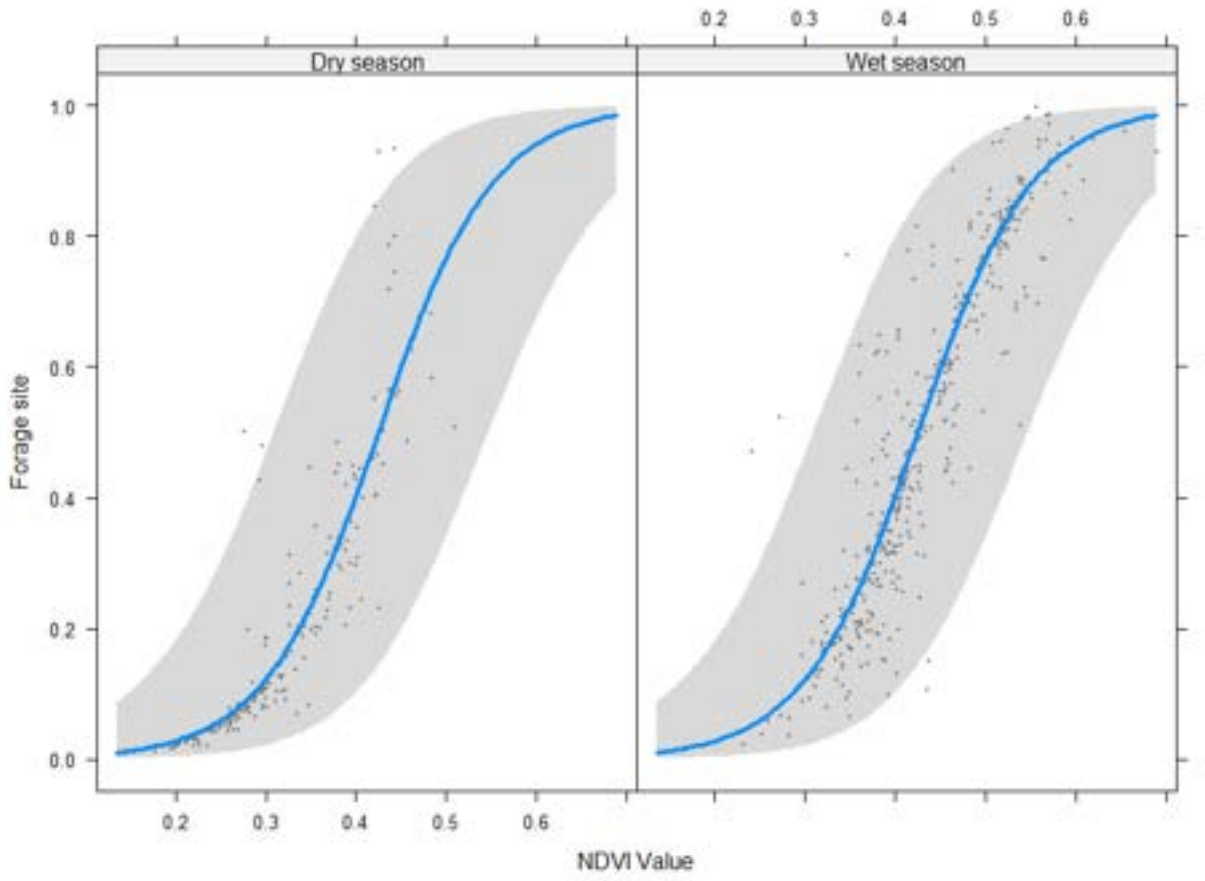


Figure 4.2-3: Predicted probabilities (solid line) with 95% confidence intervals (grey area) of black-throated finch foraging sites with respect to NDVI in: (a) dry season (June to November); and (b) wet season (December to May).

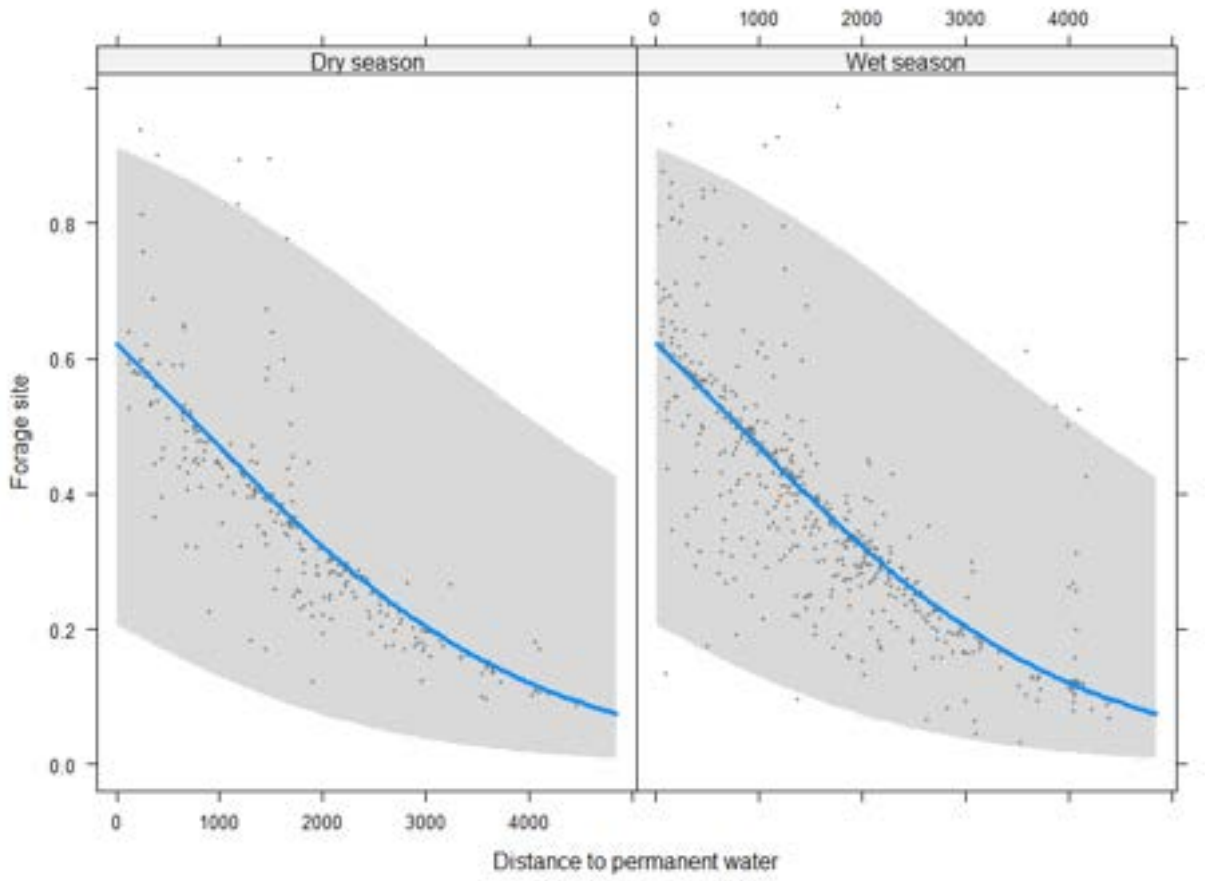


Figure 4.2-4: Predicted probabilities (solid line) with 95% confidence intervals (grey area) of black-throated finch foraging sites with respect to distance to permanent water in: (a) dry season (June to November); and (b) wet season (December to May).

At the site scale, there was a significant relationship with several variables and foraging habitat selection. The model indicates that black-throated finch are more likely to select foraging sites with higher native grass richness (ES = 1.98; $p<0.0001$; SE=0.29) and bare ground (ES = 0.06; $p<0.001$; SE=0.02) as well as lower native grass cover (ES = -0.11; $p<0.05$; SE=0.03) and shrub cover (ES = -0.28; $p<0.0001$; SE=0.07) (Figure 4.2-5).

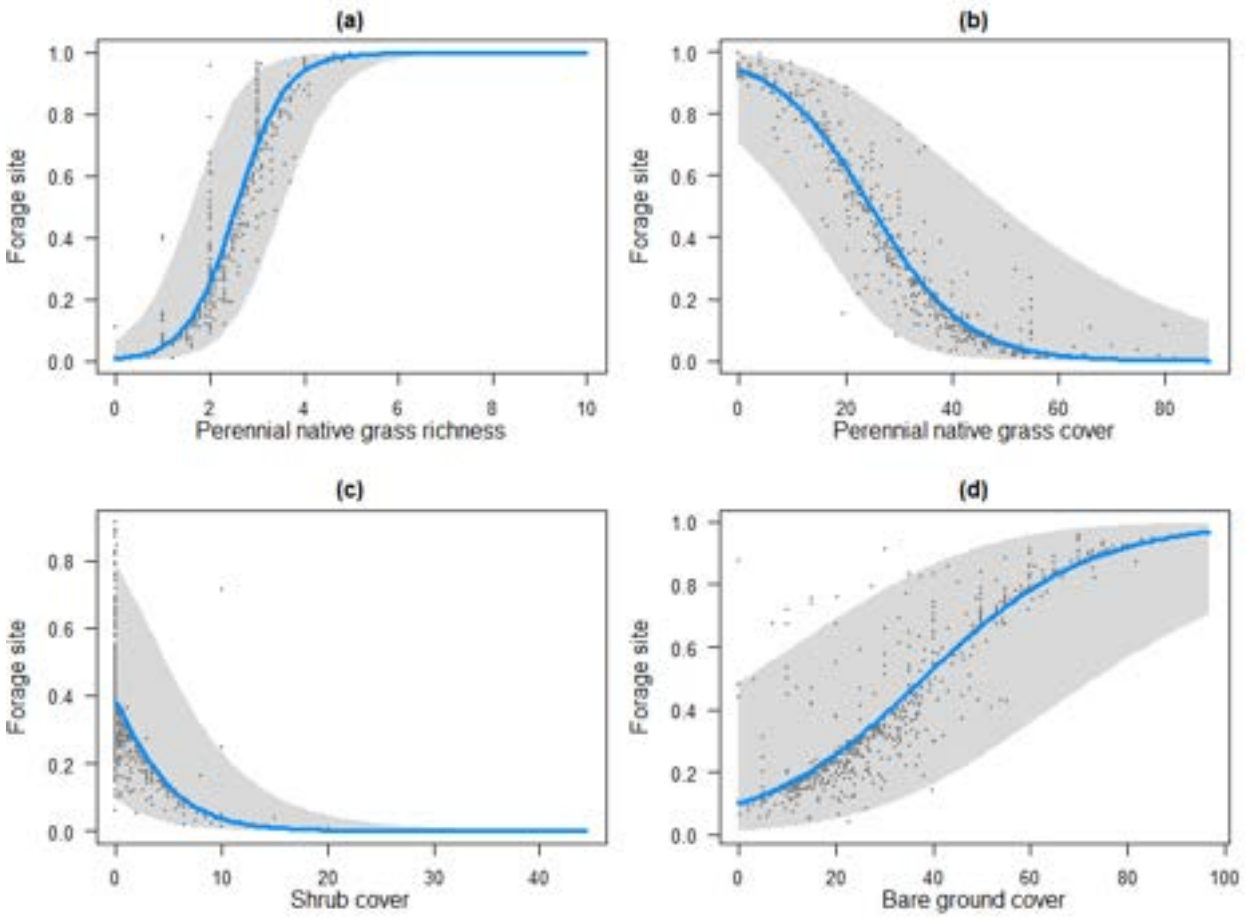


Figure 4.2-5: - Predicted probabilities (solid line) with 95% confidence intervals (grey area) of black-throated finch foraging sites with respect to site-scale variables: (a) native grass richness; (b) native grass cover; (c) shrub cover; and (d) bare ground.

Several native grass species were also identified as strong predictors of black-throated finch foraging habitat selection. The model determined that habitat containing a more extensive cover of *Digitaria ciliaris* (ES = 0.19; p<0.05; SE=0.10), *Panicum effusum* (ES = 0.14; p<0.05; SE=0.06) and *Paspalidium rarum* (ES = 0.72; p<0.05; SE=0.35) are more likely to be selected for foraging. While not significant, there was also a higher probability of black-throated finch foraging in areas with a more extensive cover of other native grasses including *Digitaria brownii, Triodia pungens, Heteropogon contortus* and *Themeda triandra*. However, the model identified a significant negative relationship with the native grass *Eriachne mucronata* (ES = -0.26; p<0.001; SE=0.08) (Figure 4.2-6).

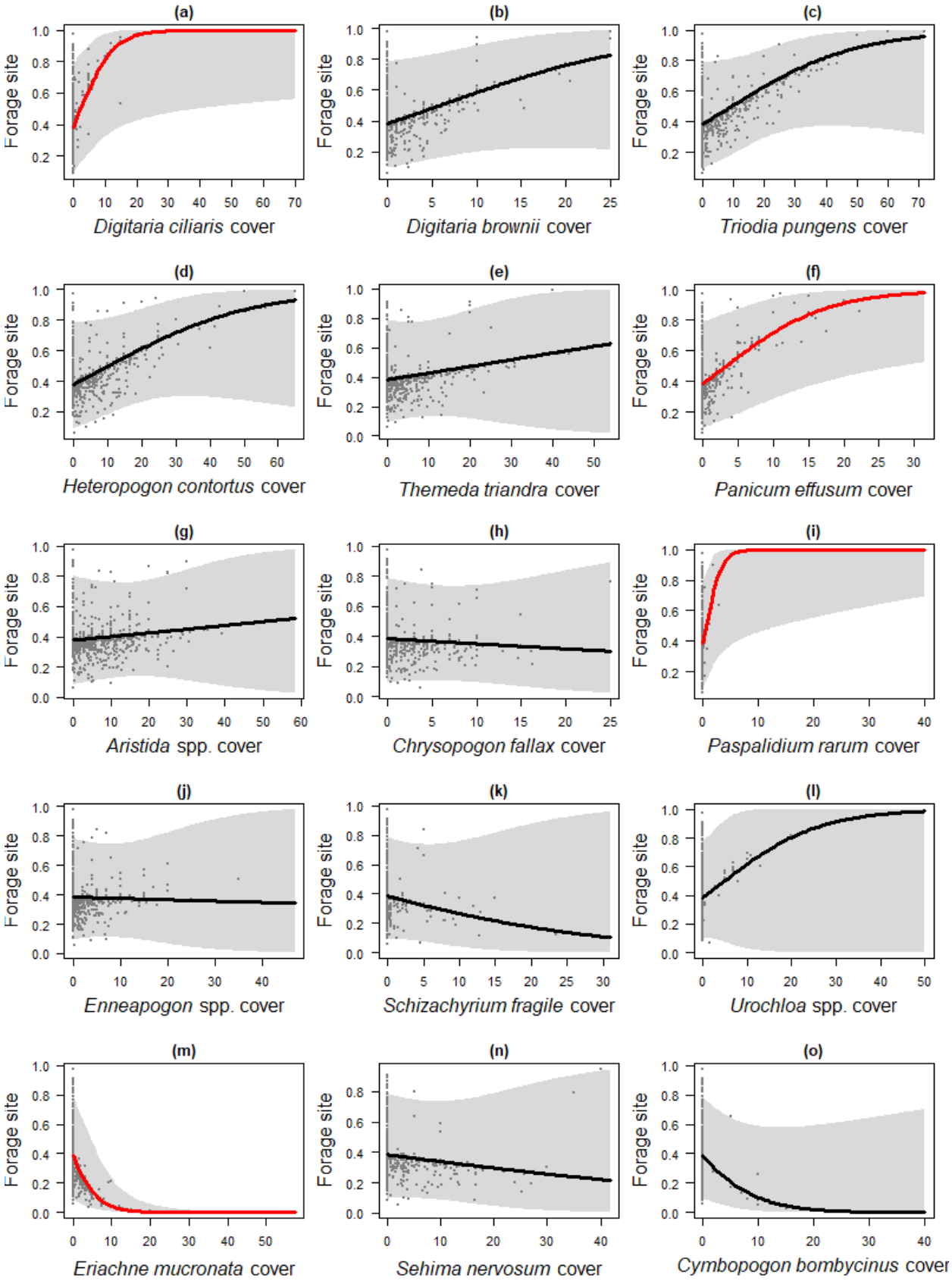

(a)
Forage site
Digitaria ciliaris cover
(b)
Forage site
Digitaria brownii cover
(c)
Forage site
Triodia pungens cover
(d)
Forage site
Heteropogon contortus cover
(e)
Forage site
Themeda triandra cover
(f)
Forage site
Panicum effusum cover
(g)
Forage site
Aristida spp. cover
(h)
Forage site
Chrysopogon fallax cover
(i)
Forage site
Paspalidium rarum cover
(j)
Forage site
Enneapogon spp. cover
(k)
Forage site
Schizachyrium fragile cover
(l)
Forage site
Urochloa spp. cover
(m)
Forage site
Eriachne mucronata cover
(n)
Forage site
Sehima nervosum cover
(o)
Forage site
Cymbopogon bombycinus cover

Figure 4.2-6: Predicted probabilities (solid line) with 95% confidence intervals (grey area) of black-throated finch foraging sites with respect to the cover of native grass species: (a) *Digitaria ciliaris*; (b) *Digitaria brownii*; (c) *Triodia pungens*; (d) *Heteropogon contortus*; (e) *Themeda triandra*; (f) *Panicum effusum*; (g) *Aristida* spp.,; (h) *Chrysopogon fallax*; (i) *Paspalidium rarum*; (j) *Enneapogon* spp.; (k) *Schizachyrium fragile*; (l) *Urochloa* spp.; (m) *Eriachne mucronata*; (n) *Sehima nervosum*; and (o) *Cymbopogon bombycinus*. Red lines indicate species that had a significant effect (p<0.05) in Model 5.

The model identified that black-throated finch are less likely to select foraging habitat with increasing cover of the exotic species *Stylosanthes scabra* cover (ES = -0.22; p<0.0001; SE=0.05) and *Cenchrus ciliaris* cover (ES = -0.08; p<0.05; SE=0.03) (Figure 4.2-7).

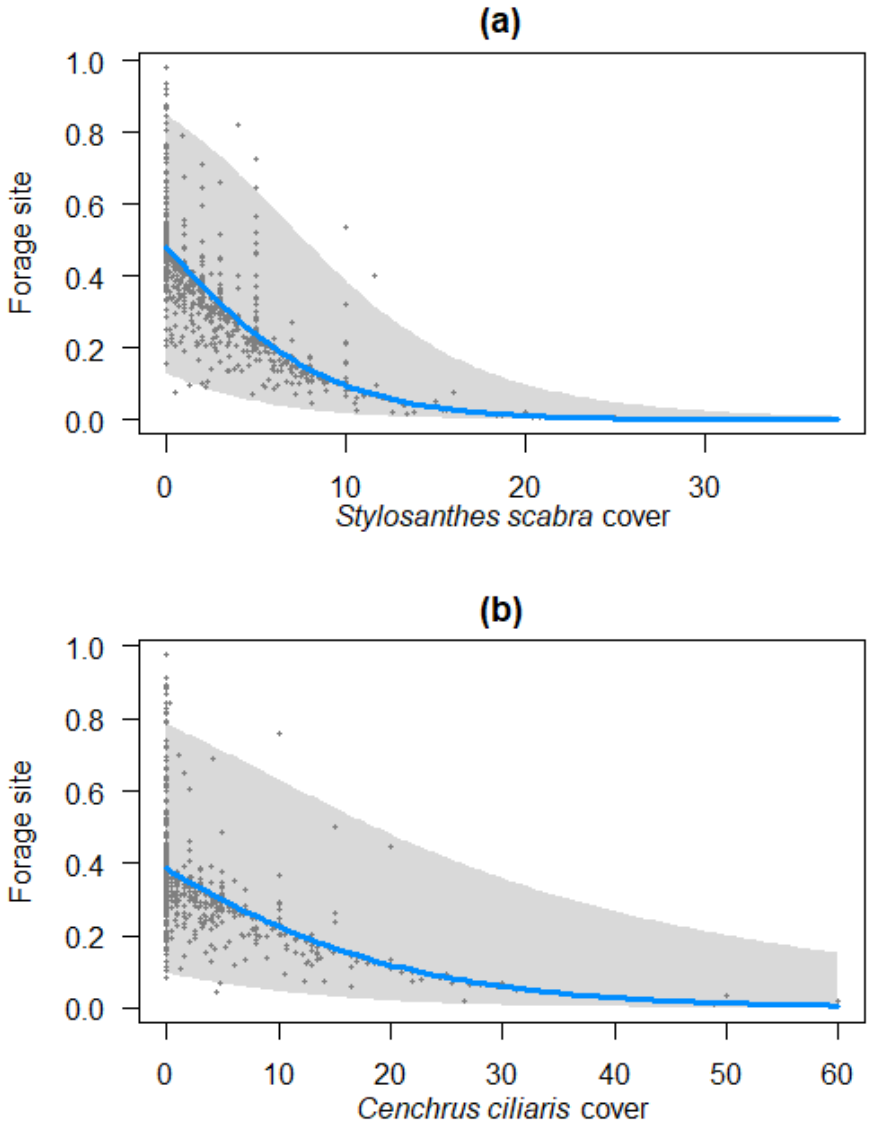


Figure 4.2-7: Predicted probabilities (solid line) with 95% confidence intervals (grey area) of black-throated finch foraging sites with respect to the cover of exotic ground cover species: (a) *Stylosanthes scabra*; and (b) *Cenchrus ciliaris*.

4.2.5 Discussion

Our study found that the selection of foraging habitat by black-throated finch was influenced by a myriad factors at the site and landscape scale, supporting previous studies that highlighted how the availability and distribution of resources influence bird behaviour and habitat selection (Barras *et al.*, 2020; Bliege Bird *et al.*, 2020; Favretto *et al.*, 2024). Birds occupying savanna woodlands in northern Australia are known to be influenced by spatio-temporal habitat heterogeneity (Price et al., 2013). Similar behaviours were observed in the black-throated finch, which showed a strong preference for woodlands dominated by *Eucalyptus melanophloia* (RE 10.5.5a) and *Eucalyptus brownii* (RE 10.3.6), with a lesser preference for *Eucalyptus similis* (RE 10.5.1a), and for areas with higher NDVI. At the site scale, ground cover composition, native grass species richness and structure also had a strong influence on the selection of foraging habitat as these factors are likely to influence the availability of food resources which is known to influence foraging behaviour of granivorous birds (Woinarski and Tidemann, 1991; Garnett and Crowley, 1994; Crowley and Garnett, 1999; Kutt and Woinarski, 2007).

Black-throated finch displayed a strong preference for foraging in habitats with higher NDVI, regardless of season. As NDVI is an indicator of landscape-scale vegetation productivity, reflecting biomass production and grass seed yield (Wang *et al.*, 2005; Flowers *et al.*, 2008), the preference shown by the black-throated finch for foraging in patches higher NDVI reinforce evidence demonstrating that NDVI is a reliable predictor of abundance, species richness and habitat quality for birds (Nieto, Flombaum and Garbulsky, 2015; Bonthoux *et al.*, 2018), but also site occupancy by ground-foraging birds (Fernández-Tizón *et al.*, 2020; Debataraja, Pradana and Winarni, 2021; Leveau and Isla, 2021; White *et al.*, 2022). The greater food abundance provided by areas with higher NDVI is typically more pronounced in semi-arid environments (Funghi *et al.*, 2020; White *et al.*, 2022), and this may be particularly important for birds such as the black-throated finch that occupy these unpredictable landscapes.

Our study indicates that black-throated finch select foraging habitat based on the availability of food to maximise their energy consumption, supporting optimal foraging theory predictions (MacArthur and Pianka, 1966). By favouring habitat containing higher abundances of profitable food sources such as *Digitaria ciliaris*, *Paspalidium rarum* and *Panicum effusum*, the black-throated finch are able to maximise energy consumption as a result of the higher caloric and nutritional value and/or the reduced handling time of these species. Where these resources were abundant, areas characterised by understory vegetation containing a mosaic of native grasses with moderately large areas of bare ground and low shrub cover were preferred.

However, the trade-off between energy consumption and expenditure also influences the selection of foraging habitat for the Desert Uplands population. Black-throated finch must drink water on a regular basis (NRA, 2007; van Osta *et al.*, 2024b), which requires birds to spend energy flying between foraging patches and water sources. Therefore, it is not surprising that proximity to water was determined to have a significant influence on foraging habitat selection, which suggests that black-throated finch aim to reduce the energy invested in travelling between foraging patches and water sources. In semi-arid areas such as the savannas of northern Australia, water can be a limiting resource and its availability across the landscape is therefore commonly an important factor in habitat utilisation, especially for granivorous birds that drink regularly (Pascoe *et al.*, 2021; van Osta *et al.*, 2024b). However, despite the greater availability of ephemeral water across the landscape during the wet season, there was minimal difference in the distance between permanent water sources and black-throated finch foraging habitat between the wet and dry seasons.

Black-throated finch demonstrated a preference for foraging habitats with higher native grass richness, despite the relatively low richness among foraging site generally. Optimal foraging theory predicts that lower value food sources will be gradually added to the diet as the availability of food decreases (MacArthur and Pianka, 1966). This has shown to be the case with the black-throated finch and other

related finches, long-tailed finch (*Poephila acuticauda*) and masked finch (*Poephila personata*), that expand their diet as their preferred grass seeds become scarcer in the dry season (Collett *et al.*, 2022). Therefore, by preferencing habitat with a greater richness of grass species, black-throated finch are likely to have more food sources available during periods of food scarcity such as drought periods and the late dry and early wet seasons.

Conversely, black-throated finch were less likely to forage in areas with more extensive grass cover, as shown by their negative relationship with native grass cover as well as their preference for areas with more bare ground, potentially due to reduced accessibility to fallen seeds on the ground. This supports previous studies that have reported that black-throated finch require sufficient bare ground and inter-tussock spaces to access seeds (NRA, 2007; Rechetelo, 2015). Furthermore, the composition of the grasses in a foraging patch is likely to be more important than grass cover alone as shown by the preference for areas with higher native grass richness. Our model identified a positive relationship between black-throated finch foraging and the cover of several grass species, particularly *Digitaria ciliaris, Panicum effusum* and *Paspalidium rarum,* some of which are well represented in the diet (Section 5.1). The model also demonstrated a negative relationship with the cover of several other native grasses, including *Eriachne mucronata*, *Themeda triandra*, *Aristida* spp. and *Chrysopogon fallax*. *Themeda triandra* and *Aristida* spp. in particular are widespread in tropical biomes (Arthan *et al.*, 2022) and are the locally common and dominant groundcover species in many parts of the study area. Areas dominated by these less preferable grasses may therefore be less desirable for foraging compared to areas with greater abundances of preferred dietary grasses.

Our modelling also revealed that foraging site selection declined as the cover of invasive grasses and shrubs increased. *Cenchrus ciliaris* and *Stylosanthes scabra* are the two most widespread and abundant weed species occurring within the study area and affected forage site selection by the black-throated finch. Exotic weeds are known to impact habitat quality by reducing the recruitment, cover and diversity of native grasses and thus the availability and quantity of grass seeds (Lenz and Facelli, 2005; Tognetti and Chaneton, 2012; Martín-Forés, Guerin and Lowe, 2017). *Cenchrus ciliaris* poses a key threat to granivores that occupy savanna woodlands in northern Australia due to its ability to outcompete native grasses and cause higher intensity and more frequent fires (Eyre *et al.*, 2009; Miller *et al.*, 2010; Fensham *et al.*, 2013; Schlesinger, White and Muldoon, 2013; Ryan-Colton *et al.*, 2024). *Stylosanthes scabra* is also highly invasive and can be difficult to control and eradicate (Gardener, 1993; Orr, 2010). While the black-throated finch displayed a strong preference for habitat containing lower cover of both of these exotic species, they have been observed foraging on *Cenchrus ciliaris*. However, the consumption of this species may be incidental, particularly while foraging on grass seeds on the ground. Areas with more extensive cover of *Cenchrus ciliaris* and *Stylosanthes scabra* are likely to have lower cover of native grass species and bare ground and therefore the availability of preferred grasses and accessibility of fallen grass seeds is likely to be reduced and consequently avoided by black-throated finch.

Shrub cover may also affect the availability and accessibility of grass seeds for granivorous birds such as the black-throated finch. The cover of native perennial grasses has a negative relationship with woody vegetation cover (Daly and Hodgkinson, 1996; Brown and Carter, 1998; Fensham, Fairfax and Archer, 2005) and the absence of fire may result in thickening of the shrub layer in savanna woodlands (Murphy *et al.*, 2014; Eldridge and Soliveres, 2015; Eldridge *et al.*, 2016). We found that black-throated finch preferred to forage in areas with lower shrub cover, which is likely due to the greater availability and accessibility to native grass seeds.

Fire is likely to be an important tool for managing black-throated finch habitat, given its role as a habitat modifier of savanna woodlands. Fire can open up the understory of savanna woodlands to promote lower shrub cover and a patchy ground layer consisting of suitable bare ground to improve accessibility to the soil seed bank (Murphy *et al.*, 2014; Williams *et al.*, 2020; Kirkpatrick and Jenkinson, 2022). Black-throated finches have been observed foraging in recently burnt areas (Brad Dreis and John van Osta, pers.

obs.), and individuals mist-netted near recently burnt sites have shown evidence of charcoal on their plumage (Dean Jones, pers. obs.). These observations suggest that black-throated finches respond rapidly to fire-affected habitats. Similar patterns of preferential foraging in recently burnt areas have been documented in other granivorous bird species (Woinarski, 1990; Fraser *et al.*, 2003; Legge, Garnett and Maute, 2015; Collett *et al.*, 2023).

Our study suggests that black-throated finch adopt a foraging strategy that aims to maximise their energy consumption rather than minimise their foraging time. Studies of their diurnal activity in the Desert Uplands have shown that black-throated finch forage throughout the day, with bimodal peaks in the early morning and late afternoon when the temperature is lower (van Osta *et al.*, 2024b). During the middle of the day, they spend a higher proportion of their time perching and therefore conserving energy (van Osta *et al.*, 2024b). By selecting habitat with greater availability and richness of grasses, especially areas where *Digitaria* spp., *Panicum effusum* and *Paspalidium rarum* seeds are more abundant, and closer to permanent water, black-throated finch are able to optimise their energy consumption.

4.2.6 Conclusion

Our study demonstrates that black-throated finch in the Desert Uplands Bioregion exhibit specific foraging habitat preferences influenced by landscape and site scale factors. Importantly, foraging habitat was characterised by areas of Eucalypt woodland dominated by *Digitaria ciliaris*, *Paspalidium rarum* and *Panicum effusum*, but also where there was high productivity in the landscape. Foraging areas were typically also closer to water regardless of any seasonal variation in rainfall. Site-scale features that were important for foraging habitat selection include grass richness, the availability of preferred dietary species and bare ground, low shrub and exotic plant cover. These results indicate that black-throated finch preferentially select patches with a higher availability and accessibility of grass seeds to maximise net energy consumption. These broad-scale as well as site-specific measures important for black-throated finch foraging are somewhat different to other regions where populations persist, such as the Townsville Plains. Consequently, it is important to conserve and manage these regions with a view to retaining these habitat qualities to ensure the persistence of the southern black-throated finch.

This study highlights the importance of maintaining habitat heterogeneity in suitable woodland habitats. In particular, the structure and composition of the shrub and ground layers is important for maintaining suitable foraging habitat for the black-throated finch. Habitat should be managed to promote low shrub cover with a ground cover that features a mosaic of bare ground and diverse native grass species, to allow access to grass seeds. In particular, these areas should be managed to promote grasses that provide the greatest profit to the black-throated finch in terms of calorific and nutrient content as well as handling time. This may be achieved through the reduction of grazing pressure and the use of fire, however additional research is required to determine the optimal fire regimes for black-throated finch habitat. The control of invasive species, especially *Cenchrus ciliaris* and *Stylosanthes scabra*, is also important for managing black-throated finch habitat. Conservation and management strategies should focus on identifying, protecting and managing areas to provide these habitat features and conditions to ensure the long-term survival of black-throated finch populations in the Desert Uplands.

5 Aim 3: Determine the dietary requirements for SBTF in the Project Area

5.1 Diet of Black-throated finch

5.1.1 Abstract

This study investigates the dietary preferences of the southern subspecies of the black-throated finch (*Poephila cincta cincta*) in the Desert Uplands bioregion of Queensland, examining their diet composition, the nutritional value of grass species and their selection preferences. Observations of birds foraging and DNA analysis of crop and faecal samples from 214 individuals identified 46 plant taxa, including 34 grass species, in the diet of black-throated finch. However, a relatively small subset of these grasses were preferred for consumption due to their higher profitability and reduced handling time. In particular, *Digitaria ciliaris*, *Paspalidium rarum* and *Panicum effusum* were strongly favoured by black-throated finch, especially during the wet season when seeds from these grasses are more available. These species have higher metabolisable energy and crude protein content, and shorter seeds than other grasses that are more abundant in the landscape but are consumed less. Seasonal differences in diet were evident, with more common, but less desirable, grasses being consumed more during the dry season as grass seed availability declines. These findings highlight the importance of maintaining habitats that support the availability of nutritionally valuable grass species to aid conservation efforts for this endangered finch.

5.1.2 Introduction

Preserving or enhancing habitat quality is essential for the survival, reproduction, and overall productivity of bird populations (Block and Brennan, 1993; Johnson, 2007), as the quality and availability of food resources, an integral component of habitat quality, play a crucial role in offspring development and adult fitness (Martin, 1987; Briga *et al.*, 2017; Mishra and Kumar, 2019). Poor nutrition can lead to reduced body size, survival, fecundity, and reproductive success in birds (Metcalfe and Monaghan, 2001; Bennett, Judd and Adams, 2003; Maute, 2011; Weier, Radford, *et al.*, 2017a; Smith *et al.*, 2019). The nutritional quality of food sources can vary across spatial and temporal scales (e.g. Brown & Nocera, 2017; Stone et al., 2019), as well as in response to environmental events such as fire (Bennett et al., 2003; Weier et al., 2017). Moreover, types of food typically occur at varying abundances, and selective feeding may occur where co-occurring food sources are consumed at different rates (Jacobs, 1974). A comprehensive understanding of the behavioural ecology of vulnerable and threatened bird species is thus critical to their conservation and requires the collection of both observational data about where birds feed and information on the broader availability of their preferred food resources (Sutherland, 2004).

Optimal foraging theory predicts that animals seek to maximise their energy gain relative to the effort spent (MacArthur and Pianka, 1966). When food is unevenly distributed in patches, animals must consider travel time, food type and food density in each patch, as well as nutritional intake and predation risk (Cowie, 1977; Benkman, 1991; Alonso *et al.*, 1995; Sauter *et al.*, 2006; King and Marshall, 2022). The availability, accessibility and profitability of food therefore strongly influences the behaviour of animals who can maximise their energy intake by selecting food items that provide the most calories or nutrients or foods they can process the fastest (Wiens, 1989; Díaz, 1990, 1996; Luis Marone *et al.*, 2022).

In the most basic form of optimal foraging theory, organisms can either minimise the time spent foraging or maximise energy intake (MacArthur and Pianka, 1966). If an individual only needs a certain minimum number of calories and gains no advantage from consuming more, its fitness will be maximised by reducing the time spent acquiring food and consuming as many calories as possible during the available foraging

time (Pyke, 1984; King and Marshall, 2022). Strategies for efficient foraging therefore must consider the trade-off between the size, handling time and availability of any food resource (Werner and Hall, 1974; Grundel, 1992; Sauter *et al.*, 2006). These more profitable food sources are preferred as they allow the remaining time to be used for other activities including nest construction, avoiding predators or defending territory.

For granivorous birds, diet preferences are a function of the interaction between the profitability of the food source (i.e. the calories or energy provided), its availability throughout a patch and its handling efficiency, based on the physical properties of the seed, including its size (Schluter, 1982; Kohlmann and Risenhoover, 1996; Molokwu, Nilsson and Olsson, 2011; Geiger *et al.*, 2014; Titulaer *et al.*, 2018). The quality and availability of food resources may influence the foraging behaviours of granivorous birds (Schluter, 1982). When food is plentiful, birds may spend less time foraging. However, long-tailed finches (*Poephila acuticauda)* and masked finches (*Poephila personata*) have been observed feeding at rates that would necessitate approximately six hours of foraging per day during times of food scarcity (Garnett and Crowley, 1994). Studies of Darwin's finches (*Geospiza* spp.) found that birds selected seeds with low energy returns where they occurred in high density and therefore reducing the need to search for more profitable food sources (Schluter, 1982).

A time-minimisation strategy relevant to granivorous birds is to select seeds that align with their morphological adaptations so as to minimise handling and processing times (Willson and Harmeson, 1973; Keating *et al.*, 1992; Nagy Koves Hrabar and Perrin, 2002; Carrillo *et al.*, 2007; Titulaer *et al.*, 2018). Birds with smaller bills may process small seeds faster than larger-billed birds due to their increased dexterity in positioning the seeds for dehusking or cracking (Díaz, 1990), while birds with larger bills can handle larger seeds more efficiently (Meij, Griekspoor, and Bout, 2004). Consequently, certain bill sizes and morphologies optimise the processing speed for specific seeds (Schluter, 1982; Grant, 1986; Diaz, 1990, 1996), and selecting seeds that are optimal for an animal's morphology can help minimise the time spent foraging.

While seed selection based on morphological characteristics and handling time has been shown to be an important strategy for granivorous birds (Molokwu, Nilsson and Olsson, 2011), research shows that the nutrient content of seeds also plays a role in diet choice (Díaz, 1996; Schaefer, Schmidt and Bairlein, 2003; Valera *et al.*, 2005; Carrillo *et al.*, 2007; Molokwu, Nilsson and Olsson, 2011; Ríos, Mangione and Marone, 2012; Weier, Radford, *et al.*, 2017a). Grass seeds provide the energy required by granivorous birds through various nutritional components including carbohydrates, protein and other compounds (GLÜCK, 1985; Díaz, 1996; Carrillo *et al.*, 2007; Brzęk *et al.*, 2010; Ríos, Mangione and Marone, 2012). Protein also contributes to body mass, reproductivity and stress and immune responses in adult birds (Pryke *et al.*, 2012), and seed nutritional content is particularly important for breeding individuals as protein, mineral such as calcium and fatty acids are all critical for egg development and hatchling growth (Klasing, 1999).

Granivorous birds inhabiting savannas in northern Australian encounter seasonal fluctuations in the availability of food sources (Woinarski and Tidemann, 1991, 1991; Dostine and Franklin, 2002). Native annual grasses produce large quantities of dormant seeds during the wet season that provide abundant food resources for granivorous birds (Mott, 1978; Andrew and Mott, 1983; Crowley and Garnett, 1999b; Franklin *et al.*, 2005). The availability of grass seeds steadily declines over the dry season due to the consumption by granivores and fire (Woinarski and Tidemann, 1991) and may be limiting in some areas resulting in increased stress levels and lower body condition of granivorous birds (Maute, 2011). The majority of viable grass seeds germinate soon after the onset of rainfall during warmer temperatures (Andrew and Mott, 1983), leading to a rapid decline in the availability of grass seeds in the early wet season (Woinarski and Tidemann, 1991; Crowley and Garnett, 1999; Dostine and Franklin, 2002). This may require birds to consume other food sources and spend prolonged periods of time foraging (Garnett and

Crowley, 1994; Dostine and Franklin, 2002). It is apparent that dietary studies need to account for these seasonal fluctuations to fully understand avian granivore requirements (Franklin *et al.*, 2005).

Seasonal changes in seed production and the availability of grass seeds are also affected by landscape scale ecological processes and land management such as fire and grazing (Bennett, Judd and Adams, 2003; Franklin *et al.*, 2005; Maute, 2011; Weier, Radford, *et al.*, 2017a). Extensive grazing and altered fire regimes in these habitats have significantly impacted ground cover, exacerbated the reduction in resource availability, contributing to the decline of numerous granivorous birds in northern Australia (Franklin, 1999; Franklin *et al.*, 2005; Maute, 2011), including the southern subspecies of black-throated finch (*Poephila cincta cincta*). The black-throated finch (southern) population has experienced a large reduction in range and abundance over the last 30 years, resulting in an Endangered listing and development of a national recovery plan (BTFRT, Department of Environment and Climate Change (NSW) and Queensland Parks and Wildlife Service, 2007; DAWE, 2009a). The decline of the black-throated finch is attributed to a combination of habitat clearing and fragmentation as well as changes in land use, particularly pastoralism and agriculture that have rendered other extant black-throated finch habitat unsuitable due to the introduction of exotic pasture grasses and altered fire regimes (DAWE, 2009a; Vanderduys *et al.*, 2016b; Mula Laguna *et al.*, 2019; Reside *et al.*, 2019b). The black-throated finch is now thought to be confined to fragmented populations in the Brigalow Belt North and Desert Uplands bioregions of Queensland (Commonwealth of Australia, 2023). Given the significant alteration of this species' foraging habitat, it is essential to enhance our understanding of their foraging behaviour and dietary preferences, including seasonal variations, to effectively inform conservation and management strategies.

Studies of black throated finch diet have focused on the Townsville Plains population (Mitchell, 1996; Rechetelo *et al.*, 2016a; Mula Laguna *et al.*, 2019; Williams *et al.*, 2020) and Atherton Tablelands (Zann, 1976), with limited knowledge of their diet in the Desert Uplands (Mula Laguna *et al.*, 2019; Commonwealth of Australia, 2023). The seeds of a variety of grasses and legumes have been recorded as consumed by black-throated finch, with birds selectively targeted different species across seasons (Zann, 1976; Mitchell, 1996a; Isles, 2007). Previous studies have also been largely limited to incidental observational records of birds foraging (Mula Laguna *et al.*, 2019; Williams *et al.*, 2020), particularly during the wet season when birds can be observed taking seed directly from grass inflorescences (Zann, 1976; Mitchell, 1996a). However, black throated finch spend much of their time foraging directly from the soil surface where they eat fallen and half-ripe grass seeds (Zann, 1976; Mitchell, 1996a; Rechetelo, 2015), making it difficult to determine what species of grass seeds they are consuming. In the Desert Uplands region, which has a semi-arid climate characterised by variable annual rainfall and distinct wet (December to May) and dry seasons (June to November) (Sattler & Williams, 1999), the availability of native grass seeds fluctuates significantly between seasons. Black-throated finch are therefore likely to adjust their diet throughout the year in response to these seasonal changes in seed availability, similar to other granivorous birds in northern Australia (Garnett and Crowley, 1994; Dostine and Franklin, 2002).

Although field observations suggest black-throated finches are selective foragers (Mitchell, 1996), a study of captive birds found them to be the least selective among four finch species, including the northern subspecies (*Poephila cincta atropygialis*) (Laguna, 2020). Captive black-throated finches showed a similar preference for three out of four commercial seed species, unlike other finches that preferred only two. While grass seeds are their primary source of food, black-throated finch have also been recorded eating invertebrates (mainly insects and their larvae), particularly during the breeding season (Higgins and Cowling, 2007; Rechetelo, 2015), as well as *Eucalyptus melanophloia* seeds (Meyer and Agnew, 2012).

Studies from Townsville are unlikely to be meaningful for understanding the foraging ecology of black-throated finch in the Desert Uplands due to differences in habitats, vegetation composition, climate and the availability of water between the two regions. Specifically, the Desert Uplands has a semi-arid climate with predominantly low fertility soils with poor structure formed on sand plains and sandstone ranges in contrast to the more fertile soils and higher rainfall of the Brigalow Belt, which the Townsville Plains is a

province of, to the east (Bastin et al., 2009; Sattler, 1999). The Alice Tablelands province of the Desert Uplands is dominated by vegetation communities characterised by *Eucalyptus similis, Acacia shirleyi, E. melanophloia* and *E. brownii* and contains extensive salt lakes and fringing dunes, whereby the Townsville Plains consists of *E. platyphylla, E. crebra* and *Corymbia dallachyana* dominated woodlands as well as salt marsh, beach scrub and mangrove communities (Sattler, 1999).

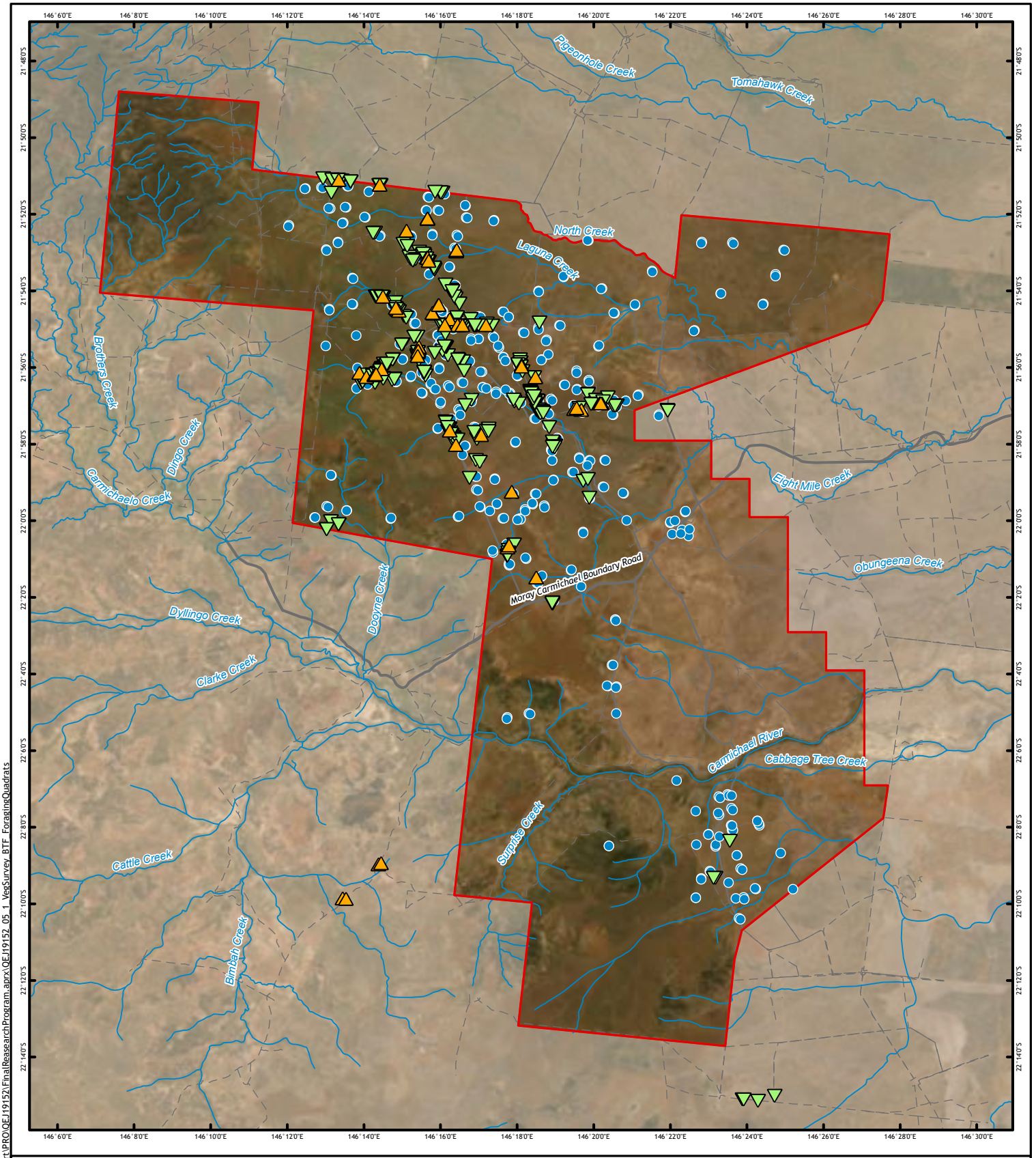
The conservation of black throated finch requires knowledge of their feeding ecology to facilitate the protection and management of critical habitat features that affect productivity and survival. This study aimed to identify the dietary preferences of the black-throated finch in the Desert Uplands Bioregion of Queensland, Australia, and determine whether they preferentially select food resources based on seasonal availability, profitability and/or seed morphology. To achieve this, we (i) investigated the diet composition by black-throated finch through field observations and analysing crop and faecal samples, (ii) compared these findings across seasons and against the composition of vegetation in the broader landscape to determine whether they display any preferences for specific grass species; and (iii) analysing the nutritional content and morphological properties of grass seeds to determine whether black throated finch select their preferred food due to profitability of handling time. By understanding the foraging behaviour and dietary requirements of the black throated finch, the results of this work will contribute to the development and implementation of management strategies to promote the availability of important food sources throughout black-throated finch habitat.

5.1.3 Methods

5.1.3.1 Study area

The study was undertaken over approximately 70,000 ha centred on 21° 59' S, 146° 21' E and located approximately 150 km north-west of Clermont and 190 km south of Charters Towers (Figure 5.1-1), within the Alice Tablelands subregion of the Desert Uplands Bioregion in Central Queensland. The Desert Uplands cover an area of approximately 70, 000 km^2 and are characterised by sandstone ranges and sand plains with low-fertility soils and eucalypt and acacia woodlands with a spinifex understorey. The region has a semi-arid climate with variable annual rainfall of approximately 460 mm, and the predominant land use in the region is cattle grazing, with some sheep grazing in the west (Sattler, 1999; Bastin *et al.*, 2009). The Desert Uplands Bioregion coincides with the southern extent of the current known range of the black-throated finch as well as the Galilee Basin where more than 50% of potential black-throated finch habitat occurs under mining leases (Vanderduys *et al.*, 2016b; Reside *et al.*, 2019).

The study area is characterised by cattle grazing and native vegetation, of which approximately 33,400 ha is managed as an environmental offset under the Commonwealth Environment Protection and Biodiversity Protection Act 1999, and approximately 33,660 ha consists of a portion of the Carmichael Coal Mine mining lease. Areas of currently active mining activities have been excluded from the study area. Approximately 79% of the study area consists of remnant vegetation, comprising 24 Regional Ecosystems (REs) (Neldner *et al.*, 2022). The vegetation is dominated by eucalypt woodlands, particularly Regional Ecosystems 10.5.5a (*Eucalyptus melanophloia* woodland on plains) and 10.5.1 (*Eucalyptus similis* and/or *Corymbia brachycarpa* woodland on sand plains) that account for 73.5% of the remnant vegetation within the study area. Other habitats include acacia woodlands and shrublands, with approximately 10, 207 ha of the study area cleared for grazing pastures, roads, and mining infrastructure. Darkies Range borders the western extent of the study area, with one major watercourse, the Carmichael River, and several other smaller creeks including North Creek and Tomahawk Creek.



Legend

BTF Foraging Quadrats
- Dry Season
- Wet Season

Vegetation Survey Sites

Major Road

Minor Road

Track

Watercourse

Project Area

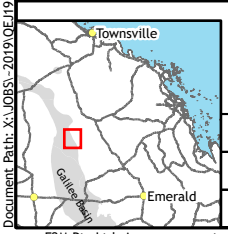
N

Scale 1:250,000 (A4)

0 5 10

Kilometres

Coordinate System: GDA2020



Notes:
Aerial Imagery: © ESRI 2021
Cadastre: © DNRME 2021
Watercourse: © Geoscience Australia 2012
Road: © DNRME 2021

Rev	Description	Drawn	Approved	Date
2	Issued for Use	JY	JMC	30/10/2025
0	Issued for Use	GO	JMC	08/08/2025

FIGURE 5.1-1: VEGETATION SURVEY SITES AND BTF FORAGING QUADRATS

BTFRP Final Report
Bravus

Map Number	Job Number	Rev
1 of 1	QEJ19152	2

Document Path: X:\JOBS\-2019\QEJ19152\GIS\Final Research Program Report\PRO\QEJ19152\FinalReasearchProgram.aprx\QEJ19152_05_1_VegSurvey_BTF_ForagingQuadrats

E2M Pty Ltd gives no warranty in relation to the data (including accuracy, reliability, completeness or suitability) and accepts no liability for any loss, damage or costs (including consequential damage) relating to any use of the data in this map.

5.1.3.2 Survey design

Seasonal surveys were undertaken over eight periods between March 2020 and November 2023 across the wet (December to May) and dry seasons (June to November) to detect any seasonal differences in habitat use and resource availability. Two surveys were conducted in each year, one during the wet season and one during the dry season. Surveys involved the collection of direct observational data on black-throated finch feeding behaviour and ground cover, as well as seed collection, with crop and faecal sample collection occurring during a subset of surveys. The methods for data collection and analysis for each data type are described below.

5.1.3.3 Foraging observations

Where foraging black-throated finch were encountered, we observed their behaviour and, where possible, the species on which they were foraging. We recorded 204 foraging observations, the majority of which were recorded in the wet season (n=182), when birds often forage directly from the grass inflorescence, facilitating identification of the forage species. During the dry season, identification of the food source was more difficult during field observations as most grasses had typically dropped all seed and birds largely forage from seeds on the ground. For each observation of black-throated finch foraging we recorded the grass species black-throated finch observed to be foraging on.

5.1.3.4 Vegetation plots

At locations where black-throated finch were observed foraging, vegetation plots were assessed to determine the composition and cover of the ground layer at each foraging site. More specifically, we assessed a 2 m x 1 m quadrat centred on the location where birds were feeding, for a total of 425 groundcover plots. This includes 192 plots assessed where birds were observed foraging but no direct observation of the specific foraging species was recorded (e.g. birds flushed before the target species could be identified). The composition of the ground layer was also assessed at 220 randomly selected sites across the study area. At each of these sites, ten 1 m x 1 m quadrats were analysed to characterise the composition and availability of groundcover across the study area. The following characteristics were visually assessed and recorded for each quadrat: (i) grass species richness, (ii) projected foliage cover of each ground cover species, (iii) projected foliage cover of each shrub species, (iv) organic litter cover, (v) bare ground cover, (vi) rock cover, (vii) coarse woody debris cover; and (viii) seed abundance and species of seeding plants.

5.1.3.5 Faecal and crop sample collection

Crop and faecal sampling were used to supplement direct foraging observations and gain a greater understanding of black-throated finch diet, particularly during the dry season. Crop and faecal samples were collected over three survey periods between October 2022 and October 2023. These samples were collected from birds captured by mist-netting at permanent and ephemeral water sources where black-throated finch are known to drink. A total of 71 crop and 176 faecal samples were collected during these surveys, including 27 and 87, respectively, during the dry season (Table 5.1-1).

Crop samples were collected using the tube insertion method (TIM), a non-destructive method of crop sampling developed for zebra finches (*Taeniopygia guttata*) (Zann and Straw, 1984) and modified for use on firetail finches. The TIM involves the insertion of a soft, flexible plastic tube into the bird's throat to collect seeds from the crop. Tubing (3.5 mm in diameter) was carefully guided into the crop by pushing on the membranous crop wall from the outside, and seeds were gently pushed into the tubing and extracted. The section of tubing containing the seed(s) was then placed in a sealed envelope surrounded by silica gel to desiccate the sample. The TIM method provides minimal disturbance to birds, and no direct effects on mortality have been reported (Zann and Straw, 1984) or were observed during the sampling.

Faecal samples were collected when birds defecated within their holding bag. The faecal sample was then transferred to a vial containing a 100% ethanol solution and stored for laboratory analysis.

Table 5.1-1: Number of crop and faecal samples taken during mist-netting

Survey	Number of samples		
	Crop	Faecal	Total
2022 Dry Season	16	51	67
2023 Dry Season	11	36	47
2023 Wet Season	44	89	133
Total	**71**	**176**	**247**

5.1.3.6 Ethics statement

Mist-netting and crop sampling were conducted under the animal ethics permit CA 2020/07/1392, issued by the Queensland Department of Agriculture and Fisheries; research permit WA0025814, issued by the Queensland Department of Environment, Science and Innovation; and Australian Bird and Bat Banding Scheme (ABBBS) authority 2832-01, issued by the Australian Government Department of Climate Change, Energy, the Environment and Water.

5.1.3.7 DNA metabarcoding

To examine potential differences in diet between the wet and dry seasons, crop (n = 71) and faecal (n = 176) samples were analysed, comprising 114 (27 crop and 87 faecal) dry-season samples and 133 (44 crop and 89 faecal) wet-season samples (Table 1). Analysis of the biological content in black-throated finch faecal and crop samples was undertaken by the Griffith University DNA Sequencing Facility through metabarcoding of the cytochrome oxidase I (COI), internal transcribed spacer 2 (ITS2) and ribulose-bisphosphate carboxylase (rbcl) gene regions. Subsequent identification of taxa from the DNA sequences was undertaken using the QIIME2 pipeline by Blue Mountain Analytics. Publicly available genomic databases were supplemented through Sanger sequencing of locally sourced plant material. DNA was extracted from this plant material using a DNeasy Pro Plant kit (Qiagen). Confirmation of DNA quality and quantity was conducted by Nanodrop quantification.

5.1.3.8 Seed nutritional content

Seeds were collected directly from the plant of grass species black-throated finch were observed consuming and other common grasses in the study area in proximity to foraging locations. Seeds were stored in paper bags, dried in an electric fan-forced drier at 60ºC for 12 hours and dehusked during the field survey (Crowley and Garnett, 2001; Stock *et al.*, 2013). A total of 64 samples, where sufficient seed mass could be collected (at least 10 g dried weight), were collected from 36 grass species between 2021 and 2022 and sent to the NSW Department of Primary Industries laboratory at Wagga Wagga for nutritional content analysis. Following the methodology outlined by Weier *et al.* (2017), the following data were recorded, for each sample: (i) total dry weight, (ii) crude protein, (iii) crude fat, (iv) fibre, (v) water-soluble carbohydrates, (vi) ash content, (vii) metabolisable energy, (viii) carbohydrate content, and (ix) amino acid suite (including threonine and tyrosine, which are essential amino acids that must be obtained directly from seeds; Weier *et al.*, 2017).

5.1.3.9 Seed morphology

Four seeds from each of 28 grass species were measured for length (mm) and width (mm), and the mean length and width of seeds from an additional 12 species were collated from the AusGrass2 database (Simon and Alfonso, 2011). A subsample of seeds from these 28 grass species were weighed using digital scales to determine the mean weight (g) of seeds.

5.1.3.10 Statistical analyses

The DNA metabarcoding of crop and faecal samples was unable to sequence some taxa below family level (i.e. Poaceae). Therefore, the statistical analysis of black-throated finch diet from the crop and faecal samples only considered taxa that was identified to genus level.

Foraging plots and systematic vegetation survey plots were compared in terms of the relative abundance of grass species found to be important in the black-throated finch diet and/or to be prevalent in foraging areas using Mann–Whitney U tests. To compare species composition in faecal and crop samples between seasons, we calculated the Bray-Curtis dissimilarity index, which quantifies compositional differences based on species presence and abundance. We then used permutational multivariate analysis of variance (PERMANOVA) with 999 permutations, implemented using the *adonis* function in R's *vegan* package (Oksanen *et al.*, 2024), to assess whether community composition varied between the wet and dry seasons. All exploratory and statistical analyses were conducted using R statistical software (R Core Team, 2023).

5.1.3.10.1 Diet selection preference

To determine whether black-throated finch in the Desert Uplands preferentially select particular grass species to forage we assessed the availability of food sources across the study area and the contribution of those species (Jacobs, 1974; Dierschke, 2002; Magome *et al.*, 2008). An availability index for each food source was calculated by dividing the total cover for each taxon by the total cover for all taxa recorded at 220 vegetation survey sites assessed across the study area. We then calculated a selectivity index for each food source based on the number of times each species had been observed foraged upon divided by the total number of foraging observations of all taxa. We also calculated a selectivity index for each taxon based on the proportion of diet calculated from the crop and faecal DNA analysis.

Diet selection preference was then calculated for both observational and DNA data using Jacobs' selectivity index (Jacobs, 1974):

$$D = (r-p)/(r + p - 2 r*p)$$

where D = Jacobs' selectivity index, r = selection index and p = availability index.

Jacobs' selectivity index values range from -1 to 1 where 1 is strong preference and -1 is a weak preference.

5.1.3.10.2 Analysis of factors influencing diet selection

To determine the factors that may be influencing the preferential selection of grass taxa by black-throated finch, we calculated the Spearman's correlation coefficient for the nutritional and physical properties of the seeds of 20 grass species against the Jacobs' Index.

5.1.4 Results

5.1.4.1 Characteristics of black-throated finch foraging locations

A total of 337 grass species were identified in the 425 foraging plots and 220 systematically collected vegetation survey plots combined. grass species recorded in foraging and vegetation plots numbered 115 and 329, respectively, with 107 species of overlap.

Foraging plots differed significantly from systematically collected vegetation survey plots in terms of the average ground cover of several types of grasses. More specifically, foraging plots had significantly higher mean cover of *Digitaria brownii, D. ciliaris, Panicum effusum, Paspalidium rarum, Enneapogon virens* and *Urochloa subquadripara* (Table 5.1-2). Mean bare ground cover was also significantly higher in foraging plots (54.4 ± 20.8%) than in vegetation plots (22.7% ± 12.7%; U = 2,218,653; $p < 0.0001$). In contrast, average cover of *Cenchrus ciliaris, Heteropogon contortus, Sehima nervosum* and *Themedra triandra* was significantly lower in foraging plots than in systematically collected vegetation plots (Table 5.1-2).

Table 5.1-2 Summary statistics (number of sites, mean percentage cover and standard deviation) and comparison (Mann-Whitney U) of the cover of important grass species in foraging and systematically collected vegetation survey plots

Species	Foraging			Systematic			U statistic	p-value
	n	mean	SD	n	mean	SD		
Aristida pruinosa	18	5.5	5.9	61	5.4	4.4	1380	0.94
Aristida sp.	64	6.2	6.0	100	8.7	8.5	11,110	0.15
Cenchrus ciliaris	78	8.4	9.9	160	12.4	11.1	27,242	**<0.001**
Chrysopogon fallax	84	5.3	5.1	164	4.8	4.2	38,835	0.07
Digitaria brownii	57	6.5	5.8	152	5.1	4.9	18,914	**0.01**
Digitaria ciliaris	162	11.0	11.7	23	5.0	6.2	4926	**<0.0001**
Enneapogon virens	75	8.1	6.5	141	4.8	4.0	33,067	**<0.0001**
Eriachne mucronata	23	8.4	5.4	117	11.7	10.7	6982	0.4
Heteropogon contortus	121	9.2	10.9	178	12.0	10.5	69,176	**<0.0001**
Panicum effusum	89	5.5	5.6	132	4.8	6.2	26,712	**<0.001**
Paspalidium rarum	77	7.8	7.4	16	1.2	0.9	1778	**<0.0001**
Schizachyrium fragile	21	5.3	3.9	68	9.3	10.6	2553	0.87
Sehima nervosum	22	7.0	8.5	111	11.1	9.7	3968	**0.03**
Themeda triandra	33	6.2	7.7	168	11.7	10.4	12,031	**<0.0001**
Urochloa subquadripara	20	5.4	3.8	4	1.1	1.7	72	**0.01**

5.1.4.2 Diet composition

5.1.4.2.1 Foraging observations

During field observations black-throated finch were observed foraging directly from grasses, mainly during the wet season, often jumping up from the ground to pull the seed head to the ground and pecking the seed head while being held down by their foot. Out of the 425 observations of birds foraging, 204 observations were able to identify the specific species being foraged. A total of 34 grass species were identified in these observations, the most common forage species being *Digitaria ciliaris* (33.3% of observations), *Paspalidium rarum* (12.9%), *Panicum effusum* (5.3%), *Digitaria brownii* (4.5%), *Cenchrus ciliaris* (4.2%), *Urochloa mosambicensis* (3.8%) and *Urochloa subquadripara* (3.8%) (Table 5.1-3). Of these, only *Digitaria ciliaris* and *Digitaria brownii* were observed being foraged on during the dry season, when birds would typically feed on fallen seeds from the ground and grasses were not usually holding seeds.

Table 5.1-3: Visual foraging observations of southern black-throated finch within the study area between 2021 and 2023

Species	Wet	Dry	Total	Percentage
Digitaria ciliaris	78	10	88	33.3
Paspalidium rarum	34	0	34	12.9
Panicum effusum	14	0	14	5.3
Digitaria brownii	9	3	12	4.5
Cenchrus ciliaris	11	0	11	4.2
Urochloa mosambicensis	10	0	10	3.8
Urochloa subquadripara	10	0	10	3.8
Mnesithea formosa	8	0	8	3.0
Dactyloctenium radulans	7	0	7	2.7
Enneapogon virens	5	2	7	2.7
Tragus australianus	4	3	7	2.7
Alloteropsis semialata	5	0	5	1.9
Chrysopogon fallax	5	0	5	1.9
Schizachyrium fragile	5	0	5	1.9
Sehima nervosum	4	1	5	1.9
Digitaria ammophila	3	1	4	1.5
Aristida pruinosa	1	2	3	1.1
Aristida sp.	2	1	3	1.1
Bothriochloa pertusa	2	1	3	1.1
Enneapogon sp.	3	0	3	1.1
Echinochloa colona	2	0	2	0.8

Species	Wet	Dry	Total	Percentage
Eragrostis cumingii	2	0	2	0.8
Eriachne mucronata	0	2	2	0.8
Paspalidium sp.	2	0	2	0.8
Urochloa sp.	2	0	2	0.8
Whiteochloa airoides	2	0	2	0.8
Aristida calycina	0	1	1	0.4
Aristida jerichoensis	1	0	1	0.4
Melinis repens	1	0	1	0.4
Setaria surgens	1	0	1	0.4
Themeda avenacea	1	0	1	0.4
Themeda triandra	1	0	1	0.4
Trianthema triquetrum	1	0	1	0.4
Urochloa holosericea	1	0	1	0.4
Total	**237**	**27**	**264**	**100**

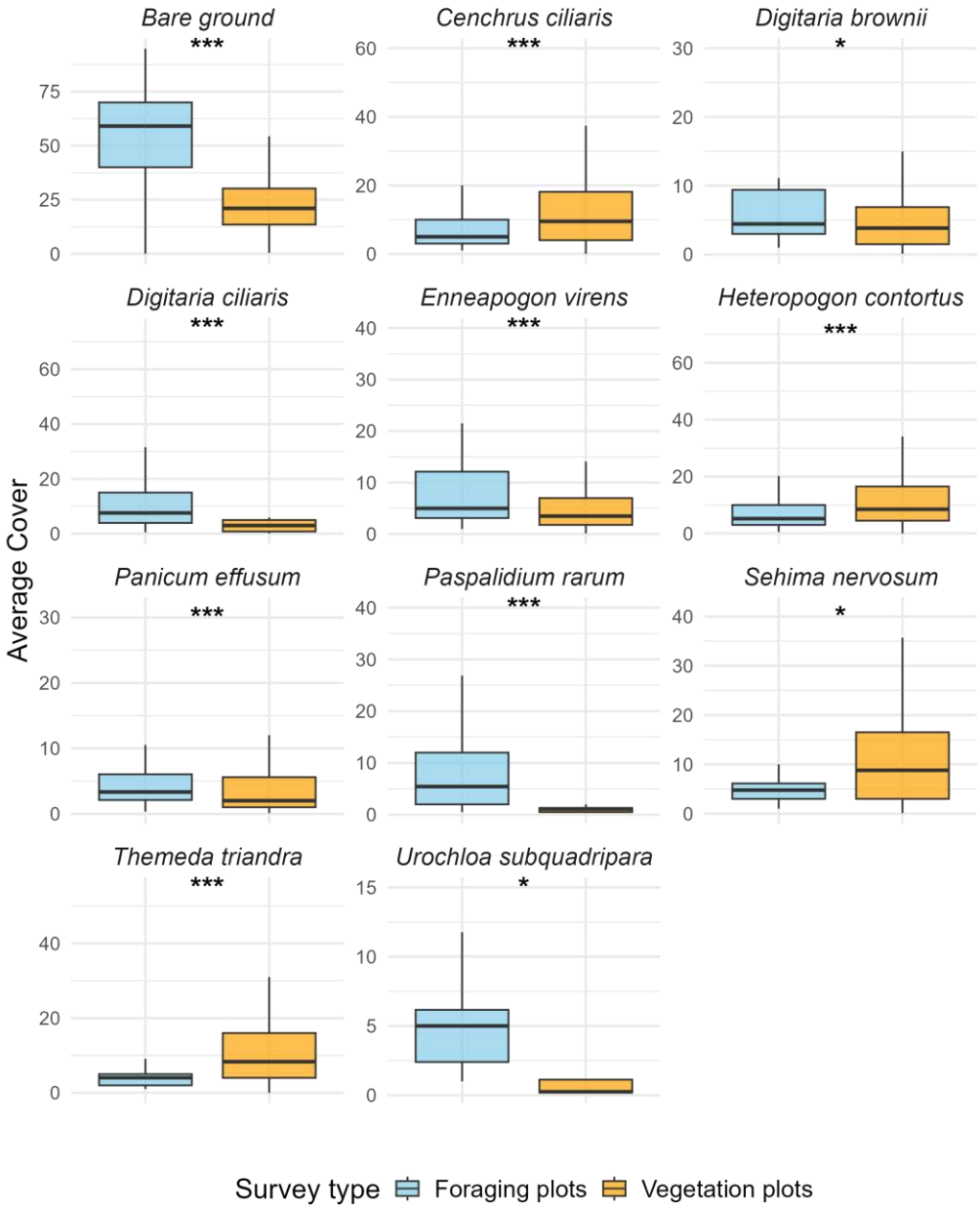


Figure 5.1-2 Box plots comparing the cover (%) of grass species and bare ground recorded at black-throated finch foraging site (blue) and systematic vegetation plots (orange) across the study area: (a) bare ground; (b) *Cenchrus ciliaris*; (c) *Digitaria brownii*; (d) *Digitaria ciliaris*; (e) *Enneapogon virens*; (f) *Heteropogon contortus*; (g) *Panicum effusum*; (h) *Paspalidium rarum*; (i) *Sehima nervosum*; (j) *Themeda triandra*; and (k) *Urochloa subquadripara*.

5.1.4.2.2 *Digitaria ciliaris* foraging observations

The high proportion of visual observations of SBTF foraging on a non-native species, *Digitaria ciliaris*, appears to contrast with findings that SBTF generally select habitat with low exotic species cover (Section 4.1 and 4.2). *Digitaria ciliaris* occurs at very low densities across the study area, with an average of 0.1% cover across all systematic vegetation monitoring plots that have been randomly placed throughout SBTF habitat (Figure 5.1-2). However, the species is present at moderate to high densities along tracks, with covers of up to 70% recorded within foraging plots. The average cover of *Digitaria ciliaris* decreases as the distance from a track increases, with the furthest record being approximately 650 m from the nearest track (Figure 5.1-3). This trend occurs at a scale too fine to be captured by the vegetation monitoring plots, as 84.9% of all plots containing *Digitaria ciliaris* are located within 10 m of a track.

Foraging observations were also strongly concentrated along tracks (Figure 5.1-3; Figure 5.1-4); however, this is likely to be at least partly due to survey bias. Survey effort for SBTF was greater along access tracks, due to radio tracking efforts commencing along tracks and incidental observations largely occurring when driving among survey sites. When sighted, SBTF foraging along tracks are easier to observe for longer and identify forage species. As a result, the average distance of foraging plots from a track was 37.9 m, compared with 195.3 m for vegetation plots. Consequently, it is difficult to distinguish survey bias in foraging observations along tracks from the high proportion of SBTF foraging recorded on *Digitaria ciliaris*.

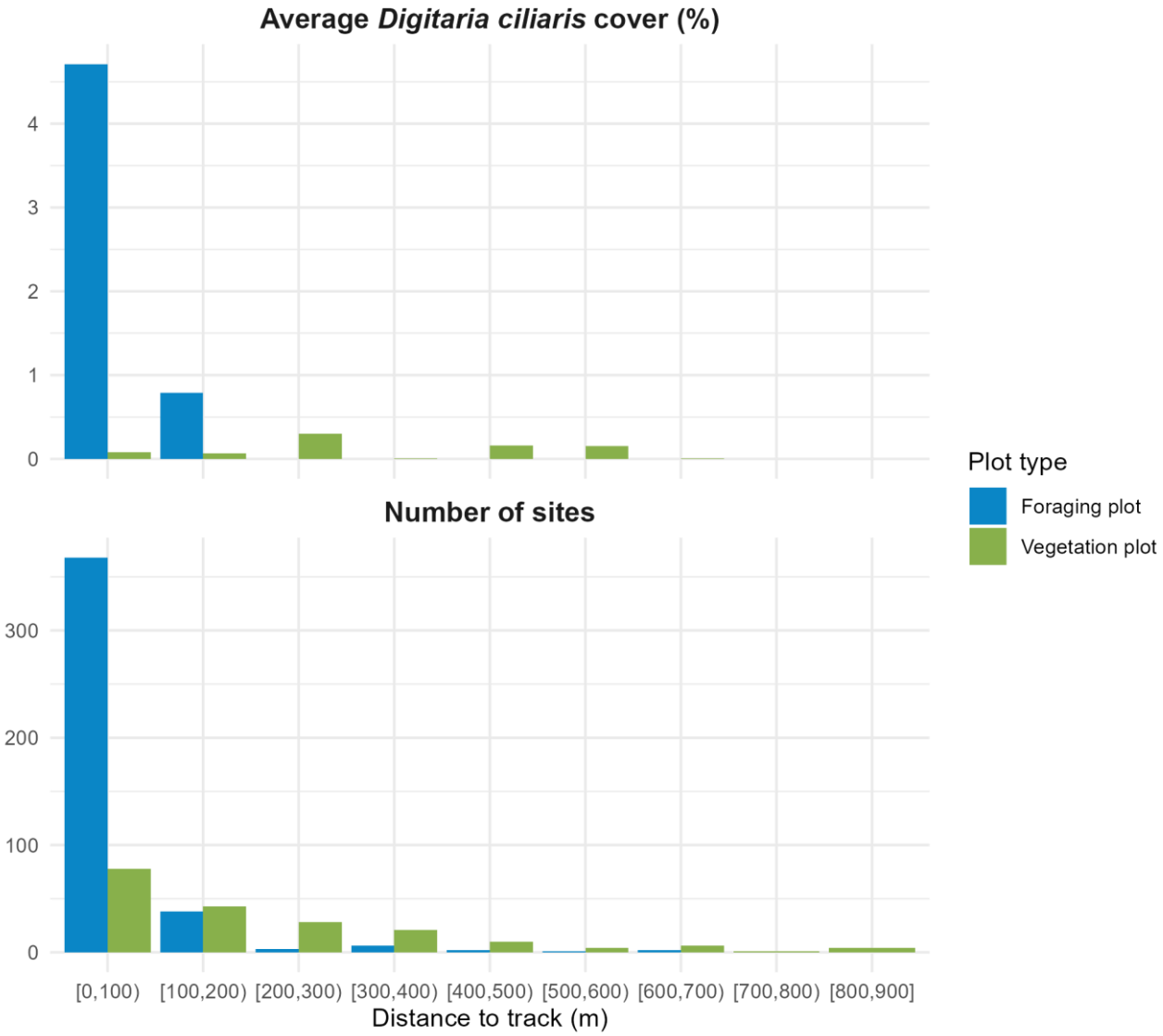
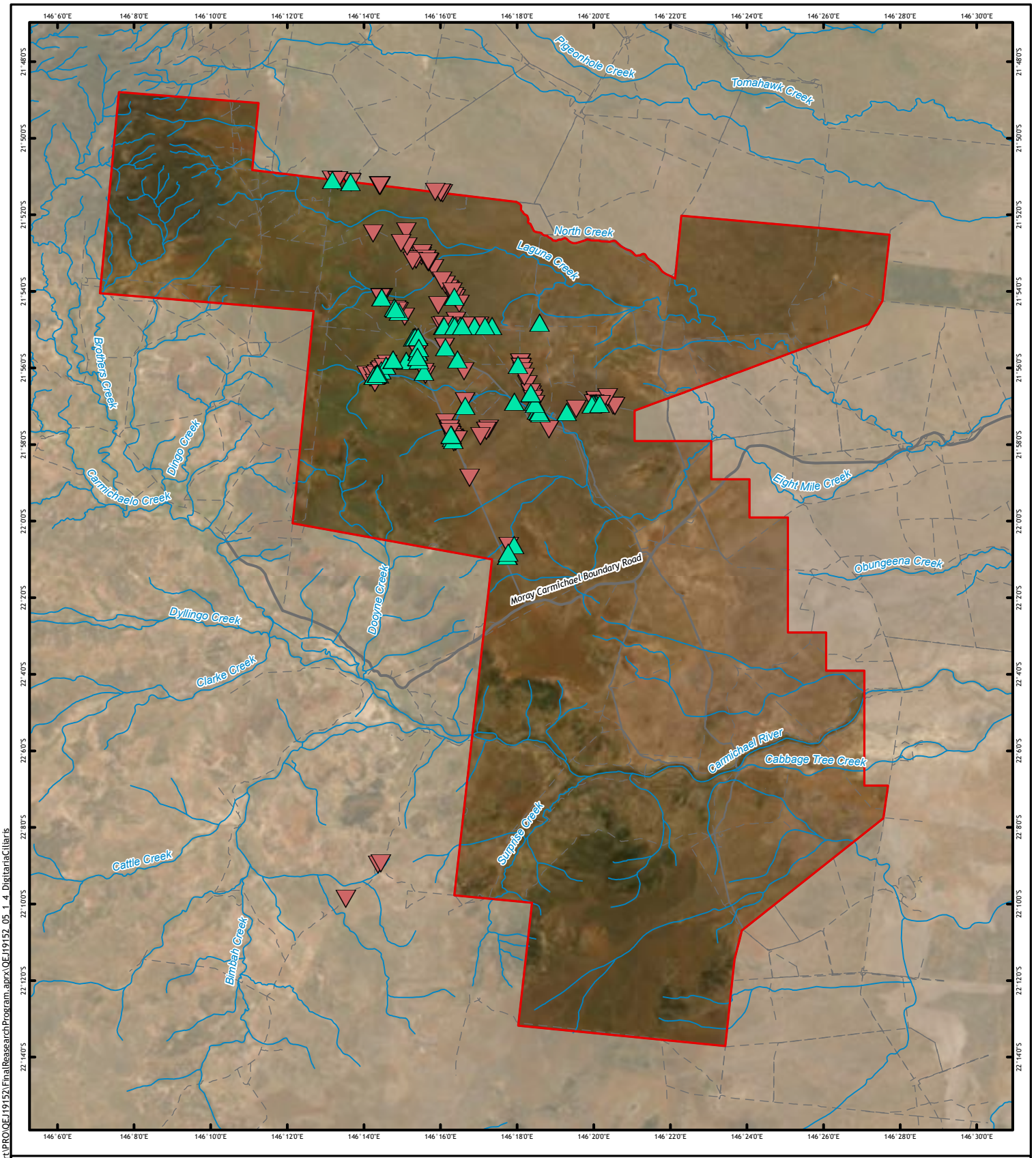


Figure 5.1-3: Histogram depicting the average percentage of *Digitaria ciliaris* cover per site (top) and number of sites (bottom). Results are divided by whether the data was captured via a systematic vegetation plot consisting of 10 1x1 m quadrats per site or an opportunistic foraging plot consisting of a 1x2 m quadrat undertaken after an SBTF was observed foraging.



Legend

- BTF Foraging on *Digitaria ciliaris*
- BTF Foraging on other species
- Major Road
- Minor Road
- Track
- Watercourse
- Project Area

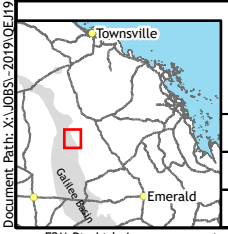
N

Scale 1:250,000 (A4)

0 5 10

Kilometres

Coordinate System: GDA2020



Notes:
Aerial Imagery: © ESRI 2021
Cadastre: © DNRME 2021
Watercourse: © Geoscience Australia 2012
Road: © DNRME 2021

Rev	Description	Drawn	Approved	Date
2	Issued for Use	JY	JVO	28/11/2025

FIGURE 5.1-4: *DIGITARIA CILIARIS* WITHIN FORAGING QUADRATS

BTFRP Final Report
Bravus

Map Number	Job Number	Rev
1 of 1	QEJ19152	2

Document Path: X:\JOBS\-2019\QEJ19152\GIS\Final Research Program Report\PRO\QEJ19152\FinalReasearchProgram.aprx\QEJ19152_05_1_4_DigitariaCiliaris

E2M Pty Ltd gives no warranty in relation to the data (including accuracy, reliability, completeness or suitability) and accepts no liability for any loss, damage or costs (including consequential damage) relating to any use of the data in this map.

5.1.4.2.3 Observation of opportunistic nectarivory

One observation of opportunistic nectarivory by black-throated finch was recorded during the 2022 wet season. Six black-throated finches were photographed and filmed actively obtaining nectar from the open flowers of a *Grevillea pteridifolia*. The foraging lasted for approximately 5 minutes, after which the flowers of the *Grevillea pteridifolia* were inspected. No insects were observed on the flowers and an abundance of nectar was present, suggesting the black-throated finch were targeting nectar available in the open flowers.

5.1.4.2.4 DNA metabarcoding of faecal and crop samples

Deoxyribonucleic acid (DNA) sequencing of 247 crop and faecal samples identified 46 plant taxa consumed by black-throated finch, including 26 grass genera, 18 herbaceous genera and two tree / shrub genera (Table 4). Over a quarter (33%) of the diet composition, as determined by the DNA analysis, could only be sequenced to family level (Poaceae) due to limitations in the DNA metabarcoding. This component of the data was excluded from further analysis.

Of the remaining taxa, six grass genera contributed 61% of black-throated finch diet recorded through the DNA analysis. *Digitaria* spp. was the mostly commonly recorded (n=166; 67% of birds) as well as providing the highest contribution to the proportion of the diet (mean = 31.8%) of black-throated finch (Table 5.1-4).

Table 5.1-4: DNA metabarcoding results from crop and faecal samples taken between 2022 and 2023 showing the number of samples in which each taxon was recorded, as well as the mean and maximum proportion contributed to each sample.

Taxa	No. of samples (n=247)	Mean	SD	Max	Wet season		Dry season	
					n	Mean	n	Mean
Grasses								
Poaceae	208	33.02%	32.44%	100.00%	115	38.93%	93	25.70%
Digitaria	166	31.83%	34.26%	100.00%	101	36.16%	65	25.09%
Aristida	139	15.43%	20.13%	92.50%	57	13.23%	82	16.95%
Enneapogon	116	26.80%	32.96%	100.00%	53	24.23%	63	28.95%
Panicum	114	12.42%	23.58%	100.00%	86	15.83%	28	1.97%
Cenchrus	110	16.27%	23.00%	94.03%	54	5.06%	56	27.07%
Heteropogon	88	15.42%	21.77%	98.09%	34	9.19%	54	19.34%
Themeda	50	6.29%	18.05%	97.40%	30	6.58%	20	5.85%
Echinochloa	45	3.45%	8.07%	46.07%	37	4.07%	8	0.56%
Urochloa	35	14.75%	21.31%	75.39%	14	18.97%	21	11.94%
Eriachne	20	1.19%	2.50%	9.79%	7	0.68%	13	8.00%
Sorghum	20	0.03%	0.04%	0.12%	8	0.03%	12	0.03%
Setaria	18	2.88%	8.16%	26.71%	6	0.03%	12	4.30%
Chrysopogon	14	2.94%	5.66%	15.24%	7	5.86%	7	0.03%

Taxa	No. of samples (n=247)	Mean	SD	Max	Wet season		Dry season	
					n	Mean	n	Mean
Eriochloa	11	0.24%	0.38%	1.35%	11	0.24%	-	-
Eragrostis	9	0.10%	0.19%	0.55%	8	0.05%	1	0.55%
Schizachyrium	8	45.36%	43.36%	99.82%	3	60.17%	5	36.48%
Triodia	8	0.66%	0.58%	1.82%	2	0.91%	6	0.58%
Chloris	4	14.53%	24.85%	51.51%	1	0.04%	3	19.36%
Alloteropsis	3	0.06%	0.07%	0.13%	1	0.06%	2	0.07%
Oxychloris	3	37.63%	50.74%	95.34%	1	0.01%	2	56.43%
Tragus	3	1.20%	1.55%	2.96%	-	-	3	1.20%
Melinis	2	0.14%	0.01%	0.15%	2	0.14%	-	-
Arundinella	1	0.27%	-	0.27%	1	0.27%	-	-
Dactyloctenium	1	0.61%	-	0.61%	1	0.61%	-	-
Elionurus	1	0.00%	-	0.00%	1	0.00%	-	-
Forbs								
Asteraceae	23	11.87%	27.25%	96.62%	11	0.55%	12	22.24%
Fabaceae	20	1.19%	2.50%	9.79%	15	0.57%	5	3.07%
Indigofera	11	0.64%	0.98%	3.20%	8	0.36%	3	1.38%
Evolvulus	8	7.61%	12.35%	34.33%	6	3.99%	2	18.46%
Achyranthes	8	2.28%	5.99%	17.09%	7	2.60%	1	0.09%
Brassica	7	1.10%	2.69%	7.21%	3	0.06%	4	1.88%
Fimbristylis	6	1.67%	2.90%	7.50%	3	3.09%	3	0.24%
Chenopodiaceae	5	9.18%	12.38%	23.30%	3	15.16%	2	0.20%
Phyllanthus	3	0.71%	1.17%	2.07%	1	0.06%	2	1.04%
Salsola	3	7.39%	6.55%	12.57%	1	0.02%	2	11.07%
Boehmeria	2	0.52%	0.72%	1.03%	1	0.01%	1	1.03%
Convolvulaceae	2	0.07%	0.09%	0.13%	1	0.13%	1	0.00%
Euphorbia	2	3.23%	3.07%	5.40%	1	1.06%	1	5.40%
Portulaca	2	0.10%	0.03%	0.12%	1	0.08%	1	0.12%
Cyperus	1	0.01%	-	0.01%	1	0.01%	-	-
Lauraceae	1	18.15%	-	18.15%	-	-	1	18.15%
Melhania	1	0.00%	-	0.00%	1	0.00%	-	-

Taxa	No. of samples (n=247)	Mean	SD	Max	Wet season		Dry season	
					n	Mean	n	Mean
Trifolium	1	0.36%	-	0.36%	-	-	1	0.36%
Trees and shrubs								
Eucalyptus	29	5.72%	15.42%	71.19%	19	3.52%	10	9.90%
Acacia	6	0.02%	0.01%	0.04%	3	0.03%	3	0.02%

Additional components of black-throated finch diet recorded from the DNA analysis of crop and faecal samples identified a minor contribution by insects and arthropods (Figure 5.1-5). This was more pronounced in the dry season, however the proportion of insects and arthropods within the crop and faecal samples was very minor (1.9%). The proportion of insects in black-throated finch diet was higher in the dry season (4.2%) compared to the wet season (0.5%) (Figure 5.1-5).

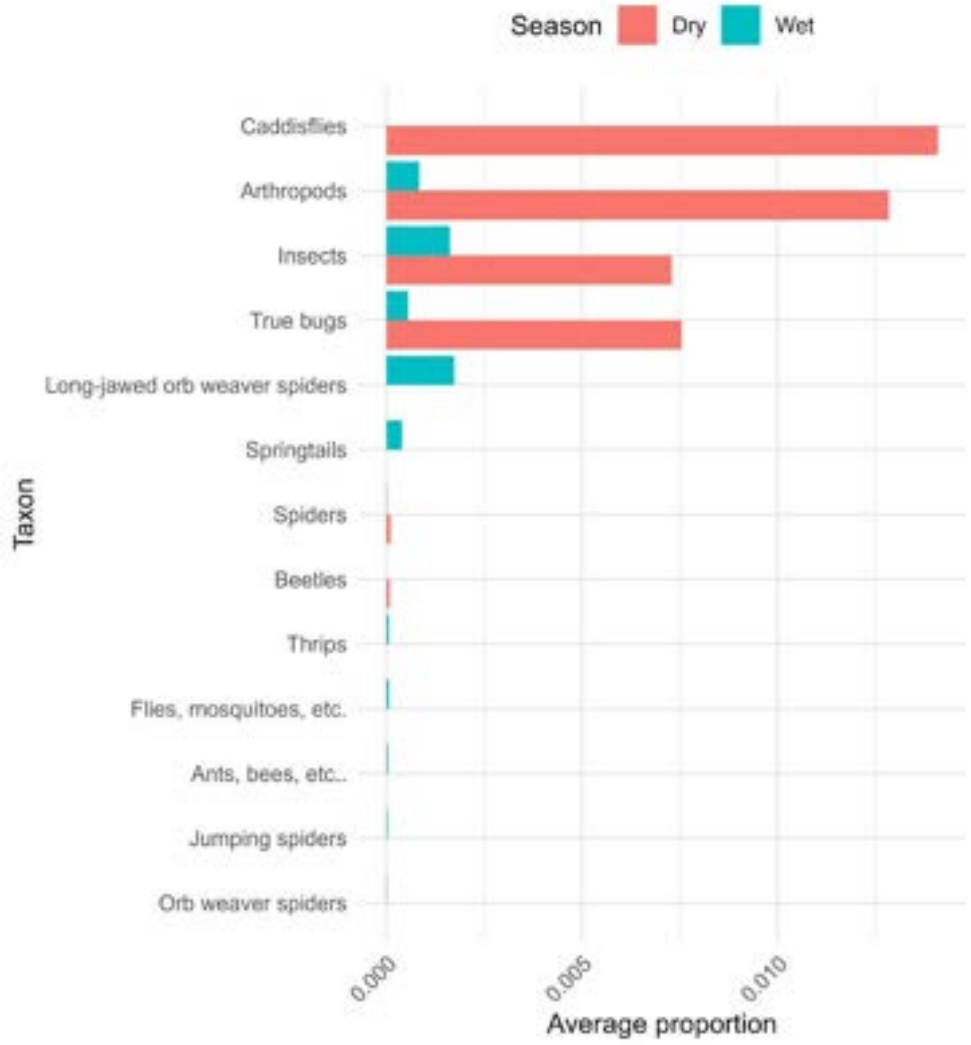


Figure 5.1-5: Contribution of insects and arthropods to the black-throated finch diet during the dry season (red) and wet season (blue) based on DNA metabarcoding records.

5.1.4.3 Seasonal differences in black-throated finch diet

The content of crop and faecal samples were compared across the wet and dry seasons to determine any seasonal variation in black-throated finch diet (Figure 5.1-6). A total of 46 genera were detected in crop (n = 71) and faecal samples (n = 176), with 40 unique genera being detected in the dry season (n = 114) and 42 in the wet season (n = 133). The number of overlapping genera between seasons was 36, with *Tragus, Dactyloctenium, Trifolium* and *Lauraceae* only being detected in the dry season and *Melinis, Melhania, Cyperus, Elionurus, Eriochloa* and *Arundinella* were exclusive to the wet season.

Bray-Curtis dissimilarity between wet and dry seasons was 0.79 (± 0.24), indicating a high degree of seasonal variation in genera composition. Despite this, PERMANOVA results showed that season explained a small but significant portion of the variation (F = 12.27, R2 = 4.8%, p = 0.001), suggesting that while genera composition varied between seasons, substantial overlap remained.

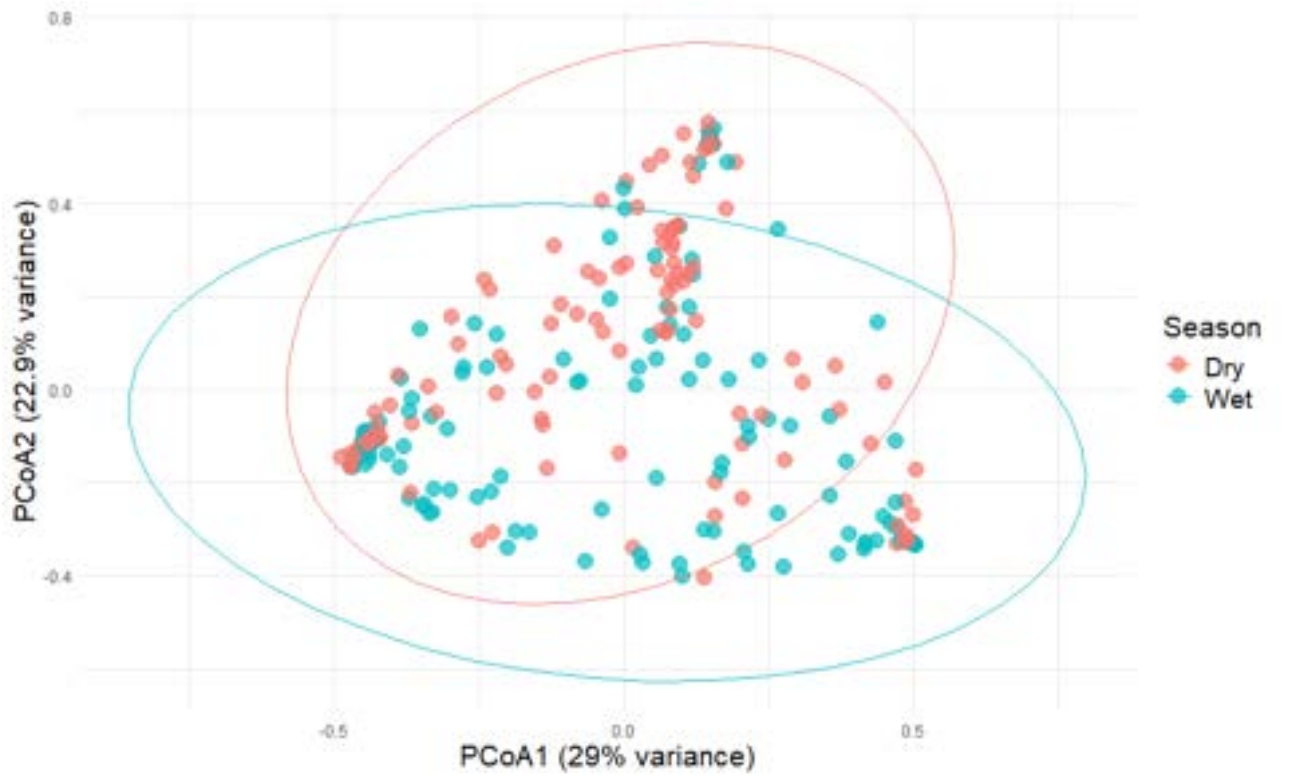


Figure 5.1-6: Principle Coordinate Analysis (PCoA) based on Bray Curtis dissimilarity metrics showing the variance of the diet composition of black-throated finch, based on DNA analysis of crop and faecal samples, between the wet (blue) and dry (red) seasons in the Desert Uplands.

5.1.4.4 Diet selection preferences

The Jacobs' Index was calculated based on forging observation (wet season only) and DNA analysis of crop and faecal samples (wet and dry seasons) to determine whether black-throated finch preferentially selected particular grass species and taxa.

5.1.4.4.1 Selection preferences based on foraging observations

The Jacobs' Index was calculated for 28 grass species that black-throated finch were observed foraging on 231 occasions during the wet season (Jacobs' Index could not be computed for *Echinochloa colona, Paspalidium* sp., *Trianthema triquetrum, Urochloa holosericea* as these were not recorded in systematically collected vegetation survey sites, preventing the calculation of an availability index (Figure 5.1-7). The majority (n=24) of these species had a positive Jacobs' Index, with 17 of these having an index greater than 0.5 and 14 having an index greater than 0.8. Four grass species had a negative Jacobs Index, including *Themeda triandra* (-0.85) (Figure 5.1-7). Grass species with numerous foraging observations and high Jacobs' Index were *Digitaria ciliaris* (Jacobs' Index = 0.99), *Paspalidium rarum* (0.99), *Urochloa subquadripara* (0.99) and *Panicum effusum* (0.68) (Figure 5.1-7). Other species with high Jacobs' Index, such as *Alloteropsis semialata* (0.96), *Eragrostis cumingii* (0.93) and *Melinis repens* (0.95) has very few foraging observations and the high Jacobs' Index is due to a very low availability index.

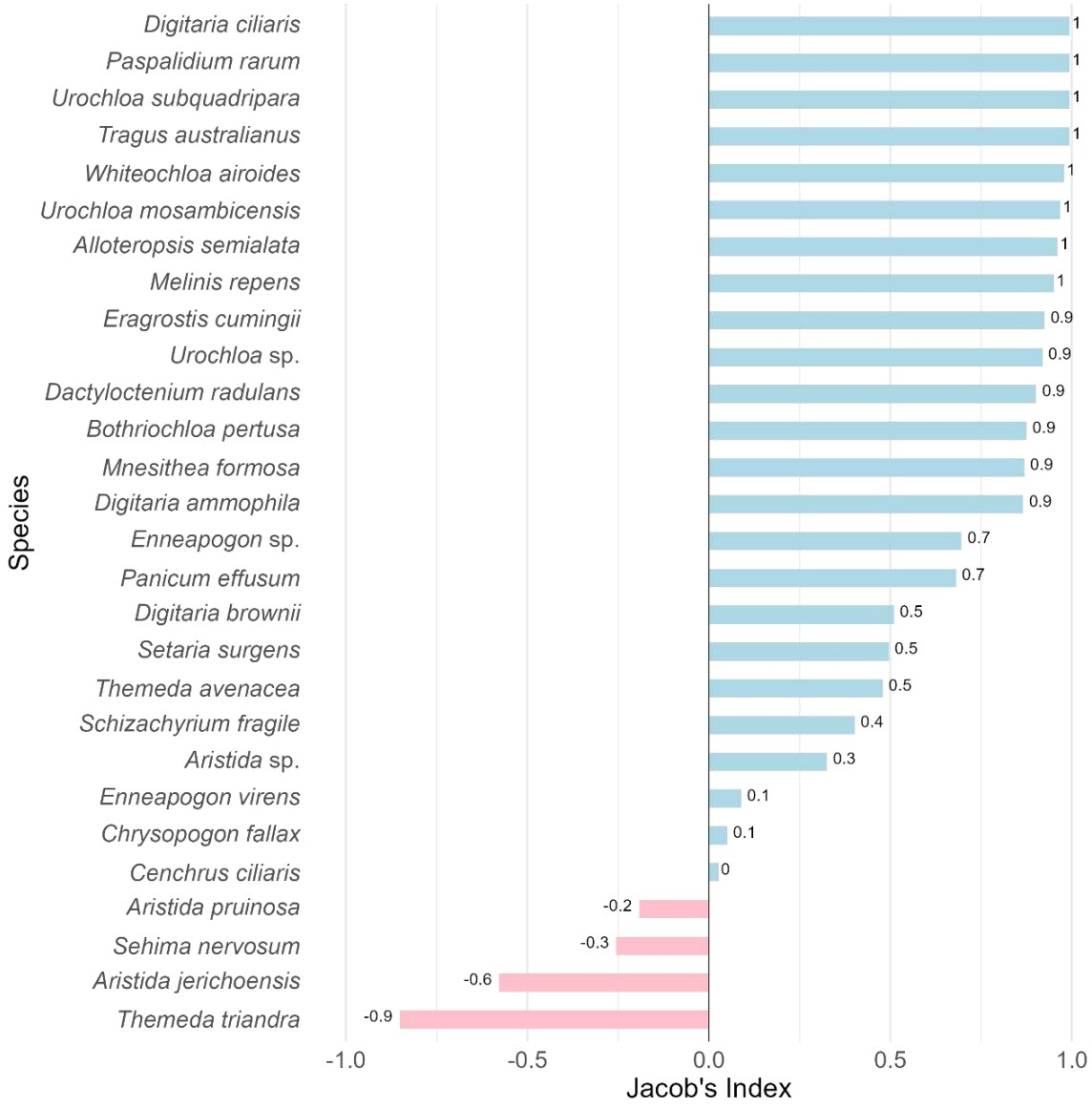


Figure 5.1-7: Jacobs' Index calculated for 28 grass species that black-throated finch were observed foraging on during the wet season between 2021 and 2023.

5.1.4.4.2 Selection preferences based on crop and faecal samples

The Jacobs Index was calculated for 20 taxa found in crop and faecal samples to determine selection preferences for the wet and dry seasons. The results indicate that black-throated finch have a strong preference for three grass taxa, namely *Digitaria* spp., *Urochloa* spp. and *Enneapogon* spp., in both the wet and dry seasons (Figure 5.1-8). Note that *Asteraceae* spp. also had a Jacobs Index of 1, however the selection index is very low (0.01), and the high Jacobs Index is due a very low availability index. Other species were only preferentially selected in one season, including *Panicum* spp. (wet season), *Achyranthes* spp. (wet), *Oxychloris* spp. (dry), *Evolvulus* spp. (dry) and *Cenchrus* spp. (dry) (Figure 5.1-8). The other taxa recorded by DNA analysis of crop and faecal samples were not preferentially selected for in either season and included grass taxa that comprise the dominant ground-cover species within black-throated finch habitat in the study area (i.e. *Aristida* spp., *Heteropogon* spp., *Triodia* spp. and *Themeda* spp.).

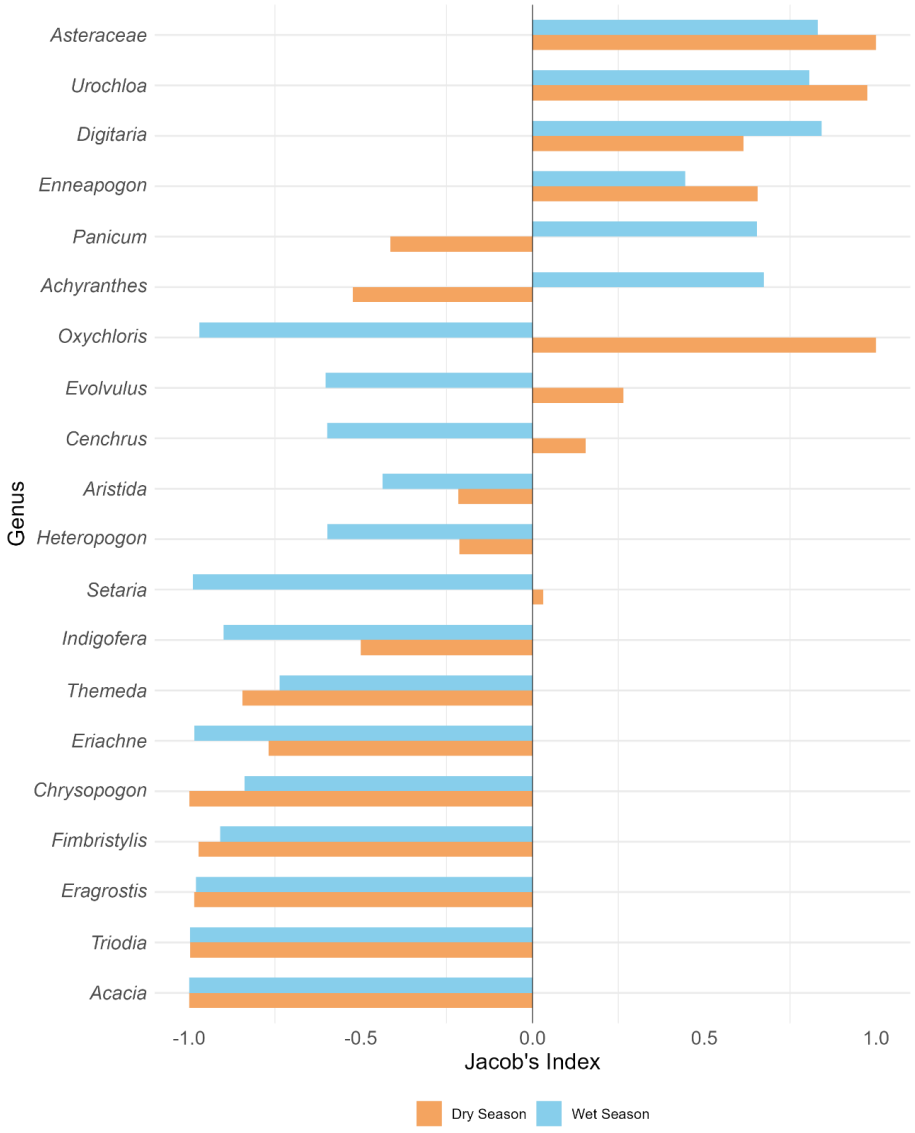


Figure 5.1-8: Jacobs' Index calculated for the wet and dry seasons of 20 of the genera that DNA metabarcoding identified in black-throated finch crop and faecal samples collected between 2022 and 2023.

5.1.4.5 Factors driving the selection of black-throated finch

To determine the factors that influence why black-throated finch prefer particular grasses, we analysed the nutritional content and morphology of grass seeds.

5.1.4.5.1 Seed nutritional content

The results of the seed nutritional analyses identified a relationship between the preferred forage species of black-throated finch and the concentration of key nutritional factors provided by their seeds. In particular, metabolisable energy (Spearman's rho = 0.62, *p* = 0.005; Figure 5.1-9) and crude protein (rho = 0.50; p = 0.03; Figure 5.1-10) were positively correlated to the Jacobs' Index of grass species, indicating that both are important factors in dietary choices of black-throated finch.

Grass seeds containing the highest metabolisable energy content include several preferred diet species; *Digitaria ciliaris*, *Urochloa mosambicensis*, *Panicum effusum* and *Paspalidium rarum* (Figure 5.1-9). Most notably, *Digitaria ciliaris* has by far the highest metabolisable energy (11.5 MJ / kg of dry matter). Grasses with higher Jacobs Index typically contain metabolisable energy levels higher than 7 MJ / kg dry matter, while species with a smaller Jacobs' Index are mostly lower (Figure 5.1-9). This indicates that black-throated finch preferentially select these grasses to maximise their energy intake.

Similarly, the grasses with a higher Jacobs Index also contain higher levels of crude protein. *Digitaria ciliaris* (13.8% of dry matter) also has the highest crude protein content of the grasses that were analysed. Other grass species with high concentrations of crude protein and high Jacobs Index include *Digitaria ammophila*, *Urochloa mosambicensis* and *Paspalidium rarum* (Figure 5.1-10).

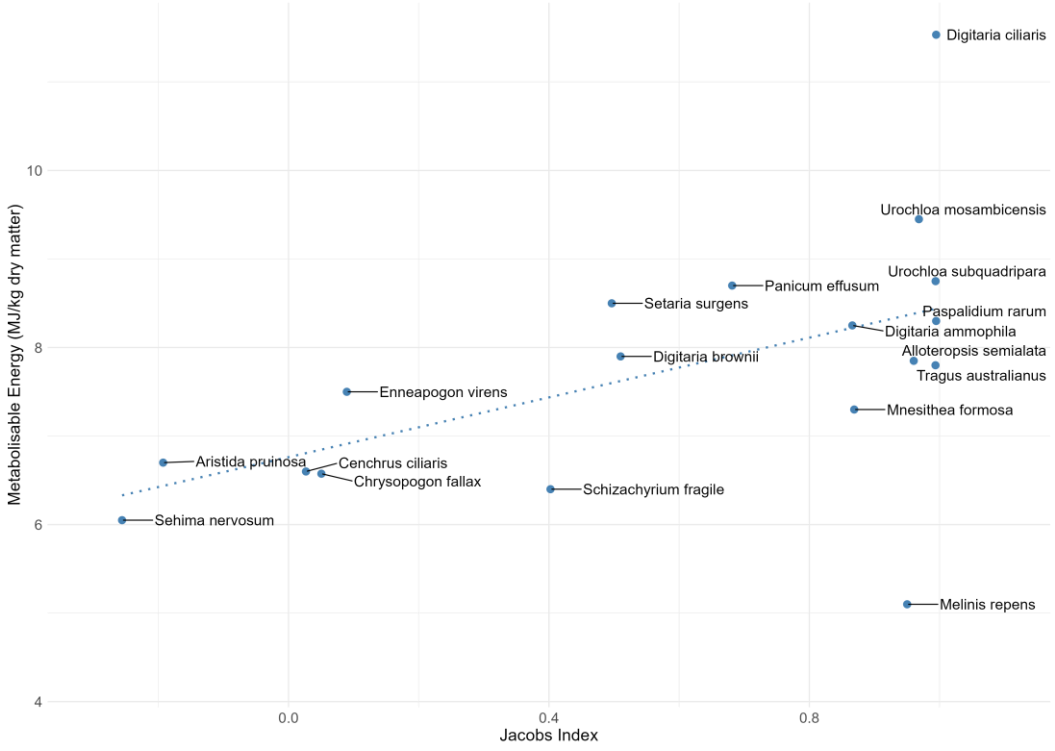


Figure 5.1-9: Relationship between Jacobs Index and metabolisable energy content of the seeds of 18 grass species that occur in the study area.

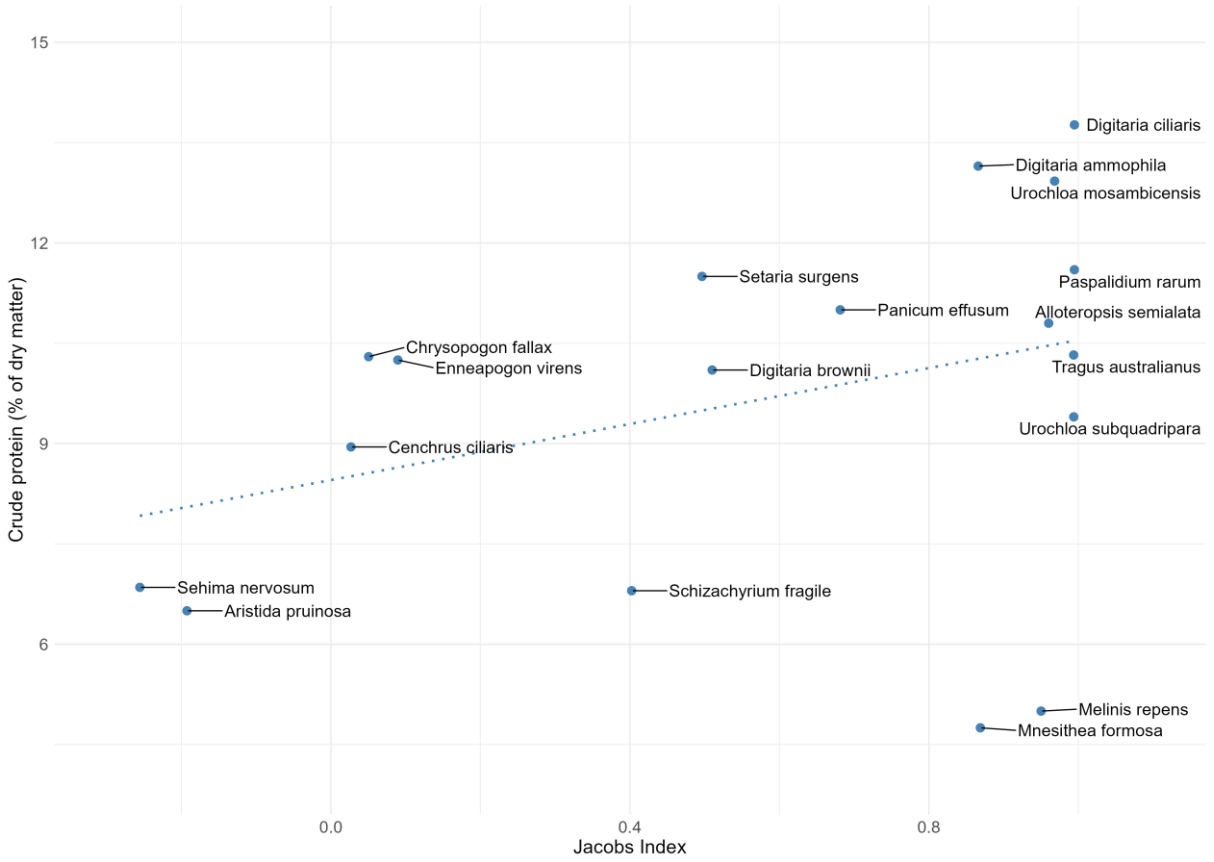


Figure 5.1-10: Relationship between Jacobs' Index and crude protein content of the seeds of 18 grass species that occur in the study area.

5.1.4.5.2 Seed morphology

A negative correlation was identified between preferred forage species of black-throated finch and the length of seeds (rho = -0.56; p = 0.009; Figure 5.1-11), indicating that smaller seeds are preferred. However, no relationship was found with seed width (rho = -0.18; p = 0.436).

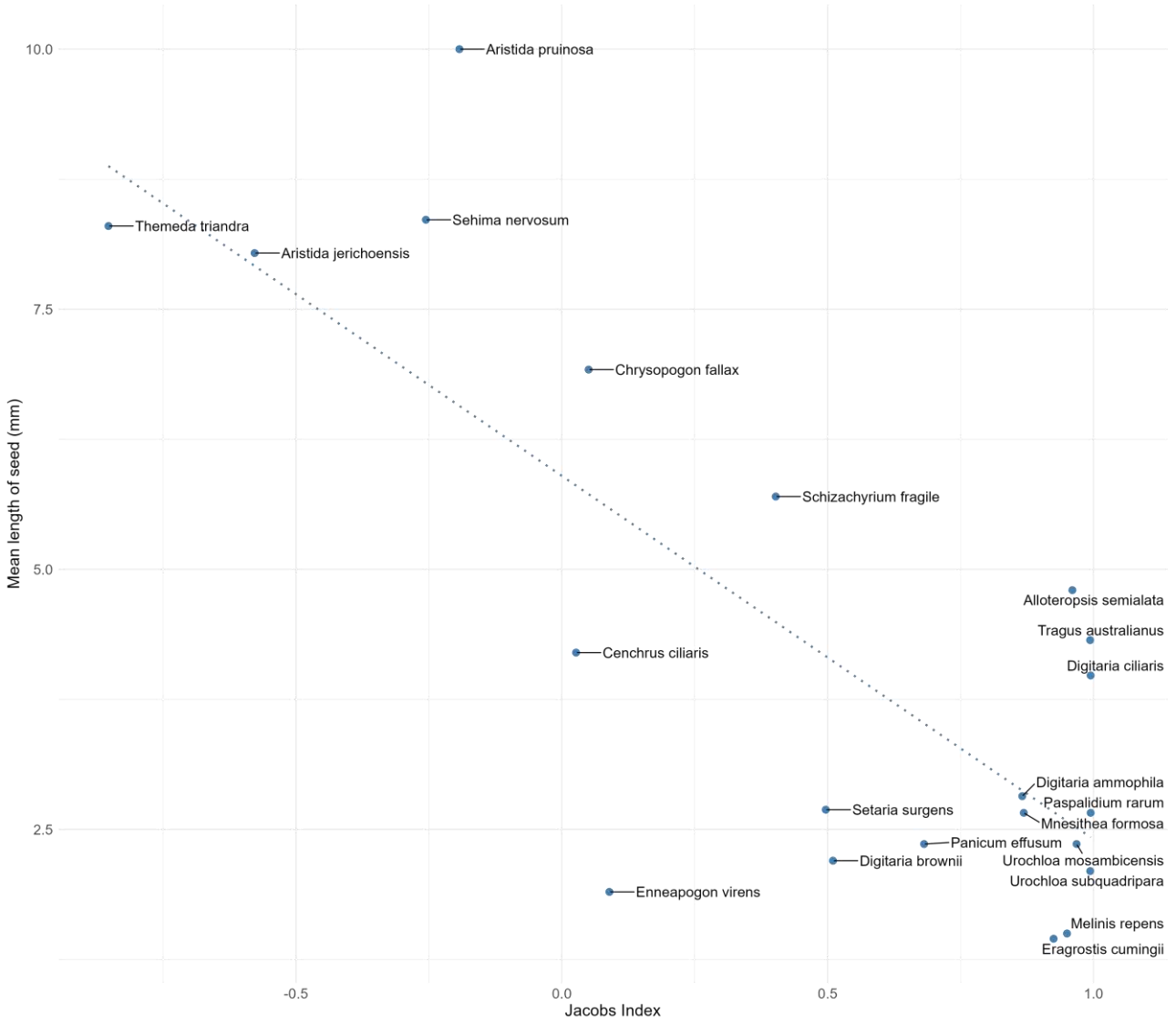


Figure 5.1-11: Relationship between Jacobs' Index and seed length content of the seeds of 20 grass species that occur in the study area.

5.1.5 Discussion

Our study found multiple lines of evidence that indicate that black-throated finch in the Desert Uplands demonstrate a strong selection preference for grass species that provide them with the greatest energy benefits. While observations of birds foraging and DNA analysis of their crop and faecal samples identified a broad range of plant taxa consumed by black-throated finch, including 34 grass species, only a relatively small number of these contributed to most of their diet, with six grass genera contributing 61% of the diet determined through DNA analysis and four grass species accounting for 56% of foraging observations.

Both the DNA analysis and direct foraging observations identified *Digitaria* spp. and *Panicum* spp. as two of the main sources of black-throated finch diet in the Desert Uplands. In the study area, three species of *Digitaria* have been recorded; *D. ciliaris*, *D. brownii* and *D. ammophila* while only *Panicum effusum* has been identified from that genus. The high Jacobs' Index for these grasses indicates that black-throated finch selectively target these species for consumption. While they are not rare grasses throughout the study area, *D. ciliaris*, *D. brownii* and *D. ammophila* generally occur in low densities and were never the

dominant grass species. This is consistent with foraging behaviour recorded during previous studies in Townsville that also identified a strong preference for a small number of grasses (Mitchell, 1996).

Conversely, the dominant grasses throughout the study area, namely *Heteropogon contortus*, *Themeda triandra* and *Aristida* spp. have a low Jacobs' Index, supporting the view that black-throated finch target specific grasses, rather than consume the most readily available. While these species were identified in the diet of black-throated finch through DNA analysis, they were rarely observed being foraged by birds during the wet season when seeds are often taken directly from the plant. Therefore, it appears that seeds from these grasses were likely to have been consumed from the ground and possibly whilst foraging in areas where more preferred grasses occur. Their proportion to black-throated finch diet, as well as the Jacobs' Index, was typically higher for these species during the dry season. This may be a consequence of the declining availability of grass seeds during this period which may force birds to consume more readily available species (Woinarski and Tidemann, 1991).

Seasonal variation in the diet of black-throated finch was evident from the DNA analysis of crop and faecal samples. While a substantial overlap in the grasses consumed in the wet and dry seasons was apparent, the relative proportion of these species to black-throated finch diet was significantly different. The consumption of very common, but less desirable grasses such as *Aristida* spp., *Heteropogon* spp., *Triodia* spp., *Themeda* spp. and *Cenchrus* spp. increased during the dry season and reduced for preferred grasses, *Digitaria* spp. and *Panicum* spp. While only a small sample size was recorded, the consumption of insects and arthropods was eight times higher in the dry season. This behaviour of diet is consistent with previous studies of black-throated finch in Townsville (Mitchell, 1996). Granivorous birds, including those in northern Australia, often exhibit these seasonal changes in diet in response to seed availability and nutritional requirements of grass seeds (Woinarski and Tidemann, 1991; Garnett and Crowley, 1994; Dostine and Franklin, 2002; Moorcroft *et al.*, 2002; Marone *et al.*, 2017).

We identified a clear positive correlation between the preferred forage species of black-throated finch and the levels of key nutritional factors provided by their seeds. In particular, metabolisable energy and crude protein were typically higher in grass species with a higher Jacob's Index. Metabolisable energy and crude protein are important for birds as they improve survival, growth, reproduction, egg development and provide energy (Pinowska, 1979; McWilliams, 2001; Honarmand, Goymann and Naguib, 2010; Maute, 2011; Pryke *et al.*, 2012). This is consistent with optimal foraging theory that suggests that animals aim to maximise energy gain relative to effort and consider the trade-off between factors including travel time, food type, density, nutritional value and predation risk (MacArthur and Pianka, 1966; Alonso *et al.*, 1995; Sauter *et al.*, 2006; King and Marshall, 2022). To optimise energy intake, granivores may choose the grass seeds with higher calories and nutrient and / or those that are easiest to process (Díaz, 1990, 1996; Dostine and Franklin, 2002; Luis Marone *et al.*, 2022). Grasses with higher amounts of protein and metabolisable energy are likely to be more profitable for black-throated finch, especially when their physical traits allow for efficient handling and processing (MacArthur and Pianka, 1966; Grubb and Greenwald, 1982; Nagy Koves Hrabar and Perrin, 2002; Soobramoney and Perrin, 2007; De León *et al.*, 2014). Therefore, by displaying a strong preference for these grass seeds black-throated finch are able to maximise their energy consumption to allow more time for other activities including breeding activities and traveling between water sources and foraging and breeding habitats.

As well as the nutritional properties of grass seeds, the morphological characteristics may also be important for granivorous birds (Díaz, 1996; Molokwu, Nilsson and Olsson, 2011; Ríos, Mangione and Marone, 2012; Titulaer *et al.*, 2018). This appears to be the case for black-throated finch who demonstrated a preference for grasses with shorter seeds, while abundant grass species such as *Themeda triandra*, *Aristida pruinosa* and *Sehima nervosum* with longer seeds were undesirable. Seed size is important for granivorous birds because it influences the handling time and therefore affects foraging efficiency and energy intake (Schluter, 1982; Diaz, 1990; Nagy Koves Hrabar and Perrin, 2002; Soobramoney and Perrin, 2007; De León *et al.*, 2014). The preference for smaller seeds, especially those

with higher nutritional values, allows smaller birds such as the black-throated finch to consume seeds quickly and increase their energy consumption.

Limitations in the DNA metabarcoding of the crop and faecal samples meant that a significant portion of the diet (33%) identified from this method could not be included in the analysis of the diet selection of black-throated finch in the Desert Uplands. Several taxa, including *Paspalidium*, *Dactyloctenium* & *Sehima*, that are known to be consumed by black-throated finch in the study area were therefore not able to be fully considered in the analysis of their diet. Future improvements in genetic databases and sequencing should allow for a more comprehensive analysis of black-throated finch diet, especially in populations that may benefit from similar studies to facilitate improved habitat management. DNA analysis of the diet of granivorous birds is a powerful tool as it allows researchers to overcome the limitations of visual observations, especially during the dry season when birds forage on fallen grass seeds which cannot be confidently identified. Visual observations are also more common along access tracks and in disturbed areas that are generally easier for an observer to access and record SBTF in. This inherent limitation may have led to the non-native species *Digitaria ciliaris* being identified as a highly preferred foraging species when the species is simply over-represented in visual observations due to observer bias. DNA analysis to species level could provide an unbiased line of evidence to determine whether this species is preferentially selected across the landscape, or simply where visual observations have been undertaken.

5.1.6 Conclusion

This study identifies the dietary selection preferences of black-throated finch in the Desert Uplands bioregion. We found that black-throated finch strongly prefer a relatively small number of grass taxa that are characterised as having higher levels of metabolisable energy and crude protein with a smaller seed length. By preferentially selecting these grasses, they are able to optimise their energy consumption by reducing handling time and maximising energy consumption. This allows birds to allocate more time to other activities such as breeding, drinking and avoiding predators. As the availability of grass seeds decline during the dry season, they broaden their diet and increase consumption of more readily available, but less desirable, grasses.

These findings provide valuable insights into the diet of this species and reinforce the importance of maintaining and managing habitats to promote a diversity of native grasses and especially those that are more profitable to support the conservation of the black-throated finch.

5.2 Grass seed phenology

5.2.1 Abstract

Seeding phenology is a key driver of food availability for granivorous birds, particularly in highly seasonal environments like tropical savannas. This study examines the seeding phenology of grasses within a 75,000 ha savanna woodland in the Desert Uplands of north-eastern Australia, a stronghold for the nationally Endangered southern black-throated finch (*Poephila cincta cincta*). Phenological data were collected across nine survey periods from February 2022 to August 2024, spanning 220 vegetation monitoring transects. We assessed seed production patterns in 31 grass groups and classified them into four broad seeding strategies: early-seeding perennials, late-seeding perennials, annuals, and a mast-seeding species. While seed availability peaked in the late wet season and declined through the dry season, the staggered seed production of the species-rich grass layer extended the period of seed availability. Our findings support the hypothesis that the late dry and early wet seasons (following the onset of rainfall), represent a critical food bottleneck for granivorous birds in tropical savannas. However, we also highlight the importance of grass species richness and diverse disturbance histories through fire, in staggering seed production and supporting year-round foraging by the southern black-throated finch.

5.2.2 Introduction

Phenology, the study of the timing of recurring biological events, is crucial to understanding how species interact with their environment and adapt to changing conditions (Lieth, 1974; Visser and Gienapp, 2019). In plants, phenological patterns, such as the timing of flowering, fruiting, and seed production, are influenced by various abiotic and biotic factors, including temperature, rainfall, and competition (Pilar and Gabriel, 1998). These phenological patterns not only affect the survival and reproduction of plants but also have significant implications for other organisms, such as granivorous birds, that depend on plant resources for their sustenance (Rathcke and Lacey, 1985).

Granivorous birds, which rely on grass seeds as a primary food source, are particularly sensitive to the phenological dynamics of grasses. The timing of seed production, seed set, and seed availability can determine the survival and reproductive success of these bird species (Crowley, Garnett and Shephard, 2003; Siriwardena, Calbrade and Vickery, 2008). A mismatch between seed availability and bird breeding periods can have significant ecological consequences, affecting population dynamics and ultimately the viability of granivorous bird species in a given area (Marone *et al.*, 2017; Wann *et al.*, 2019).

Seeding phenology also plays an important role in guiding land management actions, particularly in contexts aimed at modifying species composition to achieve conservation gain. For example, optimisation of fire regimes can support populations of threatened granivorous birds such as golden-shouldered parrot and Gouldian finch through fire's effects on seed availability (Crowley and Garnett, 1999; Collett *et al.*, 2023).

In this study, we examine the seeding phenology of grasses in a 75,000 ha study area of savanna woodland in the Desert Uplands of north-eastern Australia. The study area supports a stronghold population of the nationally Endangered granivore, the southern black-throated finch. A broader goal of the research is to support management actions to conserve the species through an understanding of key grasses that contribute to the species' diet.

5.2.3 Methods

5.2.3.1 Phenology data

We collected phenology data at ground cover monitoring sites within the study area between February 2022 and August 2024. The phenology monitoring included 220 unique vegetation monitoring transects that were included in repeat monitoring events across nine survey periods between February and Table 5.2-1. Phenology data were collected at an average of 60 sites each survey (range = 19–109 sites).

The timing of this study coincided with a period of average to above average rainfall, with annual averages of 860 mm, 543 mm and 696 mm of rainfall being recorded in 2022, 2023 and 2024, respectively, compared to an annual average of 544 mm (Bureau of Meteorology, 2025). Furthermore, a substantial "dry-season" rainfall events occurred between May and August each year, which were 2–5 times above the average dry season rainfall (Figure 5.2-1; Bureau of Meteorology, 2024).

Each vegetation monitoring site consisted of either five (n = 38 replicates) or 10 quadrats (n = 502 replicates) that were evenly distributed along a 100 m transect. Within each quadrat, the cover of each grass species was recorded and the proportion of individuals of each species holding mature seed on the inflorescence was assessed.

Table 5.2-1: Number of vegetation monitoring sites at which phenology data were collected

Survey	Number of sites
2022 early wet season (February–March)	72
2022 wet season (June)	19
2022 dry season (October)	48
2023 early wet season (February–March)	45
2023 wet season (May–June)	109
2023 dry season (October)	76
2024 early wet season (February)	49
2024 wet season (April)	81
2024 dry season (August)	41

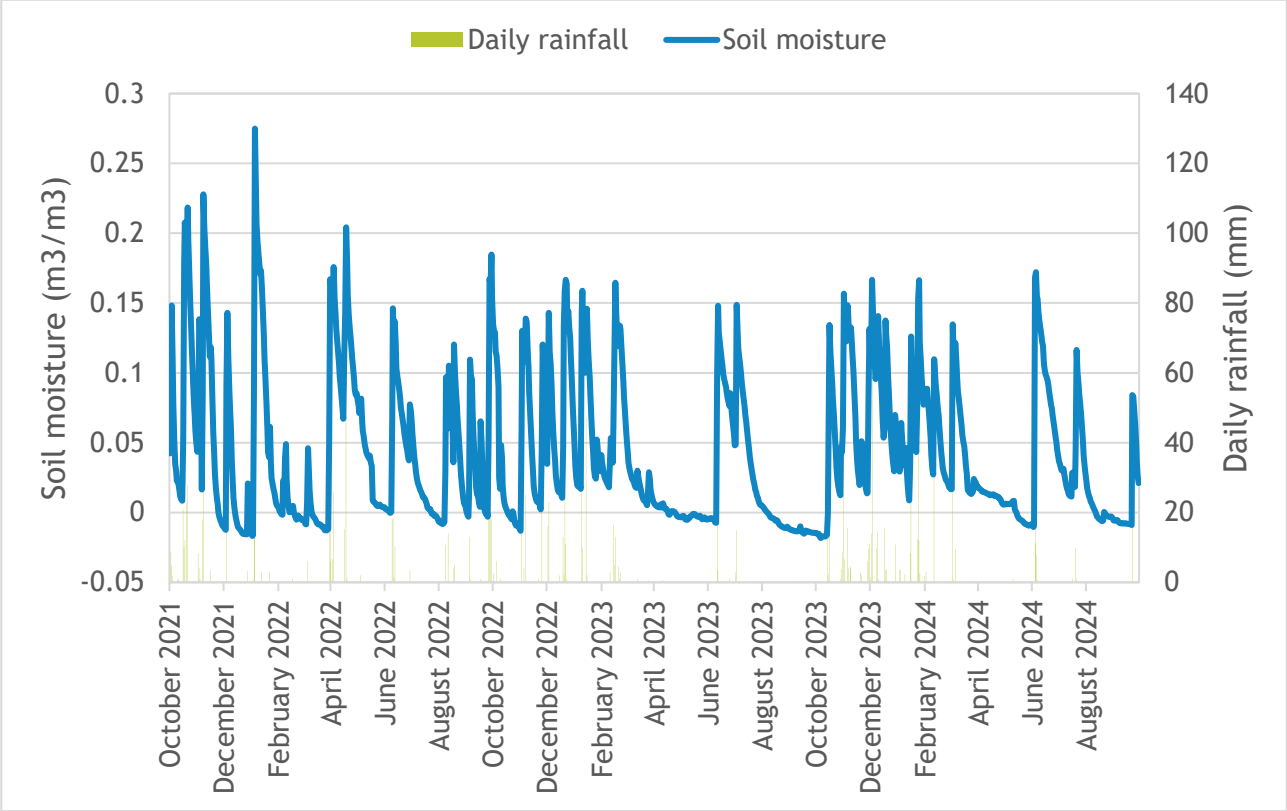


Figure 5.2-1: Daily rainfall (green) and soil moisture (blue) recorded within the study area.

5.2.3.2 Statistical analyses

While quadrat data were generally collected and analysed to a species level, we consolidated data from some species into broader species groups. This was done for closely related species that share life history characteristics, that either had uncertainty in their identification or had sample sizes too small to afford a robust annual trend of seed phenology. Species groupings included:

- *Enneapogon* spp., which comprised *E. lindleyanus*, *E. polyphyllus*, *E. robustissimus* and *E. virens*;
- *Eragrostis* annual spp., which comprised *Eragrostis lacunaria* and *Eragrostis tenellula*
- *Eragrostis* perennial spp., which comprised *Eragrostis australasica*, *Eragrostis cilianensis*, *Eragrostis cumingii*, *Eragrostis elongata*, *Eragrostis macrocarpa*, *Eragrostis schultzii*, *Eragrostis sororia*, *Eragrostis speciosa*
- *Aristida* perennial spp., which comprised *Aristida benthamii*, *Aristida calycina*, *Aristida inaequiglumis*, *Aristida ingrata*, *Aristida latifolia*, *Aristida pruinosa*, *Aristida ramosa*
- *Cymbopogon* spp., which comprised *Cymbopogon bombycinus*, *Cymbopogon obtectus*, *Cymbopogon refractus*
- *Sporobolus* annual spp., which comprised *Sporobolus actinocladus*, *Sporobolus australasicus*, *Sporobolus caroli*
- *Aristida* annual spp., which comprised *Aristida contorta*, *Aristida holathera*, *Aristida hygrometrica*, *Aristida jerichoensis*
- *Urochloa* annual spp., which comprised *Urochloa piligera*, *Urochloa subquadripara*

We then calculated a seed availability index by taking the average cover of each species group among the quadrats within each site and dividing by that by the average proportion of individuals of that species that had mature seed on their inflorescences. We then used loess regressions to examine the relationship between seed availability and day of the year (days since 1 January) for each species group. Species groups with fewer than 10 records in quadrats were removed from the analysis due to a lack of sample size to show phenological trends.

5.2.4 Results

Seed availability among all grass species was highest in the late wet season (mean seed availability index [SAI] = 0.165, 95% CI = 0.129–0.202), declined into the dry season (mean SAI = 0.140, 95% CI = 0.108–0.177) and was lowest in the early wet season (mean SAI = 0.0947, 95% CI = 0.0192–0.0.178). Dominant grasses that contributed to seed availability on inflorescences included *Aristida* perennial spp. (13.8%), *Heteropogon contortus* (9.88%), *Aristida* annual spp. (9.57%), *Enneapogon* spp. (8.03%), *Panicum effusum* (7.33%), *Cenchrus ciliaris* (6.88%), *Themeda triandra* (6.53%) and *Sehima nervosum* (5.58%).

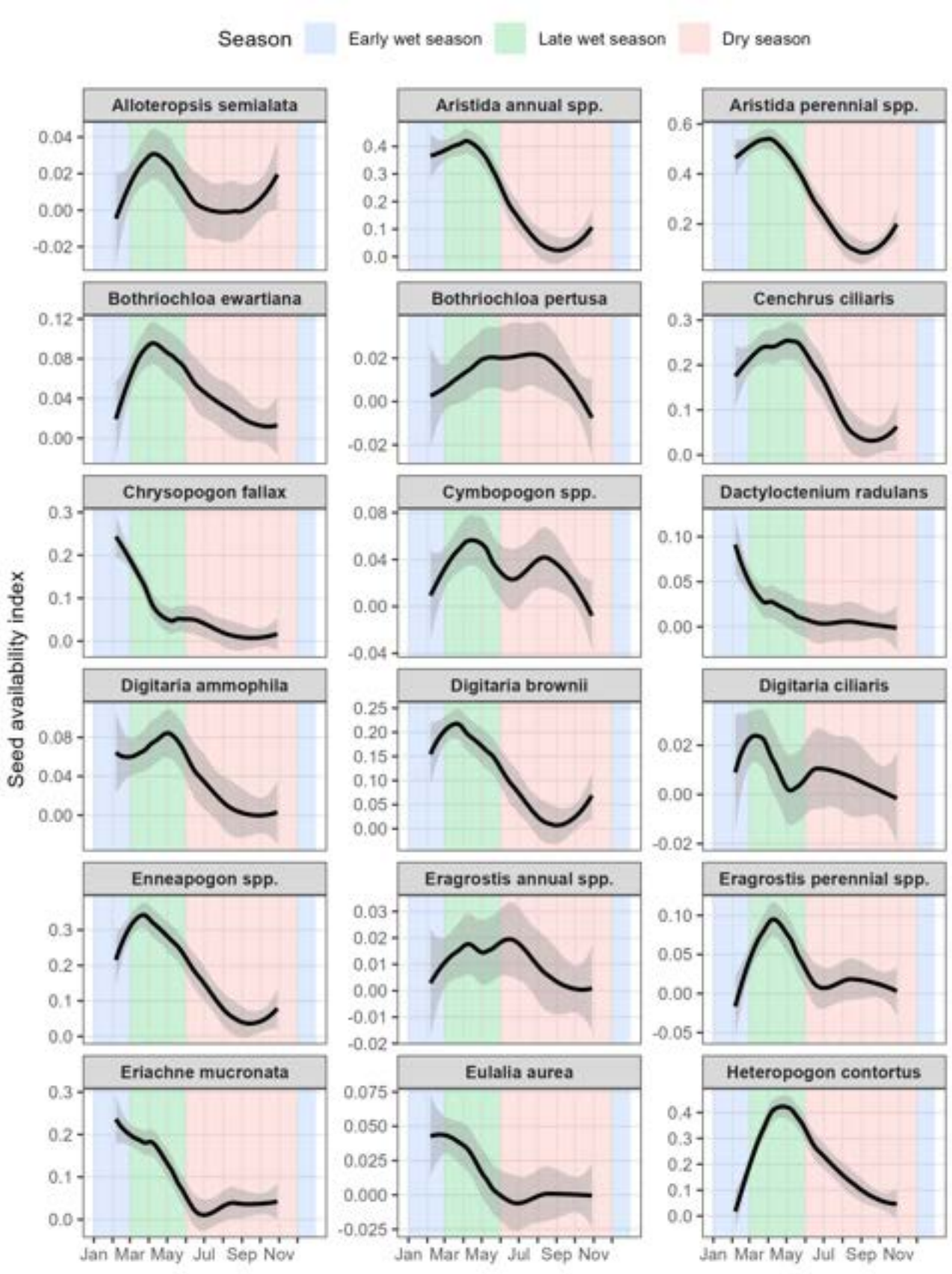
While most grasses followed a similar annual phenological trend, with seed availability peaking in the late wet season and declining into the dry season (Figure 5.2-2), species groups could be broadly divided into: 1) early seeding grasses, which had peak seed availability between January and February 2) late seeding grasses, which had peak seed availability between March and August; and 3) disturbance specialists, whose seeding was inconsistently distributed throughout the year, sometimes with multiple peaks of seed availability (Table 5.2-2).

Table 5.2-2: Summary of seeding phenology of grass genera within the study area

Genus	Number of records	% FPC	Time of peak seed availability	Total seed availability index
Early seeding perennials				
Alloteropsis semialata	42	0.23	99*	2.81
Chrysopogon fallax	1208	6.59	37	16.8
Themeda triandra	1604	8.75	37	38.3
Eriachne mucronata	896	4.89	37	23.4
Late seeding perennials				
Bothriochloa pertusa	34	0.19	205	3.71
Digitaria brownii	846	4.62	81	27.7
Enneapogon spp.	1039	5.67	82	47.1
Sehima nervosum	722	3.94	82	32.8
Eulalia aurea	38	0.21	52	3.11
Eragrostis perennial spp.	92	0.50	99	8.80
Aristida perennial spp.	2574	14.1	92	81.3
Bothriochloa ewartiana	817	4.46	98	13.2

Genus	Number of records	% FPC	Time of peak seed availability	Total seed availability index
Heteropogon contortus	1993	10.9	115	58.0
Cenchrus ciliaris	1139	6.22	124	40.4
Digitaria ammophila	107	0.58	122	10.5
Panicum effusum	683	3.73	122	43.0
Urochloa mosambicensis	22	0.12	95	1.14
Cymbopogon spp.	74	0.40	105	8.77
Annuals				
Dactyloctenium radulans	37	0.28	37	4.39
Digitaria ciliaris	70	0.22	69	2.41
Eragrostis annual spp.	166	0.23	171	2.84
Tragus australianus	92	0.07	93	2.75
Paspalidium rarum	98	0.37	37	4.82
Perotis rara	87	0.81	85	9.11
Schizachyrium fragile	99	2.08	101	14.2
Sporobolus annual spp.	93	0.58	86	4.19
Mnesithea formosa	98	0.66	99	7.33
Setaria surgens	98	0.90	97	7.95
Aristida annual spp.	97	6.83	37	56.1
Urochloa annual spp.	98	0.08	37	2.78
Mast seeding perennial				
Triodia pungens	1989	10.85	37	7.00

* *Alloteropsis semialata* responds rapidly to rainfall, producing seed within six weeks (Crowley and Garnett, 2001; Crowley, 2008). The April seed recorded in our study was likely to be secondary to an earlier peak in December–January, which were not monitored within our study.


Season
Early wet season
Late wet season
Dry season
Seed availability index
Alloteropsis semialata
Aristida annual spp.
Aristida perennial spp.
Bothriochloa ewartiana
Bothriochloa pertusa
Cenchrus ciliaris
Chrysopogon fallax
Cymbopogon spp.
Dactyloctenium radulans
Digitaria ammophila
Digitaria brownii
Digitaria ciliaris
Enneapogon spp.
Eragrostis annual spp.
Eragrostis perennial spp.
Eriachne mucronata
Eulalia aurea
Heteropogon contortus
Jan Mar May Jul Sep Nov

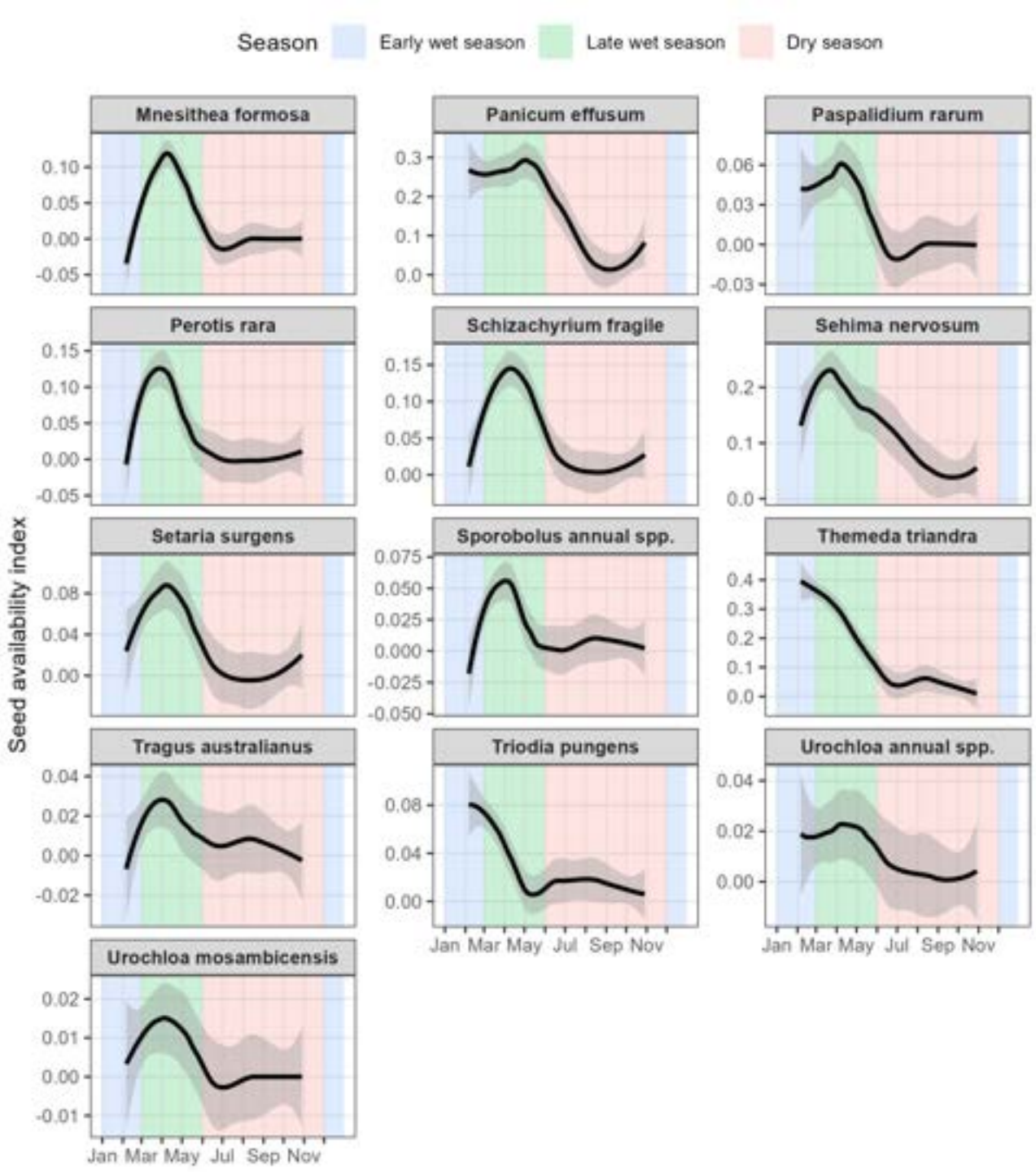


Figure 5.2-2: Mean seed availability held on inflorescences throughout an average climatic year (black line), depicted using a LOESS regression with 95% confidence intervals (shaded region).

5.2.5 Discussion

5.2.5.1 Phenological patterns

This study investigated the seeding phenology of grasses in the Desert Uplands region of Queensland, with a particular focus on species contributing to the diet of the southern black-throated finch. Our findings highlight the diverse seed phenology strategies exhibited by the high grass species richness in the open grassy woodlands where this study was conducted.

In tropical savannas, where temperature rarely limits grass growth, seasonal rainfall is the primary factor influencing phenology (House and Hall, 2001). Despite this, grasses in these ecosystems display a variety of seeding strategies. While vegetative growth and flowering generally occur throughout the wet season, grass communities consist of species that initiate flowering and seed maturation at different times through the year. Strategies are numerous and are likely to relate to competition for resources and minimisation of seed predation by granivores (Wright and Fensham, 2018; Preston and Fjellheim, 2020; Verrico and Preston, 2024). Based on our findings, we categorised grasses into four broad seed phenology strategies: late-seeding perennials, early-seeding perennials, annuals, and a mast-seeding species (*Triodia pungens*), each varying in the timing of seed production and its primary environmental drivers.

Early-seeding perennial grasses had the highest seed availability on their inflorescences in the early wet season, corresponding to December to February. These species prioritise reproduction over vegetative growth, sacrificing leaf biomass and photosynthetic potential to take advantage of optimal conditions—high soil moisture, abundant sunlight, and warm temperatures, while avoiding later-season stressors such as declining soil moisture, herbivory, and shading by larger, late-seeding grasses (Verrico and Preston, 2024). *Alloteropsis semialata* and *Chrysopogon fallax* are early-seeding perennial grass species that are important food sources for granivores in northern Australia (McIvor *et al.*, 1996; Crowley and Garnett, 2001; Crowley, 2008). Both *Alloteropsis semialata* and *Chrysopogon fallax* store energy reserves in their roots during the dry season and use these reserves to initiate seeding quickly following the onset of wet season rainfall (McIvor *et al.*, 1996; Crowley and Garnett, 2001; Crowley, 2008). *Alloteropsis semialata* can set seed within six weeks of rainfall and has no dormancy period (Crowley and Garnett, 2001; Crowley, 2008). Given this rapid response to rainfall, the peak seed availability we recorded in April was likely a secondary peak, following an initial peak in December–January, for which we lacked phenological data. Similar dual seeding peaks were also recorded by Williams (2002) for *Alloteropsis semialata* and Brennan (1996) for *Chrysopogon* fallax.

Themeda triandra is a common species throughout northern and eastern Australia (Arthan *et al.*, 2022), and within our study area it contributed to 6.53% of the total seed availability index. Tothill (1977) identify substantial phenological variation among *Themeda triandra*, which may show both early- and late-flowering characteristics. We found that *Themeda triandra* contributed a substantial proportion of early wet season seed availability, with the species continuing to have seeds available on the inflorescences extending into the late wet season, which is consistent with the seeding phenology recorded by Williams (2002).

Late-seeding perennial grasses exhibited peak seed availability from March to June, corresponding to the late wet season and early dry season (Figure 5.2-2). Late seeding perennial grasses accumulate biomass early in the wet season and delay their seed production to the late wet season, potentially resulting in a greater yield of seeds, outcompeting of lower biomass early-seeding perennial and annual grasses for space and resources but with the requirement to tolerate stress from late season drought (Verrico and Preston, 2024). As a result, late-seeding perennial grasses generally have a higher biomass than early-seeding perennial and annual grasses and produce seed once per year (obligate seeding) (Friedman, 2020). Phenological strategies of these late-seeding perennial grasses were variable, with differing timing of seed

set, tendencies to hold seed on the grass inflorescences following dehiscence, sequential seed maturation and delayed seed production to the dry season (Sarmiento, 1992).

Large perennial grasses, including *Heteropogon contortus* and dominant *Aristida* species (*A. calycina* and *A. inaequiglumis*), retained mature seed on their inflorescences for extended periods, persisting into the dry season even after leaf dehiscence. This strategy was particularly pronounced in *H. contortus*, which forms seed bundles that intertwine their awns, allowing seeds to remain among aboveground leaf matter and stems well into the dry season (Groves and Whalley, 2002; Williams, 2002).

Sequential seed maturation, where seeds on an inflorescence mature gradually rather than simultaneously, extends the period that seeds are available to dispersal and germination (Sripathy and Groot, 2023). This strategy may allow species a wider window to achieve germination under varying environmental conditions and also allows the plant to halt seed-filling when conditions worsen (Sripathy and Groot, 2023). *Enneapogon* species, particularly *E. virens*, and *Sehima nervosum* exhibited sequential seed maturation within the study area (Authors pers. obs.), which resulted in the extended period of seed availability observed for these species..

Other species, such as *Cymbopogon bombycinus*, displayed delayed seed set, with peak seed availability occurring in the mid-dry season. This phenological pattern has also been documented by Williams (2002), though the ecological drivers behind this strategy remain unclear.

Annual grasses complete their life cycle within a single growing season, which includes germination, flowering and seed production before dying (Friedman, 2020). Annual grasses must prioritise seed production over vegetative growth and survive the dry season through a soil seed bank (Verrico and Preston, 2024). Common traits of annual grasses are rapid growth, early flowering, and high reproductive effort (Friedman, 2020). Because of this, annual grasses favour environments with high levels of disturbance and unpredictable conditions (Friedman, 2020). Annual grasses may also follow a determinant flowering schedule, flowering a consistent time of year, or an indeterminate schedule, flowering when environmental conditions are conducive (Friedman, 2020). Our results show a wide variation in annual grass seed timing, which reflects the diversity of strategies annual grasses adopt. Long-term grazing trials in the Charters Towers region have shown that *Schizachyrium fragile*, *Dactyloctenium radulans*, *Paspalidium rarum*, and annual *Eragrostis* species are disturbance specialists that rapidly respond to environmental opportunities such as bare ground exposure, fire, or episodic rainfall (O'Reagain *et al.*, 2018). The variation in annual grass seeding is also demonstrated by some annual species, such as *Digitaria ciliaris*, *Tragus australianus*, *Eragrostis* and *Sporobolus* species, having multiple peaks in seed availability, which may reflect these species being indeterminant flowering species, which have multiple seeding peaks, or opportunistic germination, growth and seeding in response to favourable conditions (Groves and Whalley, 2002; Friedman, 2020).

Mast seeding grasses produce irregular but substantial pulses of seed production in response to environmental cues (Wright, Franklin and Fensham, 2022). *Triodia pungens* was the only mast-seeding grass recorded in the study area, which has seeding events triggered by consecutive years of high rainfall (Wright and Fensham, 2018). Seeding typically falls early in the year, which aligns with our observations of peak seed availability in February (Wright and Fensham, 2018). The episodic abundance of seeds during mast seeding events can overwhelm seed predators, reducing the proportion of seeds consumed and increasing the likelihood that some seeds will persist in the soil seed bank until favourable germination conditions arise (Bogdziewicz, Zwolak and Crone, 2016). By concentrating seed production in specific years, plants may also support granivore populations during resource-scarce periods (Bogdziewicz, Zwolak and Crone, 2016).

5.2.5.2 Relevance to the southern black-throated finch diet

The southern black-throated finch, as a granivorous bird, is highly dependent on the phenological patterns of grasses for food availability (Mula Laguna *et al.*, 2019). Temporal variation in seed production among different grass species is likely to play an important role in sustaining this species throughout the year (Mula Laguna *et al.*, 2019). Early-seeding perennials such as *Alloteropsis semialata*, *Chrysopogon fallax* and *Themeda triandra* provide a crucial food source at the onset of the wet season, when other resources may be scarce (McIvor *et al.*, 1996; Crowley and Garnett, 2001; Crowley, 2008). In contrast, late-seeding perennials like *Heteropogon contortus* and *Aristida* spp. extend seed availability into the dry season, potentially supporting the finch during periods of lower overall seed abundance (Section 5.1). Annual grasses and disturbance-adapted species, such as *Paspalidium rarum*, *Schizachyrium fragile* and *Digitaria ciliaris*, may provide additional food resources, particularly in response to episodic rainfall or land management interventions such as fire or grazing (O'Reagain *et al.*, 2018). However, our results indicate that seed availability is lowest in the late wet season and following the onset of early wet season rainfall, which supports suggestions that this period represents a critical food bottleneck for granivorous birds in the tropical savannas (Garnett and Crowley, 1995; Crowley, 2008). Land management measures should prioritise maintaining grassy woodlands with high grass species richness and a mosaic of land management histories, particularly fire history to stagger the timing of seed production and promote seed availability throughout the year.

6 Aim 4: Identify management strategies for SBTF habitat in the Project Area regarding fire, grazing and water

6.1 Relative impacts of grazing and fire on habitat quality for black-throated finch

6.1.1 Abstract

Savanna ecosystems are globally threatened by land use change, altered fire regimes and invasive species. Northern Australia's savannas, although relatively intact, face increasing pressures from pastoralism and fire exclusion. This study evaluates the effects of cattle grazing and fire on ground layer vegetation composition and species richness across 75,000 ha of savanna woodland in central Queensland. Over four years, 900 vegetation surveys were conducted across 239 sites, stratified by fire history and stocking rates. Bayesian regression models revealed that higher stocking rates were associated with increased cover and richness of non-native graminoids, particularly *Cenchrus ciliaris* (buffel grass), and reduced cover of native perennial grasses such as *Themeda triandra* (kangaroo grass) and *Heteropogon contortus* (black spear grass). In contrast, recent burning promoted native annual graminoids and forbs, while long-unburnt sites favoured perennial and exotic species. The study also found that Stylosanthes (stylo) species, a dominant non-native forb, responded positively to fire and negatively to grazing, highlighting the need for integrated control strategies. Additionally, grazing-sensitive native grasses were shown to decline under even low stocking rates. Controlled low-intensity burns, led by First Nations rangers, were effective in enhancing native diversity and suppressing invasive species. The study highlights that grazing and fire regimes must be jointly managed to maintain ecological integrity. Low stocking rates combined with mosaic burning can mitigate the dominance of invasive species and support a native species rich ground layer. These findings provide empirical support for integrated fire and grazing management strategies to improve the conservation value of Australian savanna woodlands.

6.1.2 Introduction

The world's savannas have been extensively cleared and degraded (Purvis *et al.*, 2019). By 2000, more than 80% of savannas had been transformed into areas dominated by human activities, including 52% converted to rangelands and 25.8% to croplands (Ellis *et al.*, 2010). The savannas of northern Australia, covering approximately 1.9 million km^2 (Russell-Smith and Yates, 2007), are the largest and least impacted savannas on earth (Murphy, Andersen and Parr, 2016). However, despite these ecosystems being less than 1% cleared (Murphy et al., 2016), there is still evidence of widespread threats to savanna habitats through increased grazing pressure, altered fire regimes, woody vegetation thickening and the spread of exotic species (O'Connor and Pickett, 1992; Franklin *et al.*, 2005; Kutt and Woinarski, 2007; Legge *et al.*, 2019).

Fire regimes play a powerful role in affecting the composition and diversity of native vegetation communities in the Australian savanna, particularly when combined with the effects of livestock grazing . The spread of pastoralism and associated land management changes has been linked to a large-scale decline in biodiversity within Australian savanna, including many threatened fauna and flora (Woinarski, Legge, *et al.*, 2011; Silcock and Fensham, 2018; Mula Laguna *et al.*, 2019). Prior to European colonisation, Indigenous burning practices consisted of regular small fires, lit throughout the year, to create a fine scale patchwork of fire histories, including long-unburnt areas with fire sensitive vegetation types (Yibarbuk *et*

al., 2001). This fire regime promotes native species diversity primarily through the disturbance of ground layers dominated by high biomass native perennial grasses, which allows less competitive species to access resources such as space and light (Russell-Smith, Start and Woinarski, 2001; Lebbink, Fensham and Cowley, 2018). Without these regular low intensity fires to control fuel loads, the fire regime within northern Australia has shifted to be dominated by high intensity, often uncontrolled, late dry season fires which burn through large areas of the landscape (Russell-Smith, Start and Woinarski, 2001; Russell-Smith and Edwards, 2006; Murphy and Russell-Smith, 2010). In addition to the significant economic risk, high intensity fires can also impact the structure and diversity of vegetation communities through the destruction of seed banks, increase in non-native cover and thickening of the woody shrubs (Williams *et al.*, 1999; Liedloff *et al.*, 2001; Yates, Edwards and Russell-Smith, 2008; Miller *et al.*, 2010; Wright and Fensham, 2016). The balance of available evidence suggests that a well-planned fire regime, conducted to avoid or actively prevent wildfires, is an important management tool for improving land condition (Murphy and Russell-Smith, 2010; Cowley *et al.*, 2014; Lebbink, Fensham and Cowley, 2018).

The spread of pastoralism has caused a shift towards the use of livestock grazing to reduce fuel loads in place of regular burns (Lebbink, Dwyer and Fensham, 2021). In addition to reducing fuel loads, livestock grazing also provides a disturbance within the landscape which, in some circumstances, can promote species diversity within the ground layer (James, Landsberg and Morton, 1999; Lunt *et al.*, 2007; Fensham and Fairfax, 2008). However, the Australian savanna is adapted to disturbance by fire whereas grazing by large, hooved mammals has been recently introduced to the landscape (Lunt *et al.*, 2007). Due to this, grazing at even low intensities within Australian landscapes may have negative effects on ecosystem structure, function and composition (Eldridge *et al.*, 2016). Although livestock grazing may provide a disturbance, it does not fill the same ecological role that fire does within the landscape (Liedloff *et al.*, 2001). Many fire-adapted species require fire or conditions created by fire such as heat, smoke or edaphic conditions to germinate (Lunt *et al.*, 2007; Lewandrowski *et al.*, 2018; Pausas and Lamont, 2022). Fire also plays an important role in maintaining vegetation community structure and the exclusion of fire due to overgrazing can lead to the thickening of woody shrubs, which in turn can suppress the grassy understorey (Liedloff *et al.*, 2001; Williams, Woinarski and Andersen, 2003). The contrasting effects of fire and grazing are particularly apparent in the ground-layer of savanna woodlands, with grass species composition being substantially altered depending on the dominant disturbance regime (Liedloff *et al.*, 2001; Archibald, Hempson and Lehmann, 2019). Fire-resistant species are often not grazing-resistant, as fire-resistant traits such as flammability and vertical growth differ from the lateral growth and below-ground protection typical of grazing adapted species (Archibald, Hempson and Lehmann, 2019). Consequently, replacing fire with grazing can shift species composition toward more grazing tolerant species, altering ground layer composition (Archibald, Hempson and Lehmann, 2019).

However, livestock grazing and fire regimes are not only inherently linked (Liedloff *et al.*, 2001) but in northern Australia they can overlap spatially and often interact, as up to 90% of land is grazed for beef cattle (Ross, 2009) and an average of approximately 20% of land is burnt annually (Russell-Smith and Yates, 2007). Graziers often undertake low-intensity controlled burns in the late wet season to manage biomass and produce new growth for livestock grazing (Russell-Smith, Start and Woinarski, 2001). However, the selective grazing of fresh shoots after burns can negate any benefits of burning for species diversity (Archibald *et al.*, 2005; Cowley *et al.*, 2014; Legge *et al.*, 2019).

The spread of pastoralism through Australian savannas has also led to the introduction of pasture species such as *Cenchrus ciliaris* (buffel grass) due to its resistance to drought and heavy grazing (Marshall, Lewis and Ostendorf, 2012). However, buffel grass is a key threat to savanna woodlands in northern Australia due to its ability to outcompete native species (Eyre *et al.*, 2009; Fensham *et al.*, 2013), with potential impacts from the species potentially being equivalent to those posed by invasive animals (Read *et al.*, 2020). In addition to competitive effects, invasion by vigorous naturalised grasses can significantly increase fuel loads and therefore increase potential fire intensity and frequency, which in turn may

impact negatively on many native species (Miller *et al.*, 2010; Schlesinger, White and Muldoon, 2013; Ryan-Colton *et al.*, 2024). Buffel grass is capable of rapid recovery from the high intensity fires it fuels, and derives a competitive advantage in the post fire community, thereby facilitating their continues dominance in these systems (Miller *et al.*, 2010; Ryan-Colton *et al.*, 2024). Areas with a history of heavy grazing may have already lost grazing sensitive species so there would be little difference between the use of livestock grazing and burning to promote diversity and burning may even promote the spread of invasive species (Lunt *et al.*, 2007).

Despite the synergistic effects of fire and grazing on savanna ecosystems, previous research has identified the lack of quantitative information as a key gap in the understanding of these interconnected processes in many Australian ecosystems (Woinarski and Legge, 2013; Fensham, Wang and Kilgour, 2015; Lebbink, Dwyer and Fensham, 2021; Radford *et al.*, 2024). Our study addresses this by analysing four years of monitoring data from a conservation program designed to enhance the ecological health and conservation value of 75,000 hectares of savanna woodland. Through intensive monitoring, we aim to quantify the relative impacts of fire and cattle stocking rates on ground layer diversity and composition within an Australian savanna.

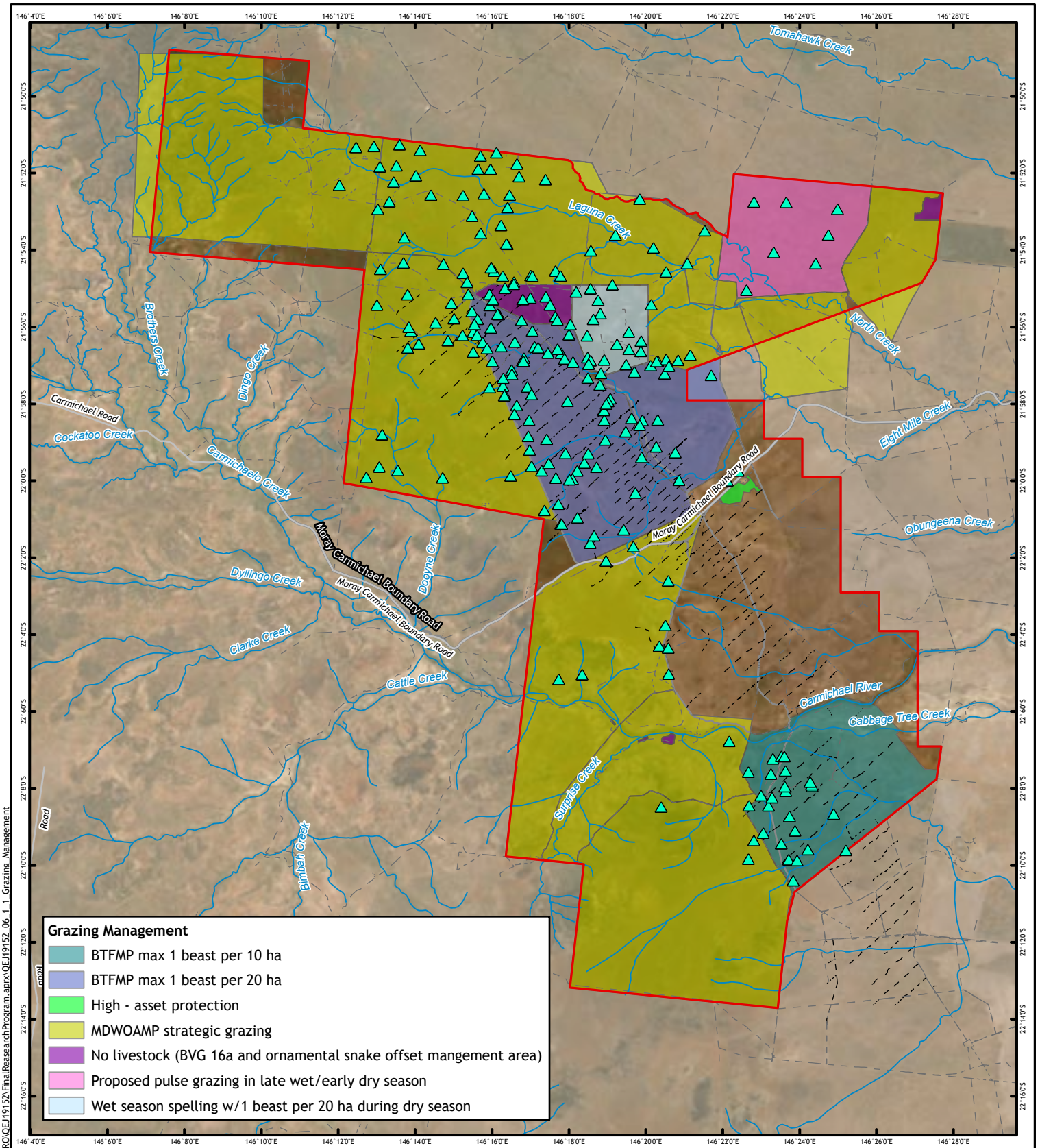
6.1.3 Methods

6.1.3.1 Study area

This study was conducted in a 75,000 ha section of the Moray Downs property, situated within the Desert Uplands Bioregion of Queensland, Australia (Figure 6.1-1). Much of the Desert Uplands Bioregion has a history of low-intensity cattle grazing since European settlement of the area in the mid-1800s (Paterson, 2005). This historically low grazing intensity can be attributed to the large property sizes with expansive paddocks, limited permanent water sources that create areas inaccessible to livestock, and the presence of *Gastrolobium grandiflorum* (heart-leaf poison bush), a toxic plant that has discouraged intensive grazing (Kutt and Woinarski, 2007).

Land use within the study area comprises a combination of cattle grazing and native vegetation managed as an environmental offset under the National *Environment Protection and Biodiversity Conservation Act 1999*. Stocking rates during the study period were generally low-intensity, with an average of 1 beast per 36 ha and ranging between no stock to a maximum of 1 beast per 11 ha. The study area includes a portion of the Carmichael Coal Mine mining lease, with the nearest vegetation monitoring site 2 km from mining infrastructure (Figure 6.1-1).

The study area is within the tropical savanna biome of northern Australia, which is broadly characterised by long annual dry seasons with summer dominated rainfall (Fox *et al.*, 2001; Woinarski, Andersen and Murphy, 2020). Uncleared remnant vegetation covers 84% of the study area, with vegetation communities largely comprising eucalypt woodlands with a grassy ground layer. These woodlands are dominated by *Eucalyptus melanophloia*, *E. similis* and *E. brownii* that occur on deep red earths (Department of Resources, 2024). Rainfall within the study area averages 541 mm per year, with the majority (~65%) of this falling within the wet season, which is typically from December to March (Bureau of Meteorology, 2023, station number 036050, 25 km from the study area).The majority of vegetation within the study area is fire-adapted eucalypt dominated woodlands (Burrows *et al.*, 2010), which are recommended to be burnt every 5 to 10 years (Queensland Herbarium, 2024).



Legend

- Vegetation Monitoring Sites
- Major Road
- Minor Road
- Exploration Track
- Track
- Watercourse
- Project Area

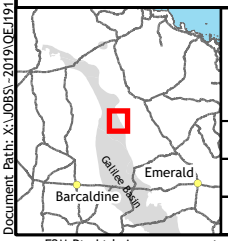
N

Scale 1:250,000 (A4)

0 5 10

Kilometers

Coordinate System: GDA2020



Notes:
Aerial Imagery: © ESRI 2021
Cadastre: © DNRME 2024
Watercourse: © Geoscience Australia 2012
Road: © DNRME 2024

Rev	Description	Drawn	Approved	Date
2	Issued for Use	JY	JMC	30/10/2025
0	Issued for Use	GO	JMC	08/08/2025

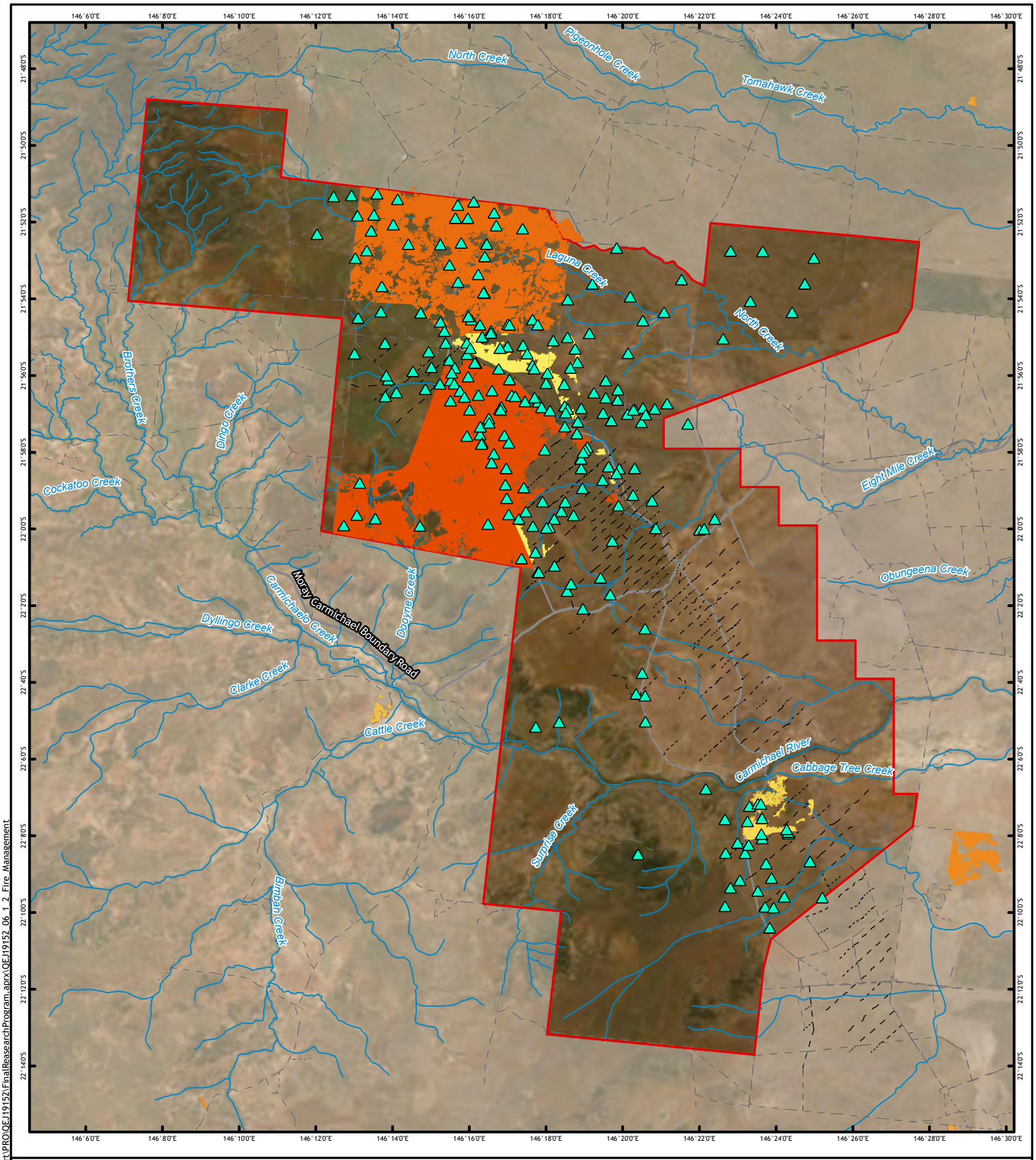
FIGURE 6.1-1: GRAZING MANAGEMENT

BTFRP Final Report
Bravus

Map Number	Job Number	Rev
1 of 1	QEJ19152	2

Document Path: X:\JOBS\-2019\QEJ19152\GIS\Final Research Program Report\PRO\QEJ19152\FinalReasearchProgram.aprx\QEJ19152_06_1_1_Grazing_Management

E2M Pty Ltd gives no warranty in relation to the data (including accuracy, reliability, completeness or suitability) and accepts no liability for any loss, damage or costs (including consequential damage) relating to any use of the data in this map.



Legend

- Vegetation Monitoring Sites
- Major Road
- Minor Road
- Exploration Track
- Track
- Watercourse
- Project Area

Burn End Date

1/07/2022 1/08/2022 1/09/2022 1/10/2022 1/11/2022 1/12/2022 1/06/2023 1/07/2023

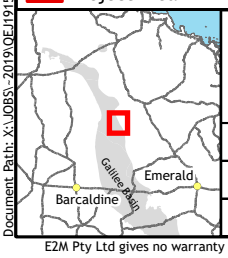
N

Scale 1:250,000 (A4)

0 5 10

Kilometers

Coordinate System: GDA2020



Notes:
Aerial Imagery: © ESRI 2021
Cadastre: © DNRME 2024
Watercourse: © Geoscience Australia 2012
Road: © DNRME 2024

Rev	Description	Drawn	Approved	Date
2	Issued for Use	JY	JMC	30/10/2025
0	Issued for Use	GO	JMC	08/08/2025

FIGURE 6.1-2: FIRE MANAGEMENT

BTFRP Final Report
Bravus

Map Number	Job Number	Rev
1 of 1	QEJ19152	2

Document Path: X:\JOBS\-2019\QEJ19152\GIS\Final Research Program Report\PRO\QEJ19152\FinalReasearchProgram.aprx\QEJ19152_06_1_2_Fire_Management

E2M Pty Ltd gives no warranty in relation to the data (including accuracy, reliability, completeness or suitability) and accepts no liability for any loss, damage or costs (including consequential damage) relating to any use of the data in this map.

6.1.3.2 Fire and grazing programme

A range of fire and grazing management obligations were imposed on the study area in approximately early-2020 arising from the environmental approval and biodiversity offsets requirements associated with the Carmichael Coal Mine (CO2 Australia, 2019; EcoLogical Australia, 2023). The requirements included stocking rate restrictions and fire management requirements (discussed below) that provided a valuable opportunity to evaluate the long-term vegetation response of these management actions.

6.1.3.2.1 Fire

An ecological and cultural burning programme was initiated in 2022 throughout the study area, which aimed to implement best-practice long term fire management principals using First Nations burning techniques focusing on the use of low-intensity mosaic burns (Russell-Smith, Start and Woinarski, 2001; Woongal Environmental, 2021). Fire planning recognised the risk of fires threatening life and property, while providing an ecologically and culturally beneficial fire program (WES, 2021). The cultural burning programme also considered historical burn regimes, prior to 2022. Fires occurred regularly in the landscape, particularly in Eucalypt dominated woodlands, which had a typical fire return interval of between every 3 and 7 years (Terrestrial Ecosystems Research Network, 2020). Fire was largely excluded within the site between 2011 and 2022, as a result of changed management (WES, 2021).

Burning was undertaken in eucalypt woodlands at the end of each wet season when temperatures are generally mild (mean daily average temperature = 27.8°C), while the soil remained moist, and the grass fuel had not cured extensively. The timing of burning and use of spot ignition aimed to provide low to moderate intensity fires and create a patchwork of burnt and unburnt areas.

During the study, eight controlled burns were conducted between 2022 and 2024, led by First Nations rangers (Figure 6.1-2). Following the implementation of each burn, fire-scars were mapped using Sentinel 2 satellite imagery, which was captured every 10 days and ground-truthed during vegetation monitoring.

6.1.3.2.2 Grazing

The majority of the study area had maximum stocking rate limits imposed by environmental approval requirements (Queensland Government Environmental Authority EPML01470513; (Department of Environment, Science and Innovation, 2024). These stocking rate limits ranged from grazing only under extenuating circumstances (e.g. to mitigate the risk of uncontrolled bushfire) within parts of the biodiversity offset area, termed ‘strategic grazing’, to a stocking rate of 1 beast per 10 ha, which is a moderate stocking rate for the region (Figure 6.1-1; Jones, Aisthorpe and Chilcott, 2006; Paton, Hamilton and Emery, 2011). Further operational constraints to stocking rates were required by the agistee to manage their business operations and to spell paddocks for two years following burning (CO2 Australia, 2019). Additional paddock stocking included paddocks with no stock for the duration of the study period and a paddock that had a pulse grazing trial, with grazing only during the dry season (Figure 6.1-1). Due to variation in the grazing treatments among paddocks, for our data analysis, we used the stocking rate logs, which provided monthly stocking rates for each paddock within the study area, in beasts per hectare.

Vegetation monitoring was conducted between March 2020 and August 2024, coinciding with both the wet season (approximately January–June) and dry season (approximately July–November) of each year. Seasonal surveys captured ground cover changes associated with rainfall and temperature driven cycles of vegetation growth and senescence (Woinarski, Andersen and Murphy, 2020). There is inter-annual variation in the onset and cessation of rainfall that defines the seasonal boundaries. We therefore categorised season for the purposes of data analysis based on the yearly timing of rainfall and growth of annual species.

Vegetation monitoring site location was selected using a stratification of habitat types and land management units across the study area to achieve replicates of vegetation monitoring sites across the variation of vegetation community, fire history, potential burn management areas and paddocks (linked to stocking rates). During the four-year study, vegetation monitoring was undertaken at a total of 239 unique sites that were surveyed on average 3.7 times per site (range = 1–9; Table 6.1-1).

Table 6.1-1: Number of vegetation monitoring sites surveyed per year and season

Year	Season		
	Dry	Wet	Total
2020	78	29	*107*
2021	82	119	*201*
2022	59	124	*183*
2023	72	136	*208*
2024	41	160	*201*
Total	*332*	*568*	*900*

Monitoring of ground layer composition was undertaken using 1 m^2 quadrats that were evenly distributed over each 100 m transect (Eyre *et al.*, 2015). The number of quadrats undertaken at each transect was either five (n = 47 sites) or ten (n = 192 sites), which varied because of differences in monitoring regimes implemented under different environmental management plans (CO2 Australia, 2019; EcoLogical Australia, 2023). We included the number of quadrats per site as a random effect in the statistical models to account for this bias. Within each quadrat, the projected foliage cover of each ground layer and shrub species was recorded, as well as the cover of organic litter, bare ground, rock and coarse woody debris (Eyre *et al.*, 2015).

6.1.3.3 Ground layer components

We grouped 'ground layer components' by assigning the plant growth form (e.g. graminoid, forb and woody vegetation) to each species using the AusTraits database (Wenk *et al.*, 2024). Trees and shrubs were combined into woody vegetation, due to their limited occurrence in the ground cover quadrats. Plant growth forms were further divided by their origin (native or non-native). Native graminoids, which comprise the majority of the ground layer biomass of savanna woodlands (Beerling and Osborne, 2006; Van Coller *et al.*, 2018) were further divided into their life history group, which included annual species (living less than 1 year) and perennial species (living more than 1 year) (Wenk *et al.*, 2024). Ground layer components that contributed less than 1% of records (e.g. ferns and palms) were excluded from statistical analyses.

6.1.3.4 Statistical analysis

We examined the effects of grazing and fire on ground layer composition by fitting three Bayesian regression models using the brms package version 2.21.0 and implemented in STAN (Bürkner, 2017). A Bayesian implementation was chosen to provide a statistically robust and highly flexible method to model multi-variate response data with both fixed and random effects (Liu and Kong, 2015; Austin, Lee and Leckie, 2020). The multi-variate response variables to the three models included: :1) the foliage cover of each ground layer component; 2) the foliage cover of each taxonomic group; and 3) the species richness of each ground layer component. Predictor variables included in each model were the average stocking rate

for each site during the study period, the time since the last burn and season. We included random effect terms that allowed site-level variation in both the initial value of each ground layer component (the intercept) and the change in each ground layer component through time (the slope). The number of quadrats per vegetation monitoring transect was added as an additional random effect term. We also tested models containing an interaction term between stocking rate and time since last fire; however, these models had a poorer prediction accuracy than the main effects model, which was assessed using leave-one-out cross-validation (Vehtari, Gelman and Gabry, 2017). We used a zero-inflated beta family for the first and second models described above, as these were in the form of proportional data bounded between zero and one (Liu and Kong, 2015; Douma and Weedon, 2019), while a Poisson family was used for the third model, which comprised count-based data (Bürkner, 2017). Uniform (flat) prior distributions were used for all families, which reflected minimal prior knowledge about the variation among the multivariate response distributions within each model (Bürkner and Charpentier, 2020). These priors were designed to be weakly informative allowing the data to predominantly influence the posterior estimates (Bürkner and Charpentier, 2020). Models were run with eight chains, each with 2000 iterations per chain, split evenly between warm-up and sampling. We checked the model fit using brms convergence diagnostics and posterior predictions (van de Schoot *et al.*, 2021).

6.1.4 Results

6.1.4.1 Ground layer composition

Higher stocking rates were associated with higher cover of non-native graminoids (slope: 8.05, 95% CI 2.64–13.5, Figure 6.1-3), particularly *Cenchrus ciliaris* (Figure 6.1-4), which contributed to 92.5% of the cover of non-native graminoids recorded. Lower stocking rates were associated with higher covers of native perennial graminoids (slope: -5.27, 95% CI -10.1–-0.666, Figure 6.1-3), with *Themeda triandra* and *Heteropogon contortus* showing increases in cover (Figure 6.1-4). Non-native forb cover also increased at lower stocking rates (slope: -5.13, 95% CI -9.74–-0.690, Figure 6.1-3), particularly *Stylosanthes* species (Figure 6.1-4), which contributed to 99.0% of the cover of non-native forbs recorded. Higher stocking rates were associated with higher cover of organic litter (slope: 3.16, 95% CI 0.390–5.93) but did not have a strong effect on the cover of native woody vegetation, bare ground, or coarse woody debris, with the 95% credible intervals overlapping with zero (Figure 6.1-3).

Recent burning was associated with increased native annual graminoids (slope: -0.0284, 95% CI -0.0463–-0.0107, Figure 6.1-3), particularly annual Aristida species that showed the strongest increase (Figure 6.1-4) and contributed to 56.1% of the cover of native annual graminoids recorded. Native forbs also increased following fire (slope: -0.0275, 95% CI -0.0426–-0.0400, Figure 6.1-3), with *Indigofera* species and *Evolvulus alsinoides* showing the strongest increases amongst the taxa assessed in the models (Figure 6.1-4). In contrast, with increased time since the last burn, increases were observed in native perennial graminoids (slope: 0.0403, 95% CI 0.0249–0.0558), non-native forbs (slope: 0.0283, 95% CI 0.0121–0.0454), and non-native grasses (slope: 0.0299, 95% CI 0.00789–0.0521)(Figure 6.1-3). These increases were driven by *Triodia pungens*, *Themeda triandra* and *Enneapogon virens* for native perennial graminoids, *Stylosanthes* species for non-native forbs and *Cenchrus ciliaris* for non-native graminoids (Figure 6.1-4). Recent burns were also associated with higher bare ground cover, lower litter and native woody vegetation cover, but did not have a strong effect on the cover of coarse woody debris (Figure 6.1-4).

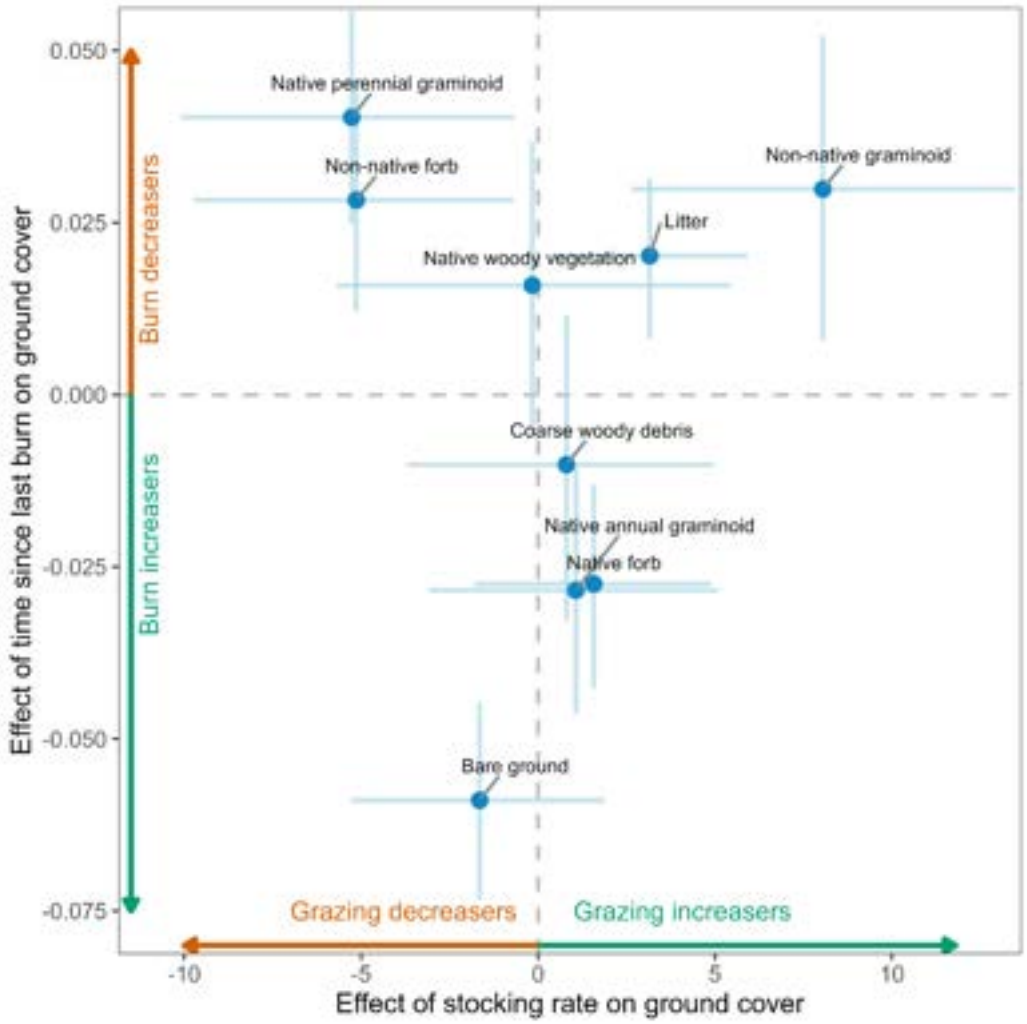


Figure 6.1-3: Effect sizes of stocking rate and time since the last burn on the cover of ground layer components. The central point (dark blue) shows the median effect size and lines (light blue) show 95% credible intervals of the median. All parameter estimates have been calculated from the posterior distribution of a Bayesian zero-inflated beta regression model. Values above 0 on the x-axis represent an increase in cover of the ground layer component with increasing stocking rate, whereas effect sizes below 0 represent reductions. Values above 0 on the y-axis represent an increase in cover of the ground layer component with increasing time since the last burn, whereas effect sizes below 0 represent reductions. Values further from 0 represent larger effect sizes than values closer to 0.

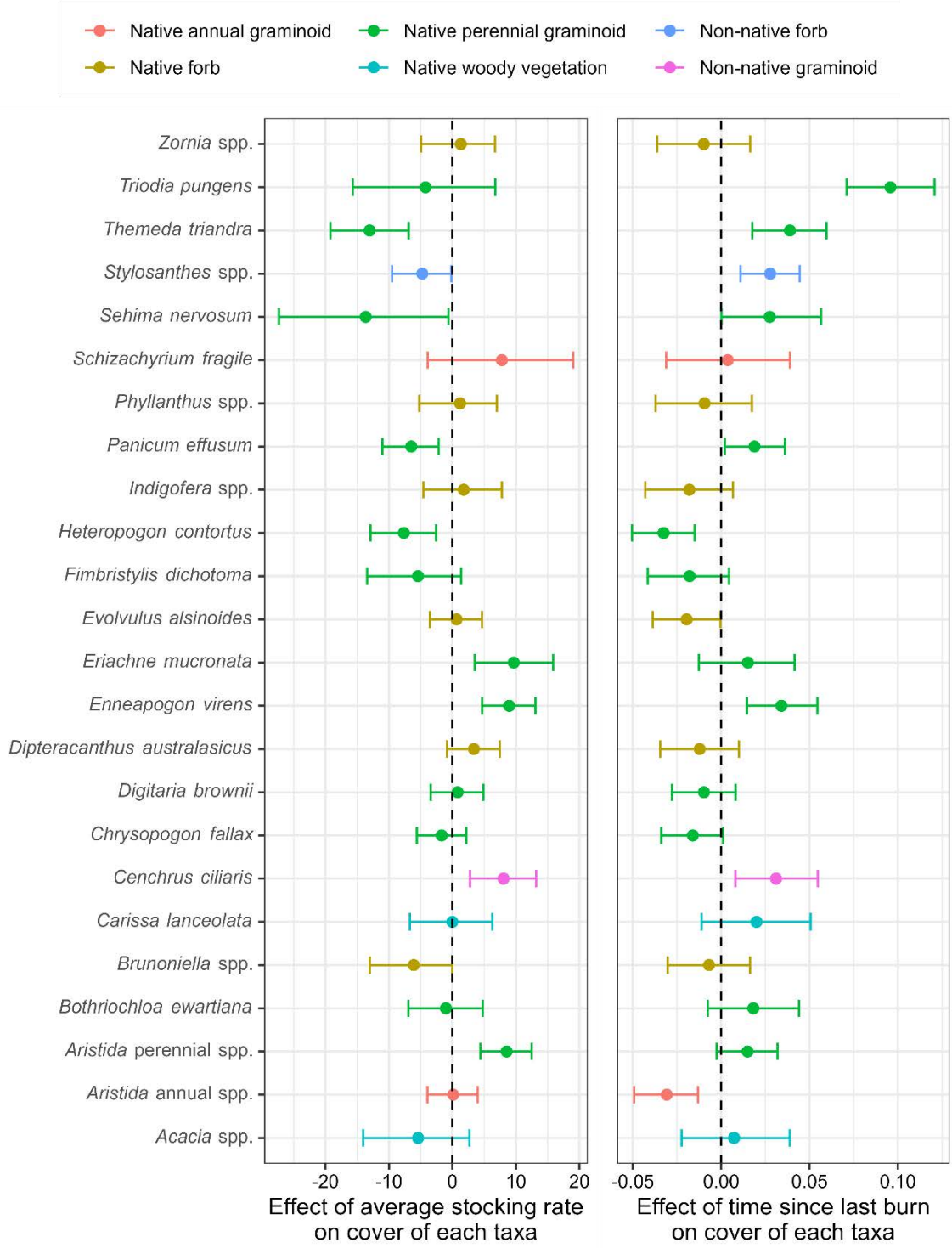


Figure 6.1-4: Effect of stocking rate over the 4-year study period and time since last burn on the cover of each taxa. Parameter estimates are shown with 95% credible intervals. Parameter estimates have been sampled from the posterior distribution of a Bayesian zero-inflated beta regression model. Values above 0 represent an increase in cover of the ground layer component with increasing stocking rate or time since burning, whereas parameter estimates below 0 represent reductions. Values further from 0 represent larger effect sizes than values closer to 0.

6.1.4.2 Species richness

The species richness of non-native graminoids increased under higher stocking rates (slope: 6.43, 95% CI 2.35–10.3, Figure 6.1-5). In contrast, the stocking rate had no strong effect on the species richness of all other ground layer components, with 95% CIs that overlapped with zero (Figure 6.1-5).

Recent burning was associated with increased species richness of annual native graminoids (slope: -0.0334, 95% CI -0.0493–-0.0174) and native forbs (slope: -0.0321, 95% CI -0.0444–-0.0196)(Figure 6.1-5), while the species richness of non-native graminoids was higher in long-unburnt sites (slope: 0.0234, 95% CI 0.00355–-0.0422). Time since the last burn had no strong effect on the species richness of non-native forbs, native perennial graminoids or native woody vegetation, with 95% CIs that overlapped with zero (Figure 6.1-5).

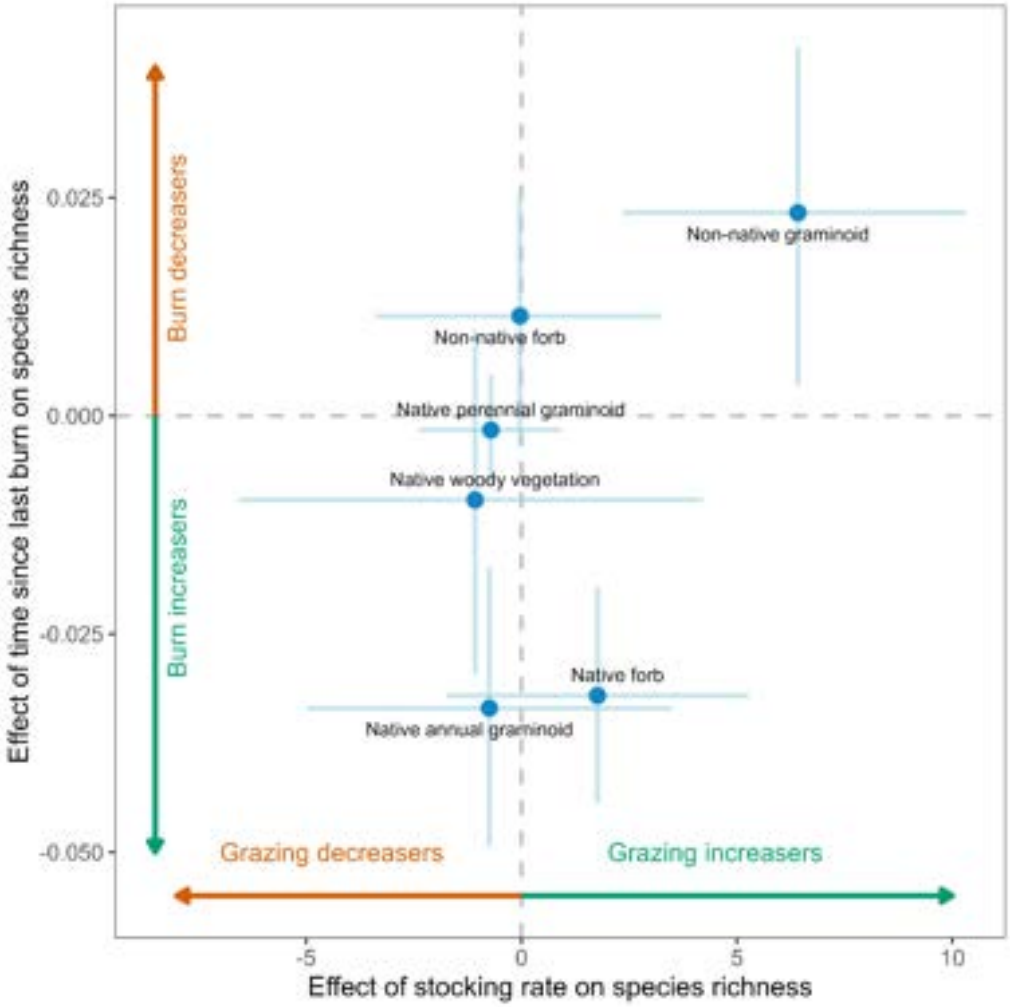


Figure 6.1-5: Effect sizes of stocking rate and time since the last burn on the species richness of growth form categories. The central point (dark blue) shows the median effect size and lines (light blue) show 95% credible intervals of the median. All parameter estimates have been calculated from the posterior distribution of a Bayesian Poisson regression model. Values above 0 on the x-axis represent an increase in species richness of the growth form category with increasing stocking rate, whereas effect sizes below 0 represent reductions. Values above 0 on the y-axis represent an increase in species richness of the growth form category with increasing time since the last burn, whereas effect sizes below 0 represent reductions. Values further from 0 represent larger effect sizes than values closer to 0.

6.1.5 Discussion

6.1.5.1 Effects of grazing on ground layer composition

We assessed the effects of cattle stocking rate and fire on the ground layer composition of a savanna woodland over a four-year period. Both stocking rate and time since the last fire had strong effects on the cover and species richness of ground layer components.

Grazing selectively removes graminoid biomass based on the plants' palatability and tolerance to herbivory (Archibald, Hempson and Lehmann, 2019; Pfeiffer *et al.*, 2019). Our study found that higher grazing pressures increased the cover of non-native graminoids at the expense of native perennial graminoids. Specifically, grasses such as *Sehima nervosum*, *Heteropogon contortus*, and *Themeda triandra* reduced in cover under higher stocking rates. These species are known for their sensitivity to grazing pressures and have previously been identified as indicators of savanna ground layers in a good condition (Ash *et al.*, 1997; McCosker, Rolfe and Fensham, 2009; Fensham, Wang and Kilgour, 2015). Our study therefore provides valuable empirical evidence in support of the widely held understanding that increased grazing pressure diminishes palatable perennial grasses at the expense of potentially less favourable ground layer components such as non-native grasses and litter (Ash *et al.*, 1997; Eldridge *et al.*, 2016; Archibald, Hempson and Lehmann, 2019; Pfeiffer *et al.*, 2019).

While some studies suggest that light to moderate grazing can enhance ground layer species richness (Symstad and Jonas, 2011; Socher *et al.*, 2013), we found that only the richness of non-native graminoids increased with higher stocking rates. Grazing's effect on species richness is influenced by multiple factors, including herbivore type, stocking rate, plant productivity, and the evolutionary history of vegetation with herbivore types (Eldridge *et al.*, 2016). Our study area had multiple factors that predisposed it to negative effects of cattle grazing, even at relatively low stocking rates for the region. In their framework to predict the effects of livestock grazing on conservation values of Australian ecosystems, Lunt et al., (2007) argue that sites with intact ground covers that are relatively uninvaded by exotic species, such as our study area, are likely to show detrimental impacts of livestock grazing. This is further evidenced by the common occurrence of grazing sensitive native species within our study area, such as *Sehima nervosum*, which are uncommon at sites with a history of high stocking rates (McCosker, Rolfe and Fensham, 2009). Furthermore, our study area occurs in a hot semi-arid climate zone (Peel et al., 2007) with low-productivity soils, with aridity and low plant productivity being key factors that increase the susceptibility of vegetation to negative grazing impacts (Lunt *et al.*, 2007; Eldridge *et al.*, 2016). Finally, Australia's native flora has had a short history of grazing by large, hard-hooved domestic stock, with less than 200 years of exposure. Prior to European colonisation and the subsequent expansion of artificial water sources and domestic livestock, much of our study area would have been remote from water and experienced low historical grazing rates (Landsberg *et al.*, 2003; Fensham and Fairfax, 2008). In contrast, grazing may be a more appropriate management tool in already heavily degraded sites, such as where control of undesirable grazing sensitive biomass is required to afford a disturbance niche for desirable plant species (Lunt *et al.*, 2007; Melzer, 2015), or ecosystems that have evolved with grazing by hard-hooved mammalian herbivores (Symstad and Jonas, 2011; Socher *et al.*, 2013). A summary of the conceptual effects of grazing and fire regimes is presented in Table 6.1-2.

Table 6.1-2: Summary of the conceptual effects of fire and grazing regimes on ground cover composition.

Management	Pressure	Effect
High grazing, low fire frequency	More grazing of palatable species. Less consumption of non-palatable biomass.	More shrub cover. More grazing tolerant species cover (annuals, non-palatable perennial species and species with high investment into rootstock). More investment into vegetative material and roots. Low grass seed production.
Low grazing, high fire frequency	Fire consumes species with a high aerial biomass. Bare ground patches following fire promote the establishment of grasses that invest into seed production.	Less shrub cover. Less high aerial biomass perennial cover. More grazing sensitive species. Higher cover of grasses that invest energy into seed production over biomass.
Low grazing, low fire frequency	Low consumption of aerial biomass. Less bare ground limiting seed establishment.	More high biomass perennial species competing for space and resources. Low grass seed production. More shrub cover.

6.1.5.2 Effects of fire on ground layer composition

The transition of ground layer vegetation observed in this study aligns with established principles of savanna vegetation response to fire (Russell-Smith, Start and Woinarski, 2001; Williams, Woinarski and Andersen, 2003; Cowley *et al.*, 2014; Lebbink, Fensham and Cowley, 2018). Fire typically opens up the ground layer, increasing bare ground cover and creating favourable conditions for fast-growing annual species that quickly reach seeding stage (Fensham, Wang and Kilgour, 2015; Van Coller *et al.*, 2018). As time since the last fire increases, perennial graminoids, such as *Triodia pungens, Themeda triandra, Enneapogon virens*, and *Cenchrus ciliaris*, gradually dominate. These species generally develop dense, aerial stems that suppress annual species (Archibald, Hempson and Lehmann, 2019). In our study, time since last burn had the strongest effect on *Triodia pungens*, a species of spinifex. Given their slow growth, dense aerial biomass and grazing resistance, Triodia species often dominate in long-unburnt landscapes (Wright and Clarke, 2007; Kenny *et al.*, 2018).

6.1.5.3 Effects of grazing and fire on exotic plant cover

Our study identified a strong increase in non-native graminoid cover at sites with high stocking rates. Buffel grass was the main non-native graminoid that increased under higher stocking rate regimes. While valuable as a pastoral species, buffel grass also threatens native vegetation due to its resilience in harsh conditions, tolerance for high grazing pressure, and strong competitiveness with native species following disturbance (Marshall, Lewis and Ostendorf, 2012; Farrell and Gornish, 2019). The species' ability to withstand grazing results from morphological adaptation, where basal tillering is triggered by defoliation, keeping plant tissue below the grazing height—an adaptation absent in native species that evolved in fire-disturbed environments (Hodgkinson *et al.*, 1989). Additionally, buffel grass possesses life history traits that enhance its competitiveness in disturbed landscapes, including a deep root system, long-lived tussocks, extended seed longevity in the seed bank, and opportunistic germination (Marshall, Lewis and Ostendorf, 2012).

Our findings suggest that low grazing pressure combined with fire can reduce buffel grass abundance, in contrast to high grazing regimes where fire is excluded. Reviews of buffel grass control identify labour-intensive methods, such as herbicide application and manual removal, as the only reliable control options (Marshall, Lewis and Ostendorf, 2012; Melzer, 2015; Farrell and Gornish, 2019). However, such methods are typically infeasible over broadscale areas like our 75,000 ha study area. Our results indicate that low or no stocking rates, combined with a regime of patchy low-intensity burns, may help build savanna woodlands' resilience to buffel grass invasion. Similarly, Jackson (2004) recommended that, despite subtle differences between buffel grass and native perennial grasses, buffel grass is more susceptible to burning, particularly in the early dry season and after out-of-season rainfall events. The findings of this study and Jackson (2004) contradict some previous research showing buffel grass' ability to exploit fire to dominate native communities; however, these results are usually associated with uncontrolled, high-intensity burns or in areas already dominated by buffel grass (Butler and Fairfax, 2008; Miller *et al.*, 2010; Melzer, 2015). In our study, we implemented low-intensity burns in areas with generally low buffel grass cover and high native grass richness. Here, reducing grazing pressure and applying fire appears to manage buffel grass not by suppressing its growth but by enhancing native species' competitive advantage through habitat restoration to conditions more favourable for them (Ash *et al.*, 1997; Alba *et al.*, 2015).

Stylosanthes species are another ecological weed, introduced widely across northern Australia for agricultural purposes (Gardener, 1993). Although highly palatable to cattle, *Stylosanthes* species are resilient to heavy grazing and can quickly colonise uninfested areas via seeds dispersed in cattle faeces (Gardener, 1993; Orr, 2010). In our study area, *Stylosanthes* species were the primary non-native forb, contributing to 99% of non-native forb cover. Our findings support the use of patchy low-intensity fires to reduce *Stylosanthes* cover. We also found that higher stocking rates led to reduced *Stylosanthes* cover. This aligns with the findings of Kemp & Kutt (2020) who found that over 10 years of cattle exclusion from a conservation area, *Stylosanthes* cover increased significantly, even in areas not previously containing *Stylosanthes* species. However, while grazing reduces aboveground biomass due to selective grazing (Coates, 1996), *Stylosanthes* can persist due to its grazing tolerance and capacity to store energy in underground tubers (Gardener, 1993). Although grazing may temporarily reduce aboveground cover, maintaining control through consistent heavy grazing risks reducing ground layer diversity, depleting grazing sensitive species and increasing invasion rates by grazing tolerant exotic species (Lunt *et al.*, 2007). Despite increasing *Stylosanthes* cover, Kemp & Kutt (2020) also reported greater native species diversity in the ground layer after excluding grazing. Conservation strategies for *Stylosanthes* species must therefore consider a long-term approach of prescribed burning combined with targeted treatments (e.g. herbicide) where required.

6.1.5.4 Secondary fauna impacts of ground layer composition

Overgrazing and the subsequent decline in ground layer condition (Ash *et al.*, 1997) has a range of conservation implications. Higher grazing pressures reduce the seed production grasses and forbs through grazing of inflorescences and defoliation reducing plant yield (O'Connor and Pickett, 1992; Ash *et al.*, 1997). The negative effects of grazing reducing seed output are more pronounced for high-palatability perennial grasses such as *Themeda triandra* and *Heteropogon contortus*, than an introduced species such as *Bothriochloa pertusa* (Ash *et al.*, 1997). Such alterations of seed production has contributed to a widespread disarray among the granivorous bird assemblage of the Australian savannas (Franklin, 1999; Franklin *et al.*, 2005), including the endangered black-throated finch (southern) that has been recorded within the study area (GHD, 2013; Mula Laguna *et al.*, 2019). There is also a deleterious interaction among livestock grazing and the ground layer's response to fire, with livestock preferentially grazing in recently burnt areas, which slows grass production (Ash *et al.*, 1997) and potentially leads to higher rates of predation from pest animals such as cats (Legge *et al.*, 2019).

6.1.5.5 Interactive effects of fire and grazing and their management

Grazing and fire dynamics in savanna woodlands are closely linked and require joint management to achieve conservation gain (Ash *et al.*, 1997; Archibald, Hempson and Lehmann, 2019). When stocking-rates are reduced, accumulated biomass can substantially increase, which leads to an increase in the risk of uncontrolled bushfire, higher fire intensities and larger patch sizes (Ash *et al.*, 1997; Liedloff *et al.*, 2001; Hunt *et al.*, 2014). A strategic mosaic of small patches that are burnt at a low intensity, particularly in the late wet season to early dry season may reduce the incidence of large late dry season fires (Fensham, 2012; Murphy & Russell-Smith, 2010). The approach follows the fire management practices of Traditional Owners by using frequent and small-scale fires to proactively manage biomass and create breaks between areas of high biomass (Fensham, 2012; Russell-Smith & Yates, 2007; Yibarbuk et al., 2001).

6.1.6 Conclusion

Broadly, our results provide valuable empirical evidence to support the recommendation that low grazing pressure and a mosaic fire regime promote ecosystem health of savanna woodlands. Our results support a long-standing suggestion that high stocking rates and altered fire regimes are degrading the ground layers of savanna woodlands (Williams, Woinarski and Andersen, 2003; Franklin *et al.*, 2005; Kutt and Woinarski, 2007; Silcock and Fensham, 2018; Legge *et al.*, 2019) and support the use of no to low grazing regimes, combined with an ecological burning program to support the health of savanna woodlands.

6.2 Response of black-throated finch to the provision of artificial water sources

6.2.1 Introduction

Australia is known for being a dry continent, with much of the landscape occurring in the arid zone (Fisher, Lindgren and Dawson, 1972; Davies, 1977). The introduction and expansion of modern animal agriculture in Australia has brought with it changes to the landscape, including the introduction of artificial water points for stock watering in previously dry or seasonally-dry landscapes (Davies, 1977; James, Landsberg and Morton, 1999). This has had an impact on the distribution of bird species (Davies, 1977; Reid and Fleming, 1992; Harrington, 2002), with some bird species increasing in range and or abundance, while others have decreased (James, Landsberg and Morton, 1999; Harrington, 2002; Abdu, Lee and Cunningham, 2018).

The foraging and nesting habitats of SBTF are constrained by the availability of a permanent nearby water source, particularly in the dry season (Rechetelo *et al.*, 2016a; E2M, 2021). Water sources could be in the form of a farm dam or trough, natural spring, perennial stream, or ephemeral creek or pool (Buosi, 2011b; E2M, 2021). In the study area, surveys to date have identified that the mean distance between nests and permanent water sources to be approximately 1.7 km (E2M, 2021). It is suggested that suitable habitat for SBTF is not being used due to the absence of appropriate nearby water sources, and therefore, it is hypothesised that the addition of watering points will increase the area of occupancy of the species, particularly through the dry season (E2M, 2021). However, potential drawbacks of introducing additional artificial water sources that implicate SBTF should be considered.

The implementation of artificial water sources in Australia has resulted in the formation of grazing piospheres (Lange, 1969) where the area surrounding the water point is severely degraded due to overgrazing stock that congregate around water points (James, Landsberg and Morton, 1999; Fensham and Fairfax, 2008). Increased artificial water sources may increase the detrimental impacts of grazing, including changes to species composition, vegetation structure and ecosystem functions (Lunt *et al.*, 2007; Eldridge *et al.*, 2016) (Figure 6.2-1). Healy, Tulloch and Fensham (2020) support the removal of artificial water sources to create ecological refuges. Granivores, such as SBTF, are particularly susceptible to changes in grazing regimes, due to their need for a specific vegetation structure, species composition, seed phenology and seed nutritional value (Weier, Radford, *et al.*, 2017b; Williams *et al.*, 2020). Thrash *et al.* (1991) commented on the increased amount of area that is available to be grazed during the dry season from the implantation of additional water points and hence decreasing the availability of untouched habitat for SBTF.

The provision of artificial water sources may also increase the frequency of feral cats to the area (James, Landsberg and Morton, 1999; Brawata and Neeman, 2011) (Figure 6.2-1). While feral cats are known to prey on estrildid finches, the threat on SBTF remains uncertain (Mula Laguna *et al.*, 2019). However, studies have found that the likelihood of cat predation is higher for those species that aggregate around water sources (Paltridge, Gibson and Edwards, 1997). Cats get most their water needs from their live prey and often use water sources to hunt, rather than drink (James, Landsberg and Morton, 1999). In addition, cats often prefer areas with open grass cover, including heavily grazed areas, most likely to improve their hunting success (McGregor *et al.*, 2014). Hence, cats may prefer to hunt around heavily-gazed, artificial water sources, increasing the threat to SBTF.

Several native birds have expanded their geographical range or abundances due to the provision of artificial water sources (James, Landsberg and Morton, 1999). This includes species such as the yellow-throated miners that are aggressive and can outcompete smaller birds for water (Grey, 1996). Aggressive species have been found to disrupt and chase away small birds and actively prey on adults and nests

(Beggs *et al.*, 2020). Therefore, the introduction of artificial water sources may result in the provision of water for aggressive species rather than SBTF (Figure 6.2-1). Additionally, Beggs et al. (2020) found that the impacts of aggressive birds are lessened in the presence of high mid-dense understorey, which is not provided in heavily-grazed artificial water source areas.

Implementation of artificial water sources may increase the chances of disease outbreak among SBTF (Figure 6.2-1). Significant outbreaks of various avian diseases, including Trichomonosis, Salmonellosis, and Psittacine Beak and Feather Disease Virus (BFDV), which have led to high avian mortality rates, have been linked to the use of feeders and bird baths (Robb *et al.*, 2008; Tizard, 2004). Adelman *et al.* (2015) found that transmission of *Mycoplasma glypticus* (causing conjunctivitis) in house finches was directly correlated to the extent in which the birds interact with feeders.

This study investigated the use of permanent artificial water sources by SBTF and species that potentially threaten SBTF in the dry tropical region of northern Queensland, Australia to inform management actions. Transect counts at various distances to permanent water sources were used to determine the reliance of SBTF and potential threatening species on water. In addition, remote camera monitoring was used to determine the influence of establishing new artificial water sources on the utilisation of adjacent habitat by SBTF. Using the data collected, in conjunction with a review of potential threats to SBTF, recommendations can be made as to whether the introduction of new artificial water sources is a beneficial land management strategy for SBTF.

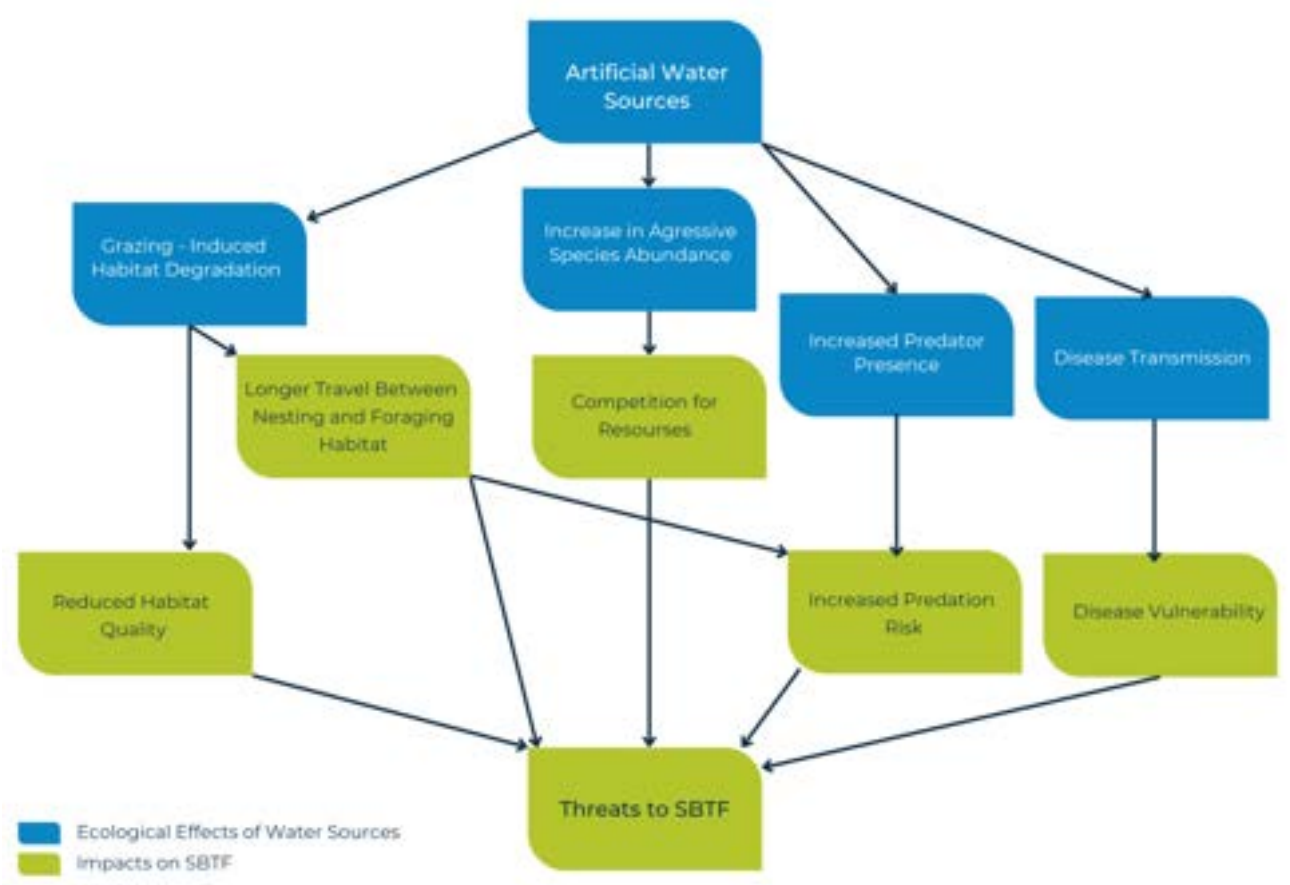


Figure 6.2-1: Ecological effects of the provision of artificial water sources and potential impacts on SBTF

6.2.2 Methods

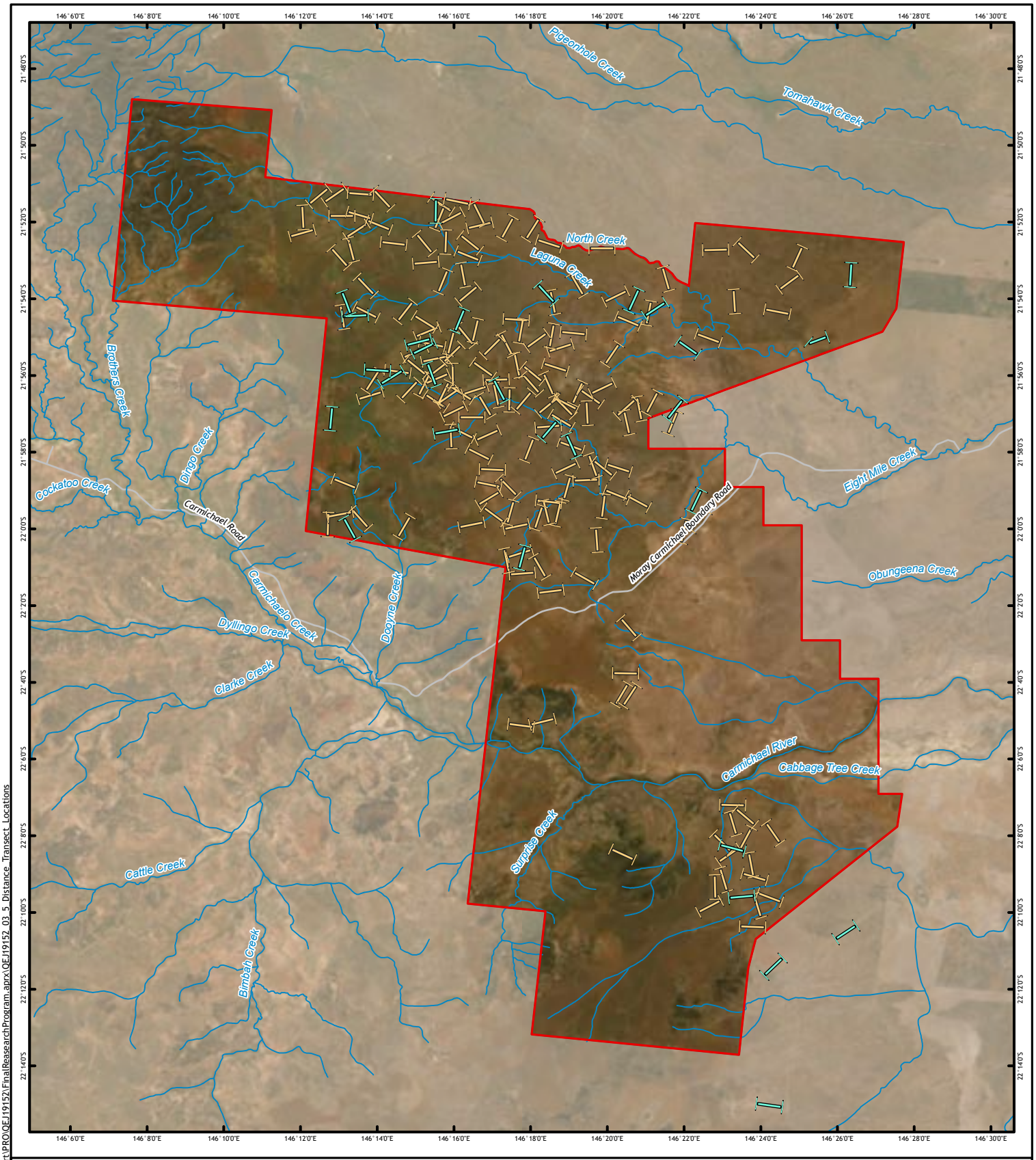
6.2.2.1 1km transects

In this study, 1km distance transects were used to determine abundance of bird species in relation to the distance to permanent water. The observer walked a pre-defined 1 km transect recorded the location and species of birds observed. All observations of SBTF were recorded; however, only the first occurrence per transect of all other bird species was recorded. The stratification of transects within the study area was determined by SBTF records and distance to water. Using this methodology, a total of 167 sites were established which are presented in Figure 6.2-2. 10 surveys were undertaken between 2020 and 2023 throughout the early wet, wet and dry season to account for variance in species behaviour between seasons. A total of 1,312 sies transects were carried out during the study (Table 6.2-1).

Table 6.2-1: Number of 1km transects carried out per season

Season	Number of transects
Early wet	458
Wet	280
Dry	574
Total	**1,312**

A Generalized Linear Mixed Model (GLMM) was used to determine the effect of distance to permanent water on the number of each species recorded per meter of transect effort. For each species, a standard GLMM was fitted using the glmmTMB package. The response variable was the species count per metre, with distance to permanent water and season as fixed effects and random intercepts for survey and site ID. If the standard model failed to provide a non-missing p-value for distance to permanent water, a zero-inflated GLMM was fitted as an alternative.



Legend

- 1km Transect
- Random Transect
- Watercourse
- Major Road
- Minor Road
- Track
- Project Area

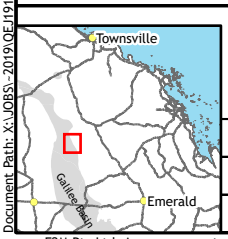
N

Scale 1:250,000 (A4)

0 5 10

Kilometres

Coordinate System: GDA2020



Notes:
Aerial Imagery: © ESRI 2021
Cadastre: © DNRME 2024
Watercourse: © Geoscience Australia 2012
Road: © DNRME 2024

Rev	Description	Drawn	Approved	Date
2	Issued for Use	JY	JMC	30/10/2025
0	Issued for Use	GO	JMC	08/08/2025

FIGURE 6.2-2: 1KM TRANSECT LOCATIONS

BTFRP Final Report
Bravus

Map Number	Job Number	Rev
1 of 1	QEJ19152	2

Document Path: X:\JOBS\-2019\QEJ19152\GIS\Final Research Program Report\PRO\QEJ19152\FinalReasearchProgram.aprx\QEJ19152_03_5_Distance_Transect_Locations

E2M Pty Ltd gives no warranty in relation to the data (including accuracy, reliability, completeness or suitability) and accepts no liability for any loss, damage or costs (including consequential damage) relating to any use of the data in this map.

6.2.2.2 Remote camera monitoring of artificial water sources

6.2.2.2.1 Water source design and site selection

The utilisation of artificial water sources by SBTF was trialled within the study area. Two water source designs were included within the research (Figure 6.2-3), which include:

- Raised water sources, which provide a cattle-excluded perched water source; and
- Cattle water sources installed primarily for the purposes of grazing management.

Raised water sources, also referred to as Tree Troughs, have been designed by Wildsip Pty Ltd (WIRES, 2020; Figure 6.2-3). The drinkers are intended to provide a temporary trial of the effectiveness of water source addition to inform the design of permanent artificial water sources. The drinkers comprise a 3 L drinking bowl that is automatically refilled from the connected 220 L water tank. The design also includes an artificial tree branch constructed from natural materials next to the water bowl to provide a drinking platform for perching birds as well as an escape ramp for fauna. The drinking bowl is located 3 m above ground level, which will exclude cattle and pest animals.

16 cattle troughs were present within the study area prior to this study. In early 2022, an additional 3 cattle troughs, as well as 6 raised water sources, were installed. Additional cattle troughs installed were assessed concurrently with the raised water source to compare patterns of usage by SBTF and response of surrounding vegetation to changes in grazing dynamics between water sources accessible and inaccessible to livestock.

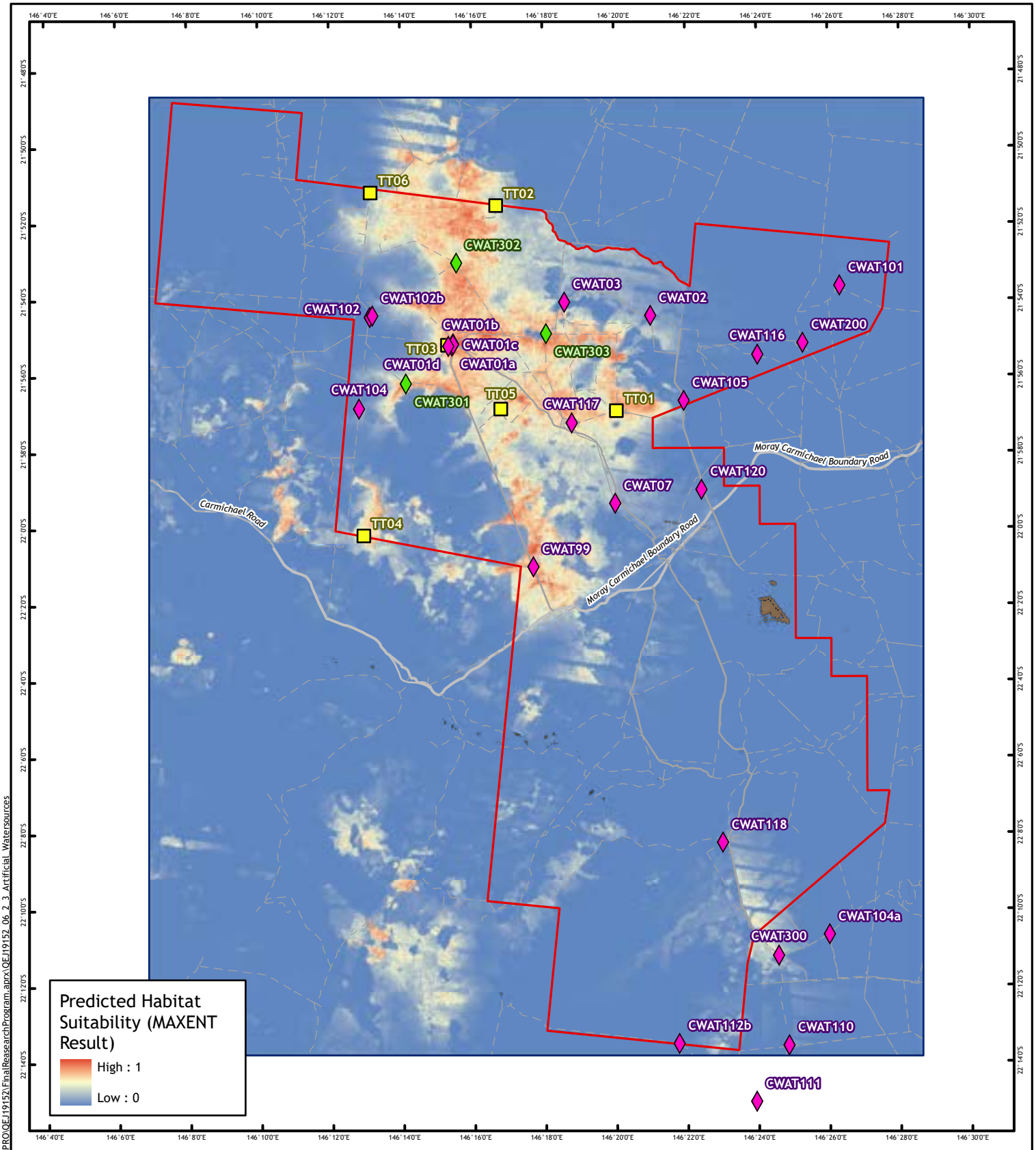
When choosing locations for artificial water sources, considerations were taken including:

- The presence suitable vegetation for SBTF,
- The proximity to existing water sources (more than 3 km),
- The proximity to areas known to support SBTF (less than 10 km); and
- The proximity to surrounding trees and shrubs (less than 10 m) on which birds can shelter as they travel to water and use as an intermediate perch while accessing water.

A single raised water source was placed in close proximity to Ten Mile Bore to test the SBTF preference for the design of the drinkers compared to other existing water sources that are available at Ten Mile Bore, including dams and cattle troughs. The locations of artificial water sources are depicted in Figure 6.2-4.



Figure 6.2-3: Raised water trough (left) and cattle trough (right) installed within the study area



Legend

- Raised Water Sources
- Cattle Trough Locations
  - Existing
  - New
- Major Road
- Minor Road
- Track
- Model Extent
- Study area

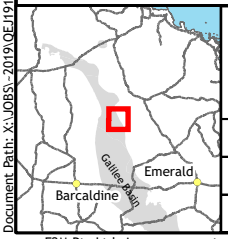
N

Scale 1:250,000 (A4)

0 5 10

Kilometers

Coordinate System: GDA 1994 MGA Zone 55
Projection: Transverse Mercator



Notes:
Aerial Imagery: © ESRI 2021
Cadastre: © DNRME 2024
Watercourse: © Geoscience Australia 2012
Road: © DNRME 2024

Rev	Description	Drawn	Approved	Date
2	Issued for Use	JY	JMC	30/10/2025
0	Issued for Use	GO	JMC	08/08/2025

FIGURE 6.2-4: LOCATION OF ARTIFICIAL WATER SOURCES MONITORED DURING THE STUDY

BTFRP Final Report
Bravus

Map Number	Job Number	Rev
1 of 1	QEJ19152	2

Document Path: X:\JOBS\-2019\QEJ19152\GIS\Final Research Program Report\PRO\QEJ19152\FinalReasearchProgram.aprx\QEJ19152_06_2_3_Artificial_Watersources

E2M Pty Ltd gives no warranty in relation to the data (including accuracy, reliability, completeness or suitability) and accepts no liability for any loss, damage or costs (including consequential damage) relating to any use of the data in this map.

6.2.2.2.2 Camera deployment

Remote cameras were used to monitor utilisation of water sources by SBTF and other species that may influence water source use by SBTF. A total of 25 water sources were monitored including:

- six raised water sources
- 16 existing cattle troughs; and
- three additional cattle troughs.

The locations of monitored artificial water sources are depicted within Figure 6.2-4. A total of 1,415 camera days of survey effort was undertaken during the study from 2022 to 2024 (Table 6.2-2).

Table 6.2-2: Number of camera days of survey effort per water source type and season

Season	Cattle Troughs	Raised Water Sources	Total
Dry	261	556	**871**
Early wet	0	103	**103**
Wet	173	322	**495**
Total	**434**	**981**	**1,415**

6.2.2.2.3 Data analysis

Images taken on the remote cameras were screened for SBTF and pest species (eg. feral cats, wild dogs, cane toads) with each image treated as a unique occurrence. The number of individuals of each species within the image was recorded along with any bands, the date, time and location. When screening images from cameras deployed at raised water sources, all animal species in every 50th image were also recorded. This subsample of photos was used to determine which other species are utilising the raised water sources.

6.2.3 Results

6.2.3.1 1km transects

A total of 139 bird species were recorded during the transects. Statistical analysis was only conducted on a subset of species where enough individuals were recorded for analysis (count = ≥ 20 individuals). Other species where the model did not converge on were also excluded. Statistical analysis was conducted on a subset of species of interest, namely SBTF, aggressive species, and other species with possible competition with SBTF.

The term ‘increaser’ and ‘decreaser’ species have been used to describe bird species that either increase or decrease with proximity to water sources (Harrington, 2002). This has been correlated to a species dependence on drinking water, which may be related to their diet (Abdu, Lee and Cunningham, 2018). The results show many species analysed did not show a statistically significant relationship between abundance and distance to permanent water, while some did (Figure 6.2-5). Several species were found to increase with proximity to permanent water (‘increaser’ species), including the magpie lark ($p < 0.0001$), galah ($p = 0.0001$), squatter pigeon ($p = 0.0003$), whistling kite ($p = 0.0005$) and Torresian crow ($p = 0.03$). No species were found to decrease with proximity to permanent water (‘decreaser’ species). SBTF did not have a statistically significant relationship between abundance and distance to permanent water ($p = 0.3$).

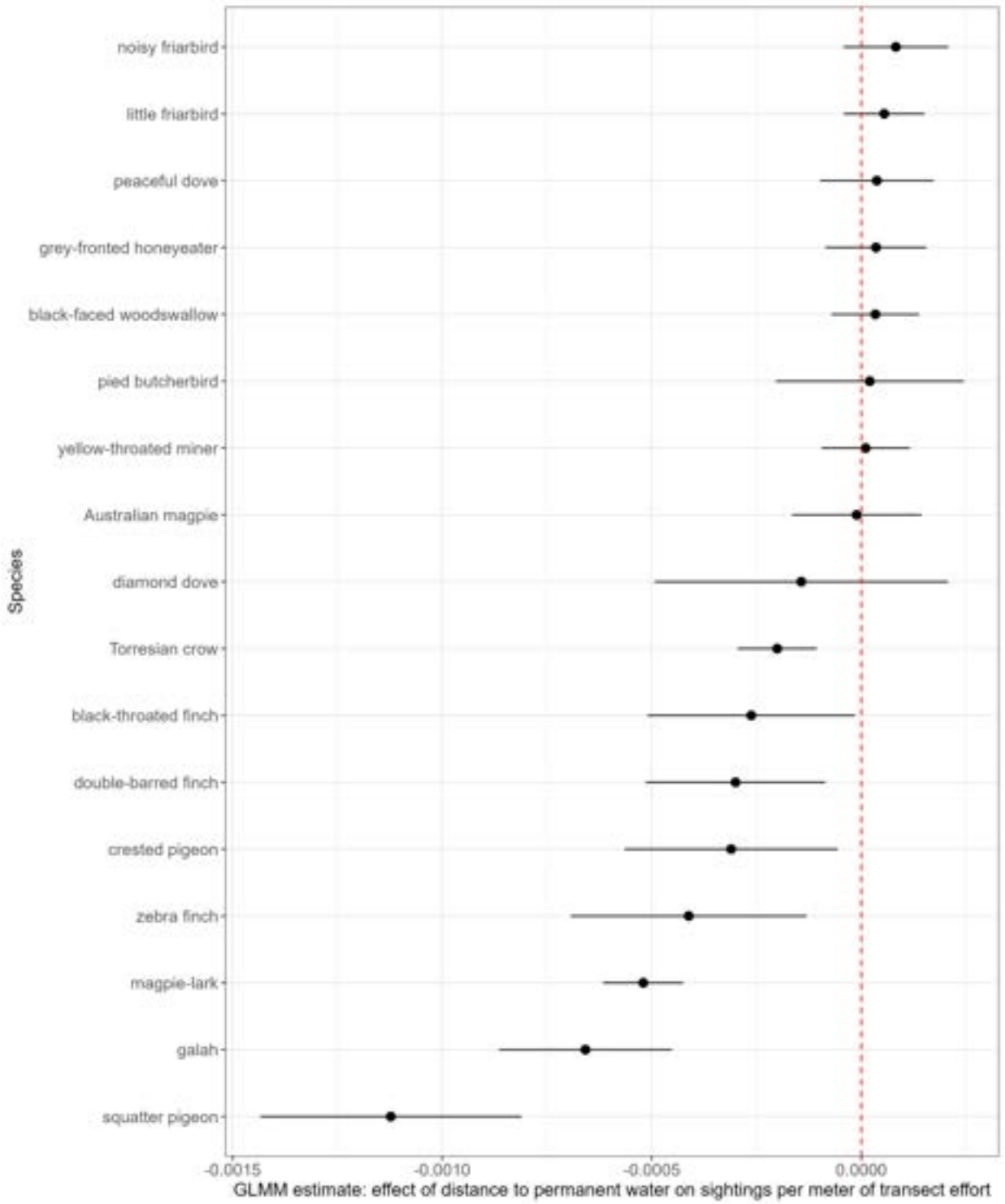


Figure 6.2-5: Abundance of birds recorded during 1 km transects in relation to their distance to permanent water. Negative values on the left represent species that have higher abundances near permanent water compared to afar. A value of 0 (red dotted line) represents birds that were recorded approximately randomly through the landscape in relation to the distance to permanent water.

6.2.3.2 Remote camera monitoring at artificial water sources

Remote cameras may capture multiple images of the same individuals during a detection event, which is defined as a 5-minute window. Within this timeframe, the maximum number of SBTF observed in any single photo was used in the analyses (Green *et al.*, 2023). This duration was selected to balance the risk of data non-independence while providing a reasonable estimate of the number of SBTF using the water source (Green *et al.*, 2023).

60% of water sources did not record any SBTF during the study. However, SBTF were recorded using 40% of the artificial water sources installed within the study area, including:

- Four existing cattle troughs (CWAT99, CWAT110, CWAT118 and CWAT300)
- Three additional cattle troughs (CWAT301, CWAT302 and CWAT303); and
- Three raised water sources (tree troughs) (TT01, TT02 and TT05).

TT01 had the highest use by SBTF for both raised water sources and cattle troughs, with an average detection rate of 42.58 SBTF per day. Of the existing cattle troughs, CWAT99 had the highest use with an average detection rate of 26.87 SBTF per day. Cattle troughs CWAT300, CWAT301 and CWAT302, which were installed within the study area in 2022, had an average detection rate of 0.56, 15.46 and 15.08 SBTF per day, respectively.

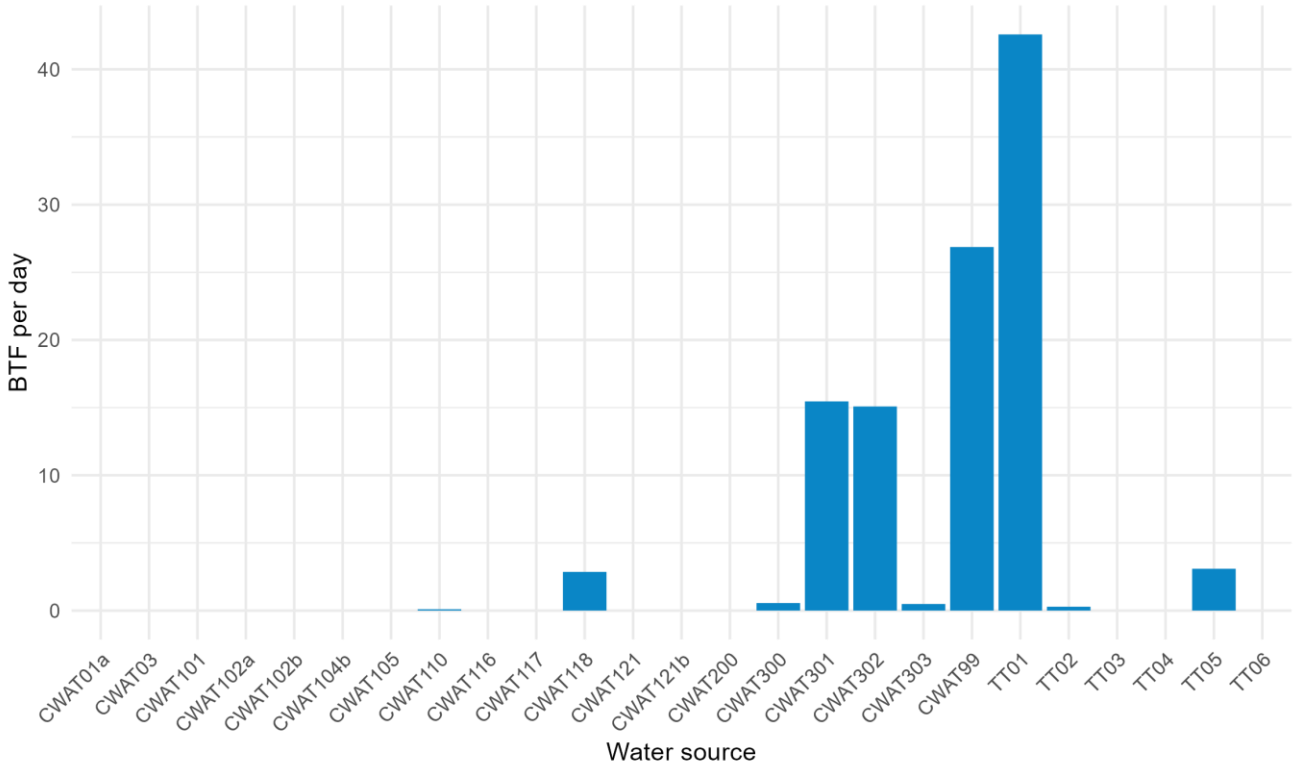


Figure 6.2-6: SBTF detection rates at water sources monitored by remote cameras

A subsample of images recorded at the raised water sources were used to determine which other species are utilising the raised water sources. To standardise sampling effort and allow for fair comparison, SBTF records were filtered to only include every 50th image in this analysis. 39 species have been observed using the raised water sources, with a vast majority of records (83.27%) comprising five species, including:

grey-fronted honeyeater, Torresian crow, yellow-throated miner, SBTF and pale-headed rosella (Figure 6.2-7). SBTF made up 15.22% of records across all raised water sources; however, the species was only recorded at three of the six raised water sources. The most recorded species when looking at raised water sources individually varied greatly between water sources (Figure 6.2-8). At TT01, SBTF was the most recorded species, making up 39.53% of records. However, at TT05, SBTF was second to grey-fronted honeyeater making up 16.93% of records. Although SBTF were recorded at TT02, the species was not in the top five species recorded at the water source.

Despite the most recorded species varying between each raised water source, the large presence of aggressive species remains consistent. Other than SBTF, the most common four species across all raised water sources were all aggressive species which collectively made up 56.15% of all records (Figure 6.2-8). At TT05, grey-fronted honeyeater made up just over 60% of records which is the highest from any site (Figure 6.2-8). However, although TT01 which had the highest proportion of SBTF use, it also had the highest number of both grey-fronted honeyeaters (247 individuals) and yellow-throated miners (152 individuals) recorded throughout the study. Other raised water sources ranged from 83 to 153 individuals for grey-fronted honeyeater and 65 to 131 for yellow-throated miner.

Torresian crow was generally recorded more at water sources where surrounding habitat had been burnt through controlled burns prior to camera deployment. This included TT02 and TT06 where Torresian crow made up 43.46% and 42.86% of records, respectively (Figure 6.2-8). The habitat surrounding TT04 was burnt during a camera deployment, after which there was a significant increase in Torresian crow records. Across all deployments this looks like co-dominance between Torresian crow (24.89%) and grey-fronted honeyeater (22.57%); however, the species were not simultaneously the most recorded species at a trough.

No pest species were recorded at raised water sources, whereas at cattle troughs, they were recorded at 12 of the 21 troughs including pigs, cane toads, wild dogs, feral cats and rabbits. Pig and cane toad records made up the majority of pest species records at cattle troughs with 41.59% and 39.37%, respectively. Feral cats which pose a potential threat to SBTF, made up only 2.54% of pest records. In addition, all feral cat records were between 5pm and 3am when SBTF are unlikely to be using the water source.

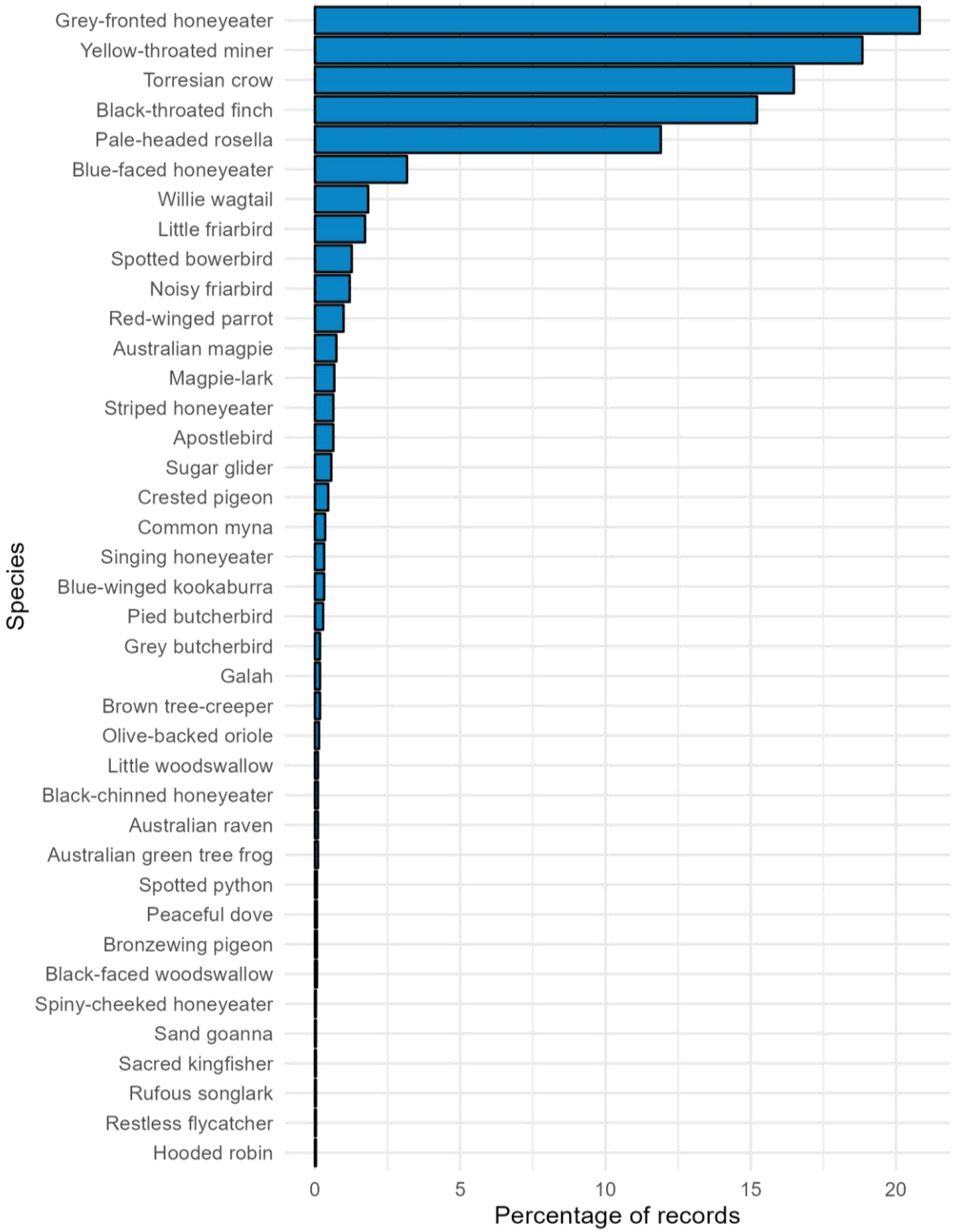


Figure 6.2-7: Percentage of species records from raised water sources

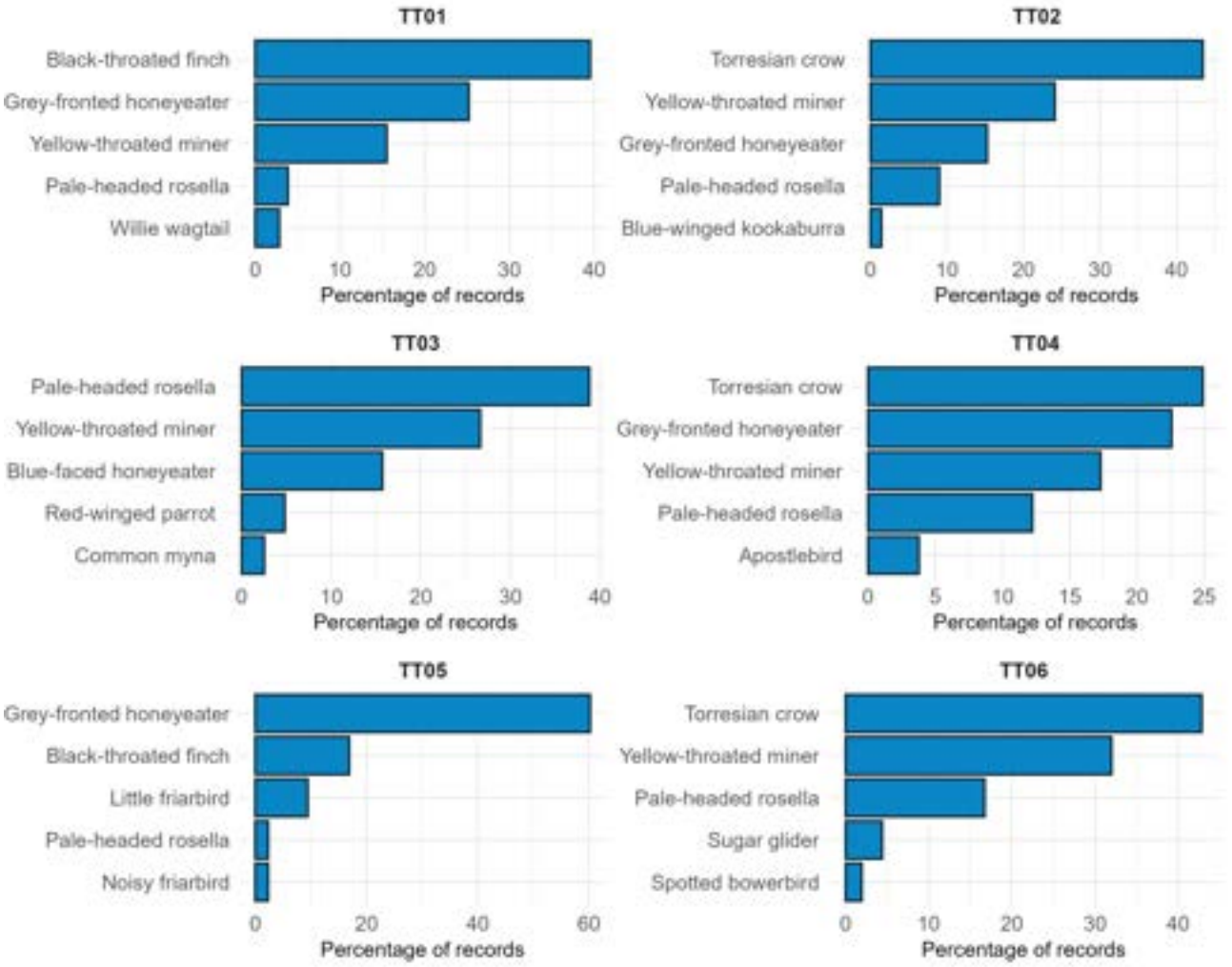


Figure 6.2-8: The five species which had the highest percentage of records for each raised water source

6.2.4 Discussion

The use of permanent artificial water sources by SBTF and species that potentially threatened SBTF were investigated to inform management actions for SBTF. Both methodologies of transect species count and remote camera monitoring of various artificial water sources have insight into whether the introduction of new artificial water sources is a beneficial land management strategy for SBTF.

6.2.4.1 1 km transects

The results of the 1 km distance transects found some species increased with proximity to permanent water ('increaser' species), while no species were found to decrease with proximity to permanent water ('decreaser' species). Increaser and decreaser species found in this study are different to those found in Harrington (2002), however comparison is difficult as different species were included in the studies' analyses. Notably, Harrington (2002) found the Australian magpie was an increaser species, while this study found no correlation with distance from permanent water. Fisher et al. (1972) classed the Australian magpie as being between a water-dependent and water-independent species, meaning it was unable to be classed clearly as one or the other.

Granivorous bird species tend to be water dependent due to the dry nature of their seed-based diet (Harrington, 2002). Granivorous species analysed in this study showed differing statistical relationships between abundance and distance to permanent water. This includes those identified as increaser species (squatter pigeon and galah) and finch species with a slightly negative but statistically insignificant relationship to distance to permanent water (zebra finch, double-barred finch and SBTF). Similarly to Harrington (2002), not all water-dependant birds were found to be increasers, notably in the case of the granivorous finch.

The analysis did not find a significant correlation between SBTF abundance and distance to permanent water. However, the three finch species analysed all show a negative but statistically insignificant correlation between their abundance and distance to permanent water, potentially reflecting more of an 'increaser' than 'decreaser' species trend. This reflects the understanding that Estrildidae species (estrildid finches) are highly dependent on available water (Fisher, Lindgren and Dawson, 1972).

Studies have found the increased abundance of some bird species due to artificial water points and other impacts of grazing can contribute to competition with other species (Reid and Fleming, 1992; James, Landsberg and Morton, 1999). This analysis found the Torresian crow, which is considered to be aggressive or in potential competition with other species at water points, is an increaser species. This species has been found to display aggressive behaviour towards other bird species (Rogers *et al.*, 2020), and are known to depredate eggs from the nests of smaller birds (Maslo *et al.*, 2016).

Some studies have found artificial water points may negatively impact native species by attracting native and introduced predators (James, Landsberg and Morton, 1999; Harrington, 2002). The whistling kite, a native predator, was found to be an increaser species despite being classed as water independent by Fisher et al. (1972). The observed relationship may not be solely be due to the presence of water, but could instead reflect another factor associated with water points, such as the availability of food resources. Harrington (2002) notes that the influencing factors of distribution and abundance on non-water-dependent species is more complex than for water-dependent species.

6.2.4.2 Use of artificial water sources

Water sources closer to known SBTF habitat generally recorded more SBTF per day. The two raised water sources utilised regularly by SBTF during the study (TT01 and TT05) are both located in known SBTF habitat that has regular foraging and nesting records, with the nearest permanent water source 2.5 to 3 km away. SBTF are known to use ephemeral water sources near these tree troughs when it is available (i.e. mainly in the wet season).

The exception to this finding that a decreased distance to SBTF habitat leads to an increase in water source use is the water sources at 10 Mile Bore (TT03 and CWAT01a). Despite being in a core area of habitat area for SBTF, the species was not recorded at either of the water sources during the study. This is likely due to the range of alternative water source options within close proximity. This may indicate that SBTF will use both cattle troughs and raised water sources in areas of habitat they are already occupying; however, will choose alternative water sources such as dams when multiple water sources are present in close proximity (less than 3km).

A majority of the artificial water sources located in peripheral areas of SBTF habitat where SBTF records are sparse were not observed being used by SBTF other than TT02. This may indicate that SBTF will not travel into new habitat to use additional artificial water sources; however, the time frame of this study may be too short to capture this movement. The records at TT02 may indicate the beginning of a movement into these new areas.

When looking at water sources records from species other than SBTF, the vast majority of raised water sources use has been from aggressive birds, including yellow-throated miner, grey-fronted honeyeater and

torresian crow. Water source installation will need to consider the potential deleterious effect of potential increases in the local abundance of aggressive species, such as miners and friarbirds, which may exclude granivores (Westgate *et al.*, 2021). However, TT01 had the highest number of SBTF records while also having a high number of grey-fronted honeyeater and yellow-throated miner records. This study's scope does not extend to assessing whether adding more water sources would lead to an increase in aggressive species. However, these results may suggest that use of raised water sources by SBTF may be related to distance to habitat more than to use by aggressive species.

Despite the presence of aggressive species, the design of the raised water sources was successful in excluding pest species as none were recorded. At cattle troughs a variety of pests were recorded; however, this made up a small proportion of records. Records of feral cats were rare and always at night when SBTF are unlikely to be using the water source. However, the scope of this study is limited to species that can be recorded by the cameras. Feral cats use may still increase in the larger area surrounding the water source, particularly if surrounding habitat is impacted by grazing (McGregor *et al.*, 2014).

6.2.5 Conclusions and recommendations

This study has demonstrated that SBTF rely on permanent water sources and are willing to use a variety of artificial water sources. Although aggressive species made up a majority of records at raised water sources, the presence of aggressive species may not influence the use of water sources by SBTF. In addition, the 1 km transect data identifies that most aggressive and exclusionary bird species, such as yellow-throated miners, noisy friarbirds, pied butcherbird and Australian magpie are distributed approximately randomly through the environment with respect to distance to permanent water. These species are all generally large and highly mobile (Jenkins, 1981; Sheard *et al.*, 2020) and their habitat use is unlikely to be constrained by distances to permanent water which are generally less than 4 km in the study area.

SBTF was recorded using new cattle troughs and raised water sources at high detection rates, demonstrating that the species is adaptive to new water sources other than when there is a high density of existing water sources. Proximity to habitat appeared to primarily influence the use of water sources by SBTF; however, a raised water source at peripheral areas of SBTF habitat with few previous SBTF records had low detection rates of SBTF. This may indicate movement of SBTF into new habitat once a water source is established; however, a longer study is required to determine this confidently.

Effects of increased grazing due to water source installation (Fensham and Fairfax, 2008) was outside of the scope of this study; however, this can be mitigated by excluding cattle and macropods using fencing or raised water sources. Raised water sources also further mitigate the threat of aggressive species as Beggs et al. (2020) found that the impacts of aggressive birds are lessened in the presence of high mid-dense understorey, which is not provided in heavily-grazed artificial water source areas. Future studies should also investigate the use of wire exclusion fencing as a method to deter aggressive species. Similar wire meshing has been successfully implemented to protect feed trays for the golden-shouldered parrot on Cape York (Baker, 2014).

The relationship between species abundance and distance to permanent water is more complicated than what has been investigated in this analysis. Other factors related to artificial water points, such as concentrated grazing impacts on vegetation, presence of mammalian predators and weeds, and food resources (James, Landsberg and Morton, 1999; Harrington, 2002) are beyond the scope of this study.

6.3 Design guidelines for black-throated finch friendly water sources

The southern black-throated finch (*Poephila cincta cincta*) (SBTF) is an endangered bird that inhabits eucalypt-dominated open woodlands and woodlands with a grassy understorey in north and central Queensland (Zann, 1972; Mitchell, 1996; Higgins, Peter and Cowling, 2006; Commonwealth of Australia, 2023). These finches show a preference for habitats closer to permanent water sources (NRA Environmental Consultants, 2005; Isles, 2007; NRA Environmental Consultants, 2009; Rechetelo *et al.*, 2016a; van Osta *et al.*, 2024b). Like many other granivorous birds, SBTFs need regular access to surface water for drinking (Mula Laguna *et al.*, 2019).

6.3.1 What type of water sources do SBTF drink from?

Southern black-throated finch use both natural water sources, such as springs, creeks, and lakes, and man-made sources like farm dams and troughs. However, due to their small size (10 cm long, 15 g), they may be limited in the types of artificial water sources they can access. For example, some concrete troughs may not be suitable if the edges are too wide for birds to grip or if the water level is too low for them to reach (Figure 6.3-1). Additionally, birds are less likely to use water sources that require crossing large, open distances without cover, as this increases their risk of predation by raptors (Figure 6.3-1).

Several factors influence the SBTF's ability to access water, including:

- A suitable perch for the birds to grasp with their feet.
- The "dipping distance," or the distance from the edge of the structure to the water level.
- The proximity of woody vegetation (trees or shrubs) to provide cover and reduce predation risk as birds fly to and from the water.
- The reliability of the water source.



Figure 6.3-1: Water sources that restrict southern black-throated finch access. Thick edged concrete troughs (left) and no tree cover near the water's edge (right).

6.3.2 Guidelines for southern black-throated finch friendly water sources

Providing reliable and accessible water sources in eucalypt woodlands allows SBTFs to forage and breed in these areas. Landowners and managers should consider the design and suitability of water sources when installing new ones within potential SBTF habitats. Where possible, ensure that water is safely accessible. If existing water sources are not suitable for SBTFs, they may be retrofitted to improve their accessibility.

Table 6.3-1: Design guidelines for southern-black throated finch friendly water sources

Water source requirement	Installing new water sources	Retrofitting existing water sources
Access to the water's edge	• Troughs with a narrow edge (<3 cm thick) are easier for small birds to perch and dip to water. • Set float valve to keep water level within 5 cm of the lip of the water source	• Retrofit steel grid or mesh • Fix a log or piece of timber

Water source requirement	Installing new water sources	Retrofitting existing water sources
Predator avoidance	Install water within 5–10 m of existing trees.	Plant trees, such as *Eucalyptus* or *Melaleuca* species within 5–10 m of existing water sources.
Water reliability	Southern black-throated finch require year round access to water, ideally within 3 km of open grassy woodlands in which they feed.	



Figure 6.3-2: Examples of water trough retrofits that are southern black-throated finch friendly.

7 Additional research

7.1 A review of southern black-throated finch survey and monitoring methods

7.1.1 Introduction

Effective management of threatened species requires tools to understand their distribution and abundance through space and time (Allen & Singh, 2016; Fraser et al., 2018). However, the rare and cryptic behaviour of many threatened species often hinder effective survey and monitoring efforts (Lambert and McDonald, 2014). Monitoring threatened bird species is particularly challenging due to their vagility and the spatiotemporal unpredictability of their occurrences, which complicates reliable abundance assessments and species detection (Legge *et al.*, 2018).

The southern black-throated finch (SBTF; *Poephila cincta cincta*) is a nationally Endangered bird that inhabits open-grassy woodlands of north-eastern Australia (Higgins, Peter and Cowling, 2006). The SBTF population has declined by over 80% since European colonisation and the species' is now restricted to two population strongholds (Mula Laguna *et al.*, 2019; Commonwealth of Australia, 2023). The SBTF populations of both populations remain threatened by land clearing for grazing, mining and urban land development as well as ongoing degradation of habitat through exotic species, overgrazing and inappropriate fire regimes (Vanderduys *et al.*, 2016a; Commonwealth of Australia, 2023). Effective methods are therefore required to both survey for the occurrence of SBTF and monitor their population, which contributes to the conservation assessments and actions (Legge *et al.*, 2018).

One of the SBTF stronghold populations occurs in the Desert Uplands bioregion of central Queensland (Mula Laguna *et al.*, 2019; Commonwealth of Australia, 2023). This population persists in large tracts of uncleared open-grassy woodland that have historically had low grazing pressure and agricultural modification (Kutt and Woinarski, 2007). Since approximately 2009, the southern Desert Uplands area has been explored and to a lesser extent developed for coal production associated with the Galilee Basin (Vanderduys *et al.*, 2016a). Environmental impact assessments for proposed coal substantially increased survey effort for the Endangered SBTF within the Desert Uplands, along with other protected environmental matters (Austecology, 2011; Amec, 2012; GHD, 2012). To date, only one of the coal mines, Carmichael Coal Mine, has been developed. Environmental approval requirements for the Carmichael Coal Mine required the implementation of a SBTF monitoring and research program, as well as the establishment of biodiversity offsets, a substantial component of which include land-based offsets for the SBTF (EcoLogical Australia, 2023). The surveys and monitoring conducted during the environmental impact assessment and subsequent programs at the Carmichael Coal Mine have yielded valuable insights to enhance future survey and monitoring efforts.

This study aims to review 12 years (2012–2024) of survey and monitoring methods for the SBTF, encompassing at least 38 survey events conducted by six different organisations, primarily consultancies engaged by the Carmichael Coal Mine proponent, Bravus Mining and Resources. More broadly, the study seeks to contribute to best-practice monitoring methods for the SBTF and offer a case study with wide applicability to survey and monitoring efforts for other threatened species.

7.1.2 Methods

Seven monitoring methods were used throughout the 12 years of monitoring included in this case study. In addition, three methods of population estimates were tested as a part of the BTFRP to determine the most effective method for future population monitoring. Some methods were used consistently throughout the case study, whereas other methods were discontinued or replaced by a method with similar purpose (Figure 7.1-1). Additional survey effort was undertaken for the purposes of BTFRP from 2020 to 2023. Broadly, survey methods have evolved from the early efforts (2012–2019), which focused on understanding the species' distribution across the study area, to the period from 2020 to the present, where survey focus has shifted toward repeated monitoring to assess population changes.

Each monitoring method, as well as their purpose, use throughout the study and associated data analysis is described in Sections 7.1.2.1 to 7.1.2.5, below.

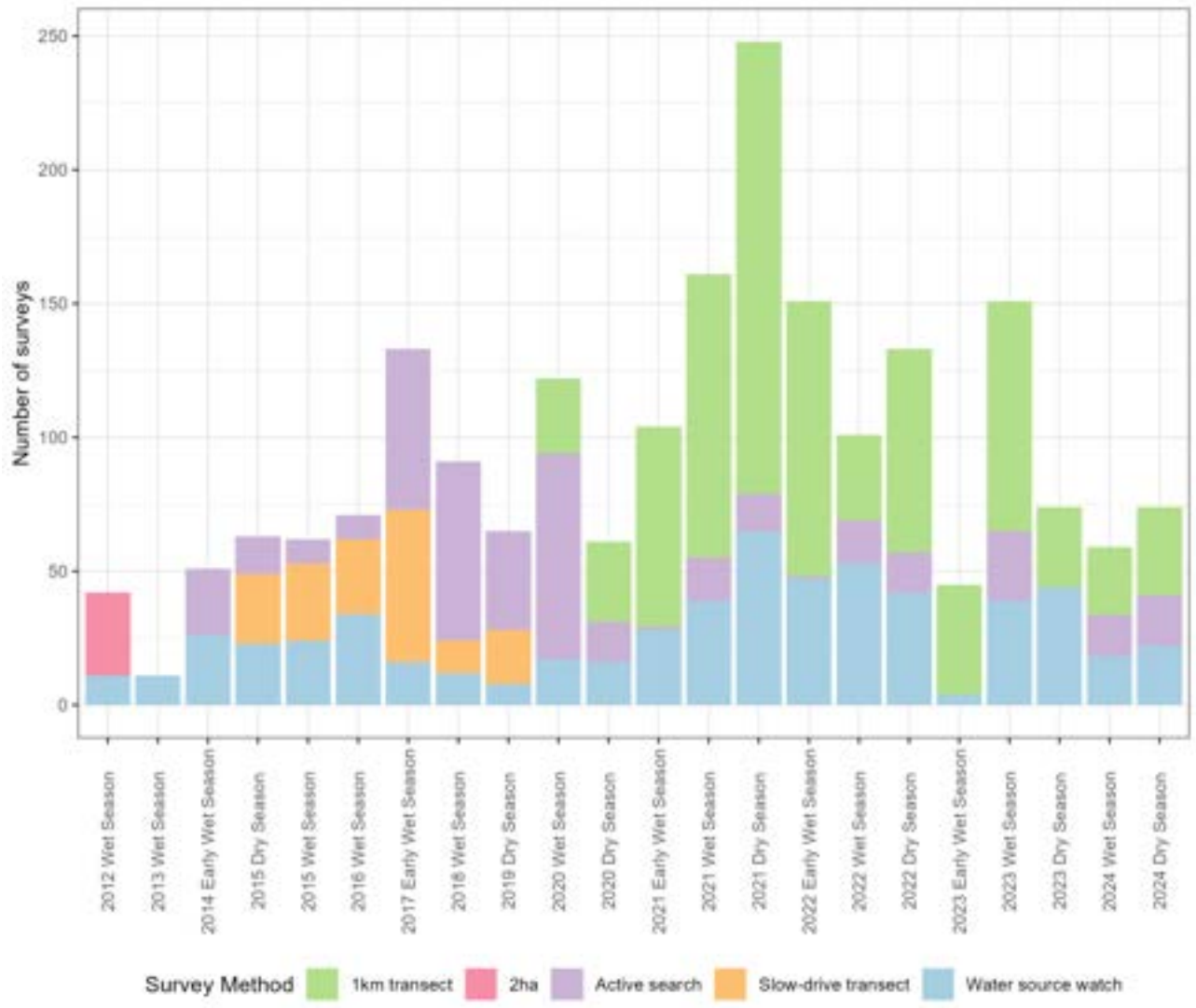


Figure 7.1-1 Survey effort by method through the 12 years of monitoring. Survey periods where survey effort was not recorded have been excluded.

7.1.2.1 Water source watches

Fixed point water source watches are the most consistently used method throughout the 12 years of monitoring. Although the timing and length of water source watches varied between surveys, most began at first light and lasted either 3 or 6 hours (Table 7.1-1). In 2023, the timing of water source watches was shifted to begin at 9 am, to focus on periods near the heat of the day when drinking rates are likely to be highest (van Osta *et al.*, 2024b). During the 2024 dry season survey, a 'census' style water source watch survey was trialled, where water source watches were conducted simultaneously to get a snapshot understanding of SBTF use of water sources at a single point in time. This was paired with the use of remote cameras on troughs to survey all water sources simultaneously. Full methodology of the census is detailed in Section 3.5.3.

Table 7.1-1: Duration and timing of water source watches during case study

Survey	Duration and timing of water source watches
Wet season 2012 Wet season 2013 Dry season 2013	Duration varied from 0.3 to 5.5 hours and watches occurred throughout the day.
Wet season 2014	3 hours following first light and 1 hour in the late afternoon (3pm to 6:30pm)
Wet Season 2015 Dry Season 2015 Dry Season 2016	3 hours after first light
Dry Season 2016 Wet Season 2017	6 hours after first light
Dry Season 2018 Dry Season 2019	3 hours after first light
Wet Season 2020 Dry Season 2020 Wet season 2021 Dry season 2021 Wet season 2022 Dry season 2022	6 hours after first light within mining lease 3 hours after first light within offset area
Wet Season 2023 Dry Season 2023 Wet Season 2024	3 hours from 9am to 12pm
Dry Season 2024	All water sources monitored simultaneously for 6 hours from 9am to 3pm

7.1.2.2 Remote cameras

Remote cameras have supplemented water source watches during every monitoring survey since 2012. Remote cameras were installed primarily to improve spatial coverage of monitoring sites at water sources with few or no previous SBTF records, in more remote sections of the offset area with less monitoring effort, or to monitor SBTF use of cattle troughs, which are better monitored using remote cameras. However, the locations and duration of remote camera placements varied among surveys challenging year-to-year comparisons of remote camera results.

In 2024, a range of changes were implemented to introduce remote cameras as a monitoring method with results that could be compared year-to-year. To reduce variability in the camera set up, a permanent post that could not be moved by cattle was installed beside each cattle trough and custom fixed angle bracket mounts were manufactured, and the camera model (Reconyx HF2X Hyperfire 2) and settings were standardised. In addition, the permanent locations of camera sites were selected, which focussed on cattle troughs that can be more thoroughly covered by a remote camera than a larger water source.

From 2012 to 2023, data analysis generally consisted of photos being manually reviewed by an ecologist. However, to manage the large number of photos generated from remote camera deployment following its implementation as a core part of the monitoring program, a convolutional neural network image detection model was developed in 2024. The model assists in pre-labelling remote camera photos collected during the 2024 wet season survey, allowing ecologists to prioritise QA to images that potentially contained a bird.

A major limitation of remote cameras is that they may capture multiple images of the same individuals during a detection event. Due to this, the number of photos of SBTF couldn't be compared to records from other methods such as water source watches or active searches. From 2022 to 2024, a five-minute window was defined as a detection event to counteract this. Within this timeframe, the maximum number of SBTF observed in any single photo was used in the analyses (Green *et al.*, 2023). This duration was selected to balance the risk of data non-independence while providing a reasonable estimate of the number of SBTF using the water source (Green *et al.*, 2023). A calibration exercise was undertaken during 2024, which involved simultaneous remote camera and human water source watches. Results of the two methods were most similar when a 10-second window was used to calibrate the remote camera results so this was used as the defined detection event for data analysis. Further details of calibration trials are presented in Section 3.5.3.

7.1.2.3 Active searches

On-foot active searches following a random meander style approach (Cropper, 1993) have consistently been part of monitoring in conjunction with water source watches since 2012; however, the methodology has varied slightly in duration and timing. Active searches consisted of random meanders targeting areas of suitable habitat for SBTF within a set radius of a water source. From 2013 to 2018, active searches were undertaken within 600 m of a water source without a set duration or timing; however, they usually occurred during water source watches (Table 7.1-1). In 2019, methodology changed to require active searches to be completed within 1 km of a water source for three continuous hours beginning at dawn, immediately prior to the water source watch at that location. Call playback was used during active searches from 2021 to 2022 before being discontinued due to limited success and potential influence on natural SBTF behaviour.

7.1.2.4 Standardised area searches

20-minute (minimum) searches of pre-determined 2-hectare plots (Barrett *et al.*, 2003) was the first standardised search methodology used to monitor foraging and breeding habitat, from 2012 to 2014. Two searches were undertaken at each site at different times of day; however, generally one search occurred within 3 hours of dawn to maximise chances of detection.

2-hectare searches were replaced by slow drive transects with call playback stations in 2015 with the aim of continuing to systematically monitor SBTF movement and habitat while sampling a larger portion of the study area. Slow drive transects traversed accessible tracks for 3 km through suitable SBTF habitat at approximately 10 to 20 km/hr. Three call-playback stations were positioned along the transect where 10 min call-playback sessions were conducted.

Slow drive transects were found to be ineffective and were then replaced with 1 km transects in 2020. At 1 km transect sites, the observer would slowly walk along the length of a pre-determined 1 km linear transect, searching for SBTF and their nests. Permanent 1 km transects were created, including one transect beginning at every water source and 15 transects randomly placed within woodland habitat. 1 km transects associated with water sources were completed at dawn at the beginning of each active search, while 1 km transects not associated with water sources had no set time to be completed.

7.1.2.5 Bioacoustics

Bio-acoustic recorders (BARs) have been used within the study area to investigate SBTF occupancy since 2016. Initial pilot surveys assessed whether automated recognition of SBTF calls was a viable method to detect SBTF presence, particularly for sites that are impractical to monitor by an ecologist due to the scarcity of SBTF use or their remote location. Following this successful pilot trial, eight solar powered BARs were deployed between 2020 and 2023 to investigate relationships between SBTF occupancy with seasonal water availability and fire. Each BAR collected audio data from 6:00 am to 6:00 pm daily.

Development of the SBTF call recogniser was completed during the reporting period as part of the SBTF research program (van Osta *et al.*, 2023, Section 7.4). As described in van Osta *et al.* (2023), the SBTF call recognition model has classification abilities comparable to an ecologist with substantial experience surveying for SBTF and is suitable for the analysis of extensive audio datasets.

7.1.3 Results

An analysis of 12 years of monitoring data revealed substantial variation in SBTF detection rates across the survey methods, as shown in Figure 7.1-2, which presents the mean number of SBTF recorded per survey event. Water source watches have consistently been the most effective method at detecting SBTF with an average of 12.8 SBTF recorded per survey event. In comparison, 2-hectare searches, 1 km transects, active searches and slow drive transects had similar but significantly lower detection rates with an average of 3.06, 2.33 1.52, and 1.11 SBTF recorded per survey event, respectively.

Of the methods not focussed on water sources, 2-hectare searches had the highest detection rate, which is particularly high given each search was undertaken in approximately 20 minutes. In comparison, 1 km transects and active searches had similar detection rates but the average length of each survey event was much longer, approximately 40 and 150 minutes, respectively. However, the sample size of 2 ha searches is much smaller as survey effort was only recorded for one survey event (2012 wet season) so the results may be influenced by seasonality or observer bias more than other methods that were sampled across multiple seasons, years and organisations (Table 7.1-2).

Slow drive transects were found to have the lowest detection rate with an average of 1.11 SBTF recorded per survey event, despite each survey event having an average duration of approximately 100 minutes.

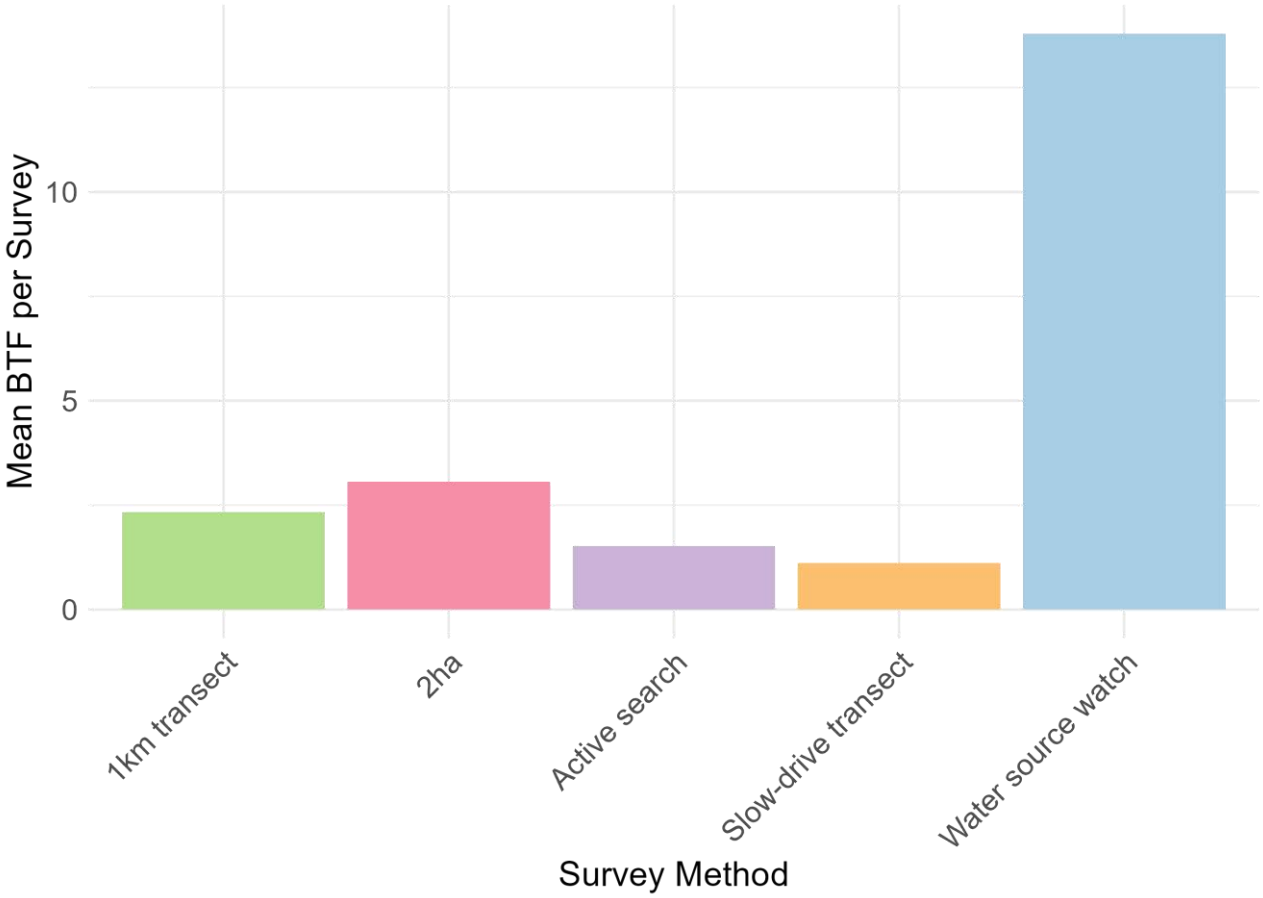


Figure 7.1-2 The average number of SBTF recorded per survey for monitoring methods utilised during the monitoring program

Table 7.1-2: Sample sizes of survey events and survey periods used to calculate SBTF detection rates

Survey Method	Number of survey events	Number of survey periods
1 km transect	834	13
2 ha search	31	1
Active search	437	18
Slow-drive transect	172	6
Water source watch	599	22
Total	**2073**	**60**

7.1.4 Discussion

There are many factors that influence the design of an effective monitoring program as each survey method differs in effectiveness, variability, survey effort and cost. Water source watches are a key survey method recommended in the *Significant Impact Guidelines for the Endangered Black-throated Finch (southern subspecies; Poephila cincta cincta)*, particularly in the dry season (DEWHA, 2009b). This method has proven to be the most effective in consistently detecting SBTF during the surveys.

A key challenge of water source watches is the changing availability of permanent and ephemeral water sources over time. Seasonal fluctuations in ephemeral water can be managed by conducting monitoring under specific conditions, particularly in the late dry season when ephemeral water is typically absent. However, the addition or removal of permanent water sources may either concentrate or disperse SBTF across available water sources, influencing the recorded rates of SBTF use in water source watches and remote camera surveys. The introduction of the 'census' style water source watch survey in 2024 aimed to address these issues and provide a rough estimate of SBTF population that could be compared year to year. Table 7.1-4 outlines the suitability of water source watches to achieve certain monitoring and detection criteria.

Although water source watches give a good insight into broad areas of SBTF habitat across a landscape, use of foraging and nesting habitat is challenged by the species low densities away from water sources. Monitoring of SBTF in their foraging and nesting habitat is important to understand their use of habitat that is at most risk of degradation (Franklin, 1999) and provide an estimate of SBTF abundance that is independent of water availability. Results can therefore be compared between years with varying water availability, which is a key limitation of water source watches. However, monitoring results from the study area show that survey methods that focus on foraging and nesting habitat have lower detection rates than compared to methods that target water sources. As a result, substantial survey effort may be required to implement a long-term monitoring program using 2-hectare searches, active searches, slow drive transects, and 1 km transects (Table 7.1-4).

High variability in results due to survey methods can make detecting true changes in populations difficult (Humbert *et al.*, 2009). Although inter-observer variation is inherent among many human-based methods, the variation is particularly impactful when the methods have low detection rates (Verner and Milne, 1989). Searcher efficiency may be increased by focussing search effort on areas of localised bare ground within open grassy woodlands, such as draining features, lightly used tracks and recently burnt areas (EcoLogical Australia, 2023). Investigating mixed flocks, such as those containing black-faced woodswallows, is also likely to increase searcher efficiency (EcoLogical Australia, 2023); however, implementing these searcher efficiency improvements must be conscious of minimising inter-observer and inter-survey methodological variation.

Slow drive transects were introduced to the monitoring program as a method to counteract this low detectability but increasing the spatial coverage of surveys. Slow drive transects were found to be ineffective for recording SBTF due to consistently low detection rates despite the high spatial coverage. In addition to having the lowest detection rate of all methods, a major limitation of this methodology is that surveys are confined to tracks which limits access to suitable habitat and often include areas of unsuitable or marginal habitat. Tracks in the Desert Uplands are generally created for agricultural purposes, such as maintenance of artificial water sources, where grazing impacts are likely to be greatest (James, Landsberg and Morton, 1999; Fensham and Fairfax, 2008). In addition, SBTF behaviour and call detectability was found to be influenced by vehicle noise. Burn history was an additional potential confounding effect on slow drive transects, which was identified by E2M following burns in July 2023. During a SBTF monitoring survey in October 2023, it was apparent that SBTF were less commonly flushed while driving along tracks within and near the burn area than on previous surveys, despite the water source watch data showing the continued presence of a high density of SBTF within the area. We suggest

that the low-intensity burns increased the areas of bare ground, which are the preferred foraging microhabitat of SBTF, which reduced SBTF's use of trackside microhabitat and thus reduced the number of foraging SBTF flocks observed while driving.

Call playback was also trialled as a method to improve detectability during slow drive transects and active searches. Call playback was a useful monitoring technique implemented in the initial years of the monitoring program, to assist in detecting SBTF in areas where their presence was uncertain. As SBTF habitat use across the study area became better defined, call playback became less useful and across both methods produced consistently low records. In addition, the method had the potential to disrupt natural behaviour and breeding activities (Debus, 1995).

Technology integration into ecological monitoring is a rapidly improving and allows for monitoring to span larger spatial and temporal scales (Marvin *et al.*, 2016; Gibb *et al.*, 2019). Survey methods that utilise technology such as remote cameras or BARs have the lowest observer variability (Kułaga and Budka, 2019; Roberts, 2011). In addition, low detectability of species can be counteracted by deploying devices for longer periods of time for a lower cost than methods that solely rely on human survey effort (Kułaga and Budka, 2019; Roberts, 2011). Remote cameras and BARs have been consistently deployed at a small scale throughout the monitoring program until 2024 when remote cameras were introduced to replace regular water source watches at cattle troughs. This change allowed a greater number of water sources to be surveyed for a longer period which led to SBTF being recorded by remote cameras at water sources where human-based water source monitoring did not detect SBTF during the survey period. In addition, it allowed for all water sources to be simultaneously surveyed as a part of the census which will provide comparable long term population data which was not produced from previous water source watches. These survey methods attain most of the monitoring criteria outlined in Table 7.1-4. However, remote cameras and BARs have their limitations. Remote cameras have a small field of view than compared to humans, which makes them unsuitable for use on large water sources like dams. In addition, both remote cameras and BARs are not able to separate the same individual being recorded multiple times as effectively as a human observer, making it difficult to compare to the results of other methods. The ability to detect individuals is complicated for remote cameras by the behaviour of SBTF to travel between water sources and nearby trees multiple times during a single drinking bout (Authors pers. obs.). Remote cameras are not able to distinguish a returning individual from a new individual, unlike a human-based water source watch in which the observer will be able to view the entire flock at once, which will often lead to an over-estimation the total SBTF recorded. However, this limitation can be mitigated by conducting a simultaneous human observation to calibrate the results, as detailed in Section 3.5.3.

The cost of large-scale remote camera and BAR deployment is often in the data processing. Manual identification of SBTF calls or photos is feasible at small scales, particularly with tools such as Megadetector (Beery, Morris and Yang, 2019) to remove false positives. However, at large scales, specific-specific AI models need to be developed. Two call detection models for SBTF are available (as of November 2024; (van Osta *et al.*, 2023; Allen-Ankins, Laguna and Schwarzkopf, 2024); however, application requires proficiency in either Python or R coding languages. Even with these detection models, the output still requires quality assurance which requires proficiency with the species call to discriminate from potential false-positive species.

A summary of methods discussed above, as well as some methods discussed in other sections of this report, is presented in Table 7.1-3.

Table 7.1-3: A review of survey and monitoring methods for the southern black-throated finch (SBTF)

Method	Objective of method	Temporal variation	Methodological considerations	Utility of methods
Water source watches	Suitable for detection of the species occurrence and as a long-term monitoring method provided seasonal timing of surveys are standardised.	High seasonal variation due to changes in the abundance of ephemeral water and temperature/humidity affecting drinking rate. Most reliable in the mid-late dry season when ephemeral water sources are not available. Cool days with dew should be avoided as drinking rates will decline (Authors pers. obs.).	A simple method that affords low surveyor variation, provided surveyors are familiar with the area's avifauna. Concentration effect of SBTF at water sources resulting in high detection rates compared to methods that search for SBTF in potential foraging and nesting habitat. The method can be used to identify the broad areas of species habitat within a site; however, the distribution of foraging and nesting habitat cannot be ascertained. Water source watches should focus on periods near the heat of the day when drinking rates are likely to be highest (van Osta *et al.*, 2024), broadly from 9 am to 3 pm.	**Moderate Reliability** Based on the water source watch census in the 2024 dry season, this method has produced a density estimate of 0.02 individuals per hectare (95% CI). Water source watches have consistently been the most effective method at detecting SBTF with an average of 12.8 SBTF recorded per survey event. Water source watches are a moderately reliable approach for detecting SBTF presence, particularly during the mid to late dry season when individuals concentrate at permanent water sources. The method benefits from high detection rates and low observer variability, making it suitable for long-term monitoring and seasonal trend analysis.

Method	Objective of method	Temporal variation	Methodological considerations	Utility of methods
DISTANCE transects and standardised area searches (e.g. 2 ha)	On-foot searches over standardised areas provide a potential long-term monitoring method for SBTF, provided sufficient replication is achieved. Buckland *et al.*, (2015) recommend that at least 60 to 80 flocks are recorded for each period to estimate population density, which may not be feasible for sites where SBTF are sparsely distributed. While standardised line and area searches have a low detection rate of SBTF they may also be used to detect the presence of SBTF within potential foraging and nesting habitat; however, substantial spatial and temporal replication is needed to ensure a robust result.	Seasonal variation in the call rates of SBTF may result in temporal variation in the species' detectability. Call rates are typically highest around the time of breeding and fledging periods, approximately December to June (van Osta, in prep.). SBTF also occur in smaller flock sizes during the breeding period, reducing the stochasticity of detection.	Detection rates are generally low due to the low abundance of SBTF within their foraging and nesting habitats. Surveyor familiarity with the species and especially the species call, will affect detection rates and potentially resultant estimates of SBTF density if unaccounted for.	**Low Reliability** Based on 154 survey events, this method has produced a density estimate of 0.03 individuals per hectare (95% CI: 0.02–0.07). DISTANCE transects and area searches exhibit low detection rates and high variability, primarily due to seasonal changes in calling behaviour and differences in observer skill. These limitations reduce their reliability for consistent population monitoring, making them more appropriate for assessing habitat use rather than informing management decisions.

Method	Objective of method	Temporal variation	Methodological considerations	Utility of methods
Active searches	On-foot searches, with a random meander style approach (Cropper, 1993) provide a detection method for SBTF, in order to assess species occurrence and habitat use. Active searches may be stratified by habitat type (e.g. distance to water) (McCaffrey *et al.*, 2014).	As per the standardised line and area search methods above, detection rates of SBTF may be affected by the species call behaviour and flock sizes.	High surveyor variation in the ability to detect SBTF makes this method challenging to implement in a long-term monitoring program. Searcher efficiency may be increased by focussing search effort on areas of localised bare ground within open grassy woodlands, such as draining features, lightly used tracks and recently burnt areas. Investigating mixed flocks, such as those containing black-faced woodswallows, likely to increase searcher efficiency.	**Low Reliability** Based on the 437 survey events, active searches have demonstrated low reliability for detecting SBTF, primarily due to high variation in observer skill and species detectability. While they offer flexibility and can be stratified by habitat type, their effectiveness is limited for long-term monitoring, making them more suitable for targeted occurrence and habitat use assessments rather than consistent population tracking or decision-making.
Bioacoustics	Acoustic recorders may be placed as temporary arrays for SBTF occurrence and habitat use studies, or permanent arrays for a long-term monitoring method.	Seasonal variation in the call rates must be accounted for to link the rate of SBTF call detections to the species abundance.	Two call detection models for SBTF are available (as of November 2024; (van Osta *et al.*, 2023; Allen-Ankins, Laguna and Schwarzkopf, 2024); however, application requires proficiency in either Python or R coding languages. Quality assurance of call detection model outputs requires proficiency with the species call to discriminate from potential false-positive species.	**High Reliability** Bioacoustic monitoring offers high potential reliability for detecting SBTF presence, especially when using validated call detection models and accounting for seasonal variation in vocal activity. With bioacoustics recording daily, this approach is well-suited for long-term monitoring and the analysis of seasonal trends.

Method	Objective of method	Temporal variation	Methodological considerations	Utility of methods
Remote cameras	Remote cameras may be placed deployed by water sources for short studies of SBTF occurrence or deployed systematically for a long-term monitoring method.	Similar to water source watches, method has high seasonal variation due to changes in the abundance of ephemeral water and temperature/humidity affecting drinking rate. Most reliable in the mid-late dry season when ephemeral water sources are not available.	Only able to monitor drinking sites and with a lower field of view than a water source watch so less practical for large water bodies. More scalable than water source watches as most cost is from initial cost of device or data management. Manual QA is a feasible option at small scales, particularly with tools such as Megadetector (Beery, Morris and Yang, 2019) to remove false positives. However, large scaling requires the development of species-specific AI models. Even with an AI detection model, QA is still influenced by human variation.	**High Reliability** Remote cameras offer high reliability for detecting SBTF, particularly when deployed during the mid to late dry season when birds are more likely to visit permanent water sources.

Method	Objective of method	Temporal variation	Methodological considerations	Utility of methods
Slow-drive transects	Travelling pre-determined tracks can give consistent sampling of a large spatial extent if SBTF detection is high enough.	As per the standardised line and area search methods above, detection rates of SBTF may be affected by the species call behaviour and flock sizes.	Despite high spatial coverage and ability to survey all habitat types, generally has low detection rates due to surveyor being confined to tracks that may limit access to suitable habitat. SBTF behaviour and call detectability are also influenced by vehicle noise. Results also vary highly based on surveyor and ground cover density due to influences such as fire.	**Low Reliability** Based on 172 survey events, slow-drive transects demonstrate low reliability for detecting SBTF, primarily due to low detection rates and limited access to suitable habitat from vehicle tracks. While they offer broad spatial coverage, their effectiveness is compromised by factors such as vehicle noise, dense ground cover, and high variability in observer skill. Consequently, this method is unsuitable for consistent population monitoring or informing robust management decisions.

Method	Objective of method	Temporal variation	Methodological considerations	Utility of methods
Mark recapture and resight	Population estimates can be used for monitoring population changes. Methods and results detailed in Section 3.5.1.	Mist-netting and reliable resighting of SBTF occurs at water sources with highest detectability in the late dry season when ephemeral water sources are not available.	Colour banding occurs during mist-netting meaning the sampling is limited to only drinking sites and has animal ethics and research permit considerations. Population estimates have a limited spatial extent depending on the location of resighting which can under-represent areas with low SBTF density.	**Moderate reliability** Based on a sample of 774 captured SBTF, this method has yielded a density estimate of 0.0011 individuals per hectare (95% CI: 0.009–0.012). Mark-recapture and resight techniques offer moderate reliability for estimating population changes within defined areas, particularly during the late dry season when detection rates peak. However, their effectiveness is constrained by limited spatial coverage, ethical and permit requirements, and dependence on resighting at drinking sites. As such, these methods are best suited for targeted population monitoring rather than broad-scale management decision-making.

Method	Objective of method	Temporal variation	Methodological considerations	Utility of methods
Genetics	Assessments of effective population size, genetic structure and population connectivity through gene flow can be used to monitor populations. Methods and results detailed in Section 3.2	Although mist-netting required to get blood samples is most successful in the dry season when flocks are larger and more concentrated around water sources, the results of the genetic studies do not vary seasonally.	Blood samples acquired through mist-netting means the sampling is limited to only drinking sites and has animal ethics and research permit considerations. A research institution capable of analysing samples and interpreting results is required. Although effective population size can inform management decisions, it is a measure genetic diversity and not a census population size. Regional genetic analysis of the species is required to define the exact spatial coverage of the estimate.	**Moderate Reliability** This method has provided a density estimate of 0.05 per ha (95% CI:0.04-0.06) from 158 DNA samples. Genetic monitoring provides moderate reliability for assessing SBTF population structure, connectivity, and genetic diversity, offering valuable insights into long-term population health. However, its application is limited by the need for mist-netting at drinking sites, ethical and permit constraints, and reliance on specialised laboratory analysis, making it more suitable for strategic conservation planning than routine population monitoring or short-term decision-making.

Method	Objective of method	Temporal variation	Methodological considerations	Utility of methods
eDNA	Potential for detection and monitoring of species occupancy. Methods and results detailed in Section 7.3.	Similar to other methods concentrated on water source use, method has high seasonal variation due to changes in the abundance of ephemeral water and temperature/humidity affecting drinking rate. Most reliable in the mid-late dry season when ephemeral water sources are not available.	This method only samples drinking sites and is only able to show presence or absence of the species. In our trial, detection was inconsistent failed to detect SBTF at water sources with high likelihood of SBTF use. Reliability was affected by high water turbidity, DNA degradation factors (e.g., UV radiation and temperature), and dilution in larger water sources.	**Low Reliability** eDNA monitoring currently shows low reliability for detecting SBTF, based on inconsistent detection results across 17 sampled water sources and its sensitivity to environmental factors such as turbidity, UV exposure, and temperature. While it offers a non-invasive method for presence/absence monitoring, its effectiveness is limited to drinking sites and remains unsuitable for informing confident management decisions without further refinement and validation.

Table 7.1-4: Selection table of SBTF detection and monitoring methods

Criteria. The suitability of the method to:	Active search	Water source watch	Acoustic recorders	Remote cameras	Distance sampling	Mark and recapture/ resight	Genetic studies
Monitor SBTF use of water sources	Moderate	High	Moderate	High[1]	Low	NA	NA
Monitor SBTF use of foraging and nesting habitat	Moderate	Low	High	Low	Moderate	NA	NA
Detect SBTF occurrence at low densities	Low	High	High	High[1]	Low	NA[2]	NA[2]
Scale to large sites and long-term monitoring programs	Low	Moderate	High	High	Low	Moderate	High
Afford high survey efficiency	Moderate	High	High[3]	High[3]	Low	Low[4]	Moderate[4]
Monitor population size	Low	Moderate	Moderate	Moderate	High	High[4]	High[4]
Provide demographic information (e.g. sex, age classes)	Low	Low	Low	Low	Low	High	High

[1] Remote cameras are a suitable method for small water sources only, which can be sufficiently covered within the camera's field of view.

[2] Mark and recapture/resight methods and genetic studies require the mist-netting of SBTF. As such, these methods are only suited to known populations of the species.

[3] Scaling the use of acoustic recorders and remote cameras requires the use of call and image detection models (e.g. van Osta *et al.*, 2023; Allen-Ankins, Laguna and Schwarzkopf, 2024) to achieve a high survey efficiency.

[4] Mist-netting SBTF can be laborious and is best undertaken following a site reconnaissance survey to understand the water sources that SBTF drink from and the approach that they use (i.e. their flyway between the vegetation and the water's edge). However, for genetic samples, a robust effective population size can be calculated from approximately 25 birds (Adam Stowe pers. com).

7.1.5 Conclusion

While there are pros and cons to each survey method, the choice of methods depends heavily on the specific objectives of the monitoring program. A combination of methods may even be required, depending on whether the aim is to determine presence/absence, track habitat use during foraging, evaluate access to water sources, gather genetic information, or estimate population size. It is particularly important to consider species-specific factors, such as seasonal behaviours or daily activity patterns, which can influence timing and approach. Additionally, methods with lower detection rates may be crucial for understanding and tracking habitat use, such as active searches in foraging and nesting areas. Technology-based monitoring offers the advantage of providing consistent data across large temporal and spatial scales, though it comes with limitations in terms of data processing and cost. Combining technology with human-based surveys, such as the census method, can be an effective strategy to address these challenges. Ultimately, a robust monitoring program over a large temporal and spatial scale often requires the integration of multiple methods to achieve comprehensive and reliable results, as is recommended for ongoing monitoring in the Project Area in Section 9.2.2.

7.2 A survey of southern black-throated finch presence and habitat characteristics within the Desert Uplands Bioregion

7.2.1 Introduction

The black-throated finch is a small granivorous bird with a disjunct distribution and distinct northern and southern subspecies are recognised. The southern subspecies (SBTF; *Poephila cincta cincta*) is listed as Endangered under Commonwealth and Queensland legislation due to significant declines over the past 30 years, including a range contraction of over 80% arising from current and past land-use practice (BTFRT, 2007; DAWE, 2009a).

Black-throated finch (southern) previously occurred between the headwaters of the Gilbert, Mitchell and Burdekin Rivers in north-eastern Queensland, to the north-west slopes of New South Wales in the south and up to 700 km inland from the coast (Ford, 1986b; Schodde and Mason, 1999; Higgins and Cowling, 2007; BTFRP, unpublished). Populations of SBTF have steadily declined throughout the southern extent of its former range. In NSW, SBTF were reported to be abundant in the north of the state from 1839 to 1840 and subsequently several records of the species were recorded near Inverell and Tenterfield in the 1960s to 80s (Ley and Cook, 2001; Higgins and Cowling, 2007). By the 1930s they were considered extinct in the Brisbane region and the last records in south-east Queensland were from the Noosa area in 1958 and Stanthorpe around 1980 (Higgins and Cowling, 2007). Declines continued in central Queensland in the latter part of the 20th century and SBTF are considered to have disappeared from the Rockhampton area in the mid-1970s with a further decline in numbers reported around Charters Towers (Higgins and Cowling, 2007). In the mid to late 1960s, the SBTF declined in the Mareeba area and the northern subspecies, *Poephila cincta atropygialis*, became more numerous.

The SBTF is now thought to be confined to two to six fragmented populations across the northern Brigalow Belt, Desert Uplands and the Einasleigh Uplands bioregions (Buosi *et al.*, 2021). The Townsville area was previously considered to be a stronghold for SBTF; however, declines have been apparent since 2006, likely as a consequence of habitat loss and fragmentation, with evidence that flock sizes have declined, and disappeared from several sites in the area (Laguna *et al.*, 2019; Garnett and Baker, 2020).

It is currently unclear whether the Townsville Plains or Desert Upland regions supports the greatest numbers of SBTF, due to limited survey effort across the Desert Uplands (DCCEEW, 2023). However, there is anecdotal evidence to suggest the Desert Uplands region may support a larger SBTF population, including SBTF flocks observed on Hyde Park Station to the north of the BTFRP Study Area (Section 3.2) and small flocks observed on Moonoomoo and Doongmabulla Stations (BTFRP, unpublished). However, there has been comparatively little research completed on this population compared to the Townsville population.

The Desert Uplands covers approximately 70,000 km^2 and consists of four sub-regions: Jericho, Cape-Campaspe Plains, Prairie-Torrens Creek Alluvials and Alice Tablelands (Morgan *et al.*, 2002). SBTF have been recorded in all sub-regions, with the Alice Tablelands considered to be of particular importance and believed to contain the largest population of SBTF (Laguna, 2020; DCCEEW, 2023). To effectively protect and manage the Desert Uplands SBTF population, it is important to understand their current area of occupancy and distribution of suitable habitat. This study aims to:

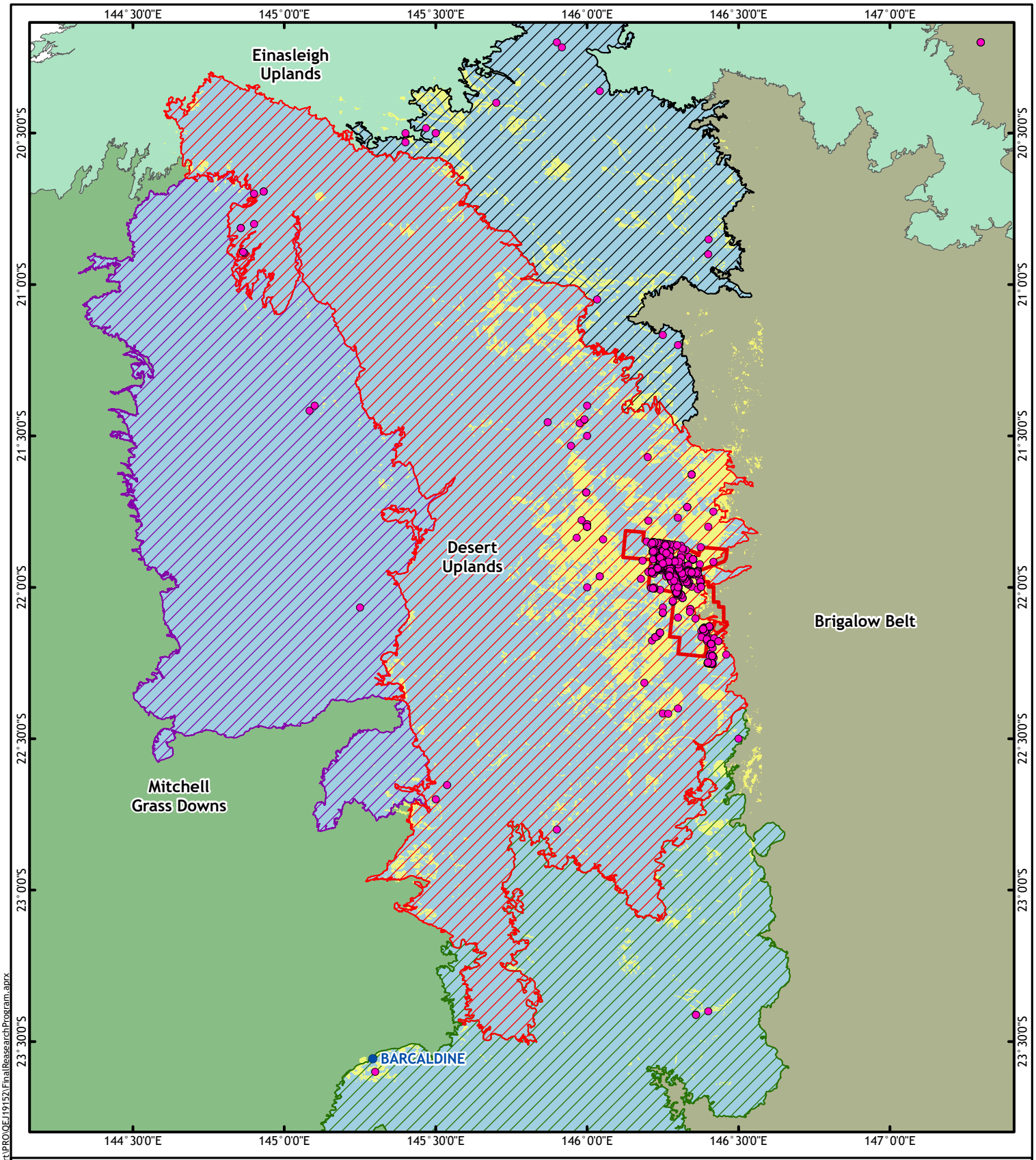
- Improve understanding of the extent and condition of SBTF habitat in the Desert Uplands
- Survey properties with potential SBTF habitat that have had little to no historical search effort
- Identify threatening processes to SBTF in the Desert Uplands; and

- Collect data on the occurrence of SBTF and habitat occupied by SBTF to refine habitat modelling.

7.2.2 Methods

7.2.2.1 Study area

The study area for this assessment is the Desert Uplands Bioregion in central Queensland which is approximately located between the townships of Blackall (in the south) and Pentland (in the north) (Figure 7.2-1). The Desert Uplands is characterised by sandstone ranges and sand plains with low fertility soils and eucalypt and acacia woodlands with a spinifex understorey (Sattler and Williams, 1999; Morgan *et al.*, 2002). The region has a semiarid climate with variable annual rainfall of approximately 460 mm. The predominant land use in the region is cattle grazing, with some sheep grazing in the west (Sattler and Williams, 1999; Bastin *et al.*, 2009).



Legend

- BTF Records
- Project Area
- Modelled BTF Habitat (>0.6)

Biogeographic Subregion

- Alice Tableland
- Cape - Campaspe Plains
- Jericho
- Prairie - Torrens Creeks Alluvials

Bioregion

- Brigalow Belt
- Desert Uplands
- Einasleigh Uplands
- Mitchell Grass Downs

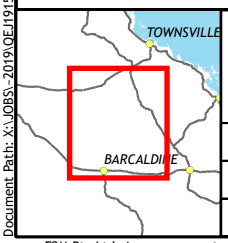
N

Scale 1:1,900,000 (A4)

0 32 64 96

Kilometers

Coordinate System: GCS GDA 1994



Notes:
Aerial Imagery: © ESRI 2021
Cadastre: © DNRME 2021
Watercourse: © Geoscience Australia 2012
Road: © DNRME 2021

Rev	Description	Drawn	Approved	Date
2	Issued for Use	JY	JMC	30/10/2025
0	Issued for Use	GO	JMC	08/08/2025

FIGURE 7.2-1: STUDY AREA

BTFRP Final Report
Bravus

Map Number	Job Number	Rev
1 of 1	QEJ19152	2

Document Path: X:\JOBS\-2019\QEJ19152\GIS\Final Research Program Report\PRO\QEJ19152\FinalReasearchProgram.aprx

E2M Pty Ltd gives no warranty in relation to the data (including accuracy, reliability, completeness or suitability) and accepts no liability for any loss, damage or costs (including consequential damage) relating to any use of the data in this map.

7.2.2.2 Habitat model

A habitat model was developed to determine priority areas to survey within the Desert Uplands. The model was developed using MaxEnt (Phillips and Dudík, 2008) which is a software program for modelling species distributions by inputting a set of environmental variables and species presence records to predict the probability the species will use a habitat (Phillips and Dudík, 2008; Elith *et al.*, 2011; Merow, Smith and Silander, 2013). MaxEnt generates a species distribution model by assessing the relative correlation of environmental variables to species occurrence records (Elith, Phillips, Hastie, Dudík, *et al.*, 2011).

7.2.2.2.1 Species records

Southern black-throated finch presence records used for the MaxEnt model were sourced from observations collected during field surveys undertaken across the Moray Downs property between 2011 and 2022 as well as the Atlas of Living Australia (ALA), the BirdLife Australia Atlas and the Wildnet databases. In total, 1,278 records were collated from these sources (Table 7.2-1). To reduce the sampling and spatial bias of these records from data collected from surveys across Moray Downs, as well as duplications between the database records, we spatially filtered the data to remove records within 2 km of each other using the spfilteR package (Juhl, 2021). The remaining 87 SBTF records were then used to create a species distribution model in MaxEnt.

Table 7.2-1: Records of southern black-throated finch collated from field surveys on Moray Downs between 2011 and 2022 as well as from the Atlas of Living Australia, BirdLife Australia Atlas and Wildnet databases

Source	Number of records
Moray Downs surveys	1,136
Wildnet	59
ALA	54
Birdlife Atlas	29
Total	**1,278**

7.2.2.2.2 Environmental variables

Environmental variables selected for the model were those considered as having the highest likelihood of influencing the distribution and abundance of SBTF based on field observations and literature (Vanderduys *et al.*, 2016b; Bradie and Leung, 2017). These variables include:

- Climatic factors (annual mean temperature, temperature seasonality, maximum temperature of warmest month, minimum temperature of coldest month, annual precipitation, precipitation seasonality, precipitation of wettest quarter and precipitation of driest quarter)
- Topographic factors (digital elevation model)
- Vegetation characteristics (regional ecosystem)
- Fire regimes (cumulative fire scars and time since last burn); and
- Water availability (Euclidean distance to perennial water sources and distance to all water sources).

Environmental variables and their data sources are summarised within Table 7.2-2.

Table 7.2-2: Environmental variables used in MaxEnt modelling

Variable name	Description	Source	Date range	Type
bio1_z55	Annual mean temperature	ANUCLIM (Xu & Hutchinson 2010)	1976 to 2005	Continuous
bio4_z55	Temperature seasonality	ANUCLIM (Xu & Hutchinson 2010)	1976 to 2005	Continuous
bio5_z55	Maximum temperature of warmest month	ANUCLIM (Xu & Hutchinson 2010)	1976 to 2005	Continuous
bio6_z55	Minimum temperature of coldest month	ANUCLIM (Xu & Hutchinson 2010)	1976 to 2005	Continuous
bio12_z55	Annual precipitation	ANUCLIM (Xu & Hutchinson 2010)	1976 to 2005	Continuous
bio15_z55	Precipitation seasonality	ANUCLIM (Xu & Hutchinson 2010)	1976 to 2005	Continuous
bio16_z55	Precipitation of wettest quarter	ANUCLIM (Xu & Hutchinson 2010)	1976 to 2005	Continuous
bio17_z55	Precipitation of driest quarter	ANUCLIM (Xu & Hutchinson 2010)	1976 to 2005	Continuous
Mixed mosaic	Digital elevation model mosaiced with the Burdekin catchment	Combination of Adani supplied lidar data at 10m resolution with 25m Lidar DEM downloaded from Qspatial (Department of Natural Resources, Mines and Energy 2019)	2012 and 2019	Continuous
Dry season Euclidean distance	Euclidean Distance raster of permanent water sources	Permanent water sources, supplied by Adani and mapped using dry season satellite imagery (Department of Natural Resources, Mines and Energy 2019)	Current	Continuous
Wet season Euclidean distance	Euclidean Distance raster of permanent and ephemeral water sources	Permanent water sources and DNRME mapped drainage features (Department of Natural Resources, Mines and Energy 2019)	Current	Continuous
Fire scar	Cumulative fire scars over the date range	Annual Fire Scars - Landsat, QLD DSITI algorithm (Terrestrial Ecosystems Research Network (TERN) 2020)	1997 to 2016	Continuous

Variable name	Description	Source	Date range	Type
Time since last burn	Time since last burn (years)	Annual Fire Scars - Landsat, QLD DSITI algorithm (TERN 2020)	1997 to 2016	Continuous
Groundcover	Seasonal (Spring 2020) groundcover - green ground cover fraction (in percent) +100	TERN (Department of Environment and Science)	2020	Categorical
Bare earth	Seasonal (Spring 2020) bare earth - bare ground fraction (in percent) + 100	TERN (Department of Environment and Science)	2020	Categorical
Persistent green	Spring 2020 - This product captures variability in persistent green cover at seasonal (i.e. three-monthly) time scales, forming a consistent time series from 1989 to the present season (minus 2 years)	TERN (Department of Environment and Science)	2018 - 2020	Categorical
Persistent green wet	Summer 2021 - This product captures variability in persistent green cover at seasonal (i.e. three-monthly) time scales, forming a consistent time series from 1989 to the present season (minus 2 years)	TERN (Department of Environment and Science)	2019 - 2021	Categorical
Wet bare earth	Seasonal (Autumn 2021) bare earth - bare ground fraction (in percent) + 100	TERN (Department of Environment and Science)	2021	Categorical
Wet groundcover	Seasonal (Autumn 2021) groundcover - green ground cover fraction (in percent) +100	TERN (Department of Environment and Science)	2021	Categorical
GTRE	Ground-truthed regional ecosystem	E2M ground-truthed regional ecosystem mapping. Where ground-truthed regional ecosystem was not available, Queensland regional ecosystem mapping (Department of Environment and Science 2020) was used.	Current	Categorical

7.2.2.3 Site selection and land access

Areas of potential high suitability SBTF habitat were defined as those with a MaxEnt score greater than 0.6. Properties containing large areas (>1,000 ha) of potential high value SBTF habitat were prioritised for field surveys. Properties with known recent SBTF records were not considered for the survey as we prioritised search efforts within properties that had historically few, if any surveys. Land access was subject to approval from landholders. Where contact could not be initiated or access was denied, alternative properties were selected. Access to five properties was granted by landholders and surveyed (Table 7.2-3; Figure 7.2-2). Surveys were also undertaken in roadside vegetation in areas within and adjacent to potential high value SBTF habitat.

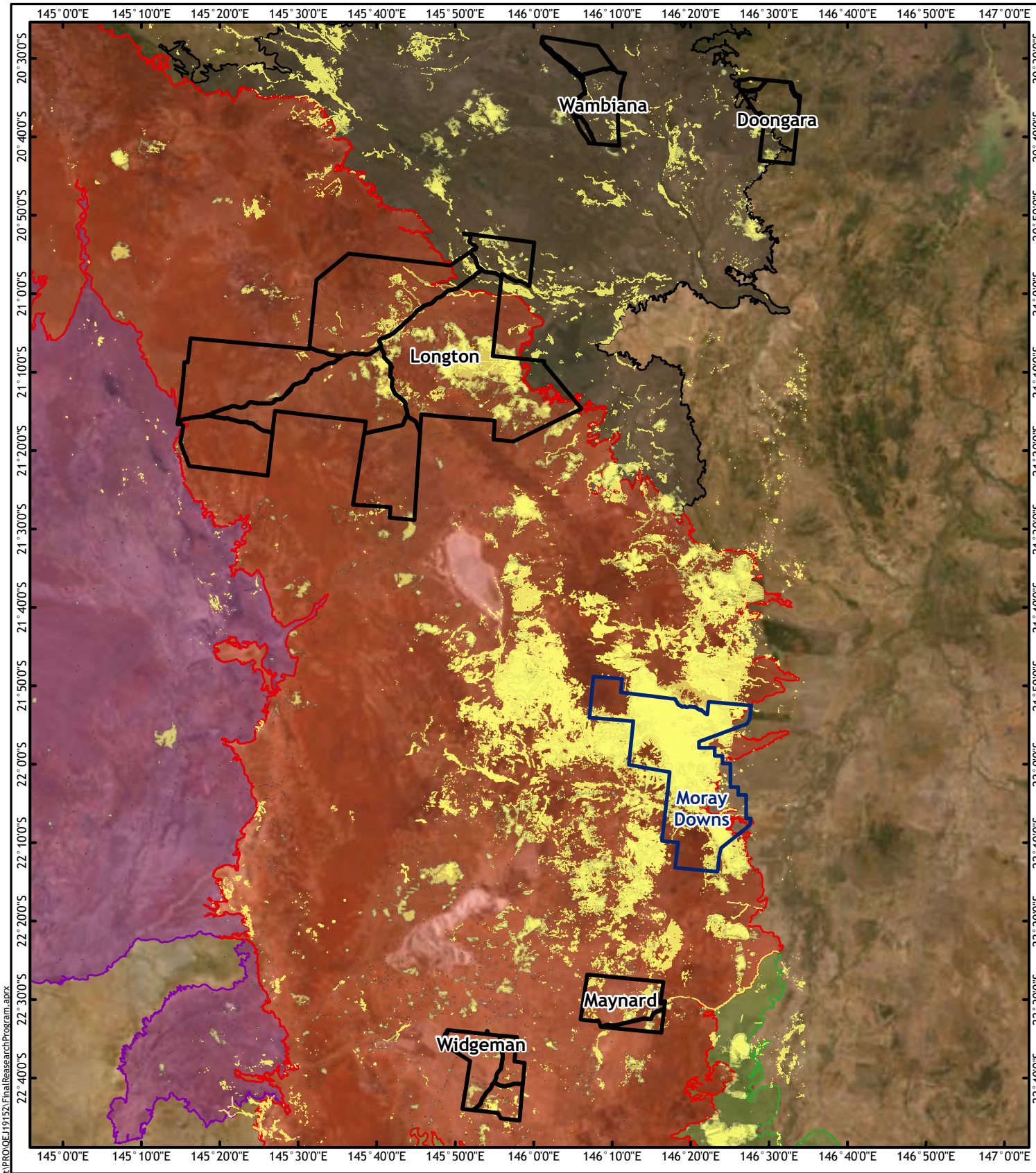
7.2.2.4 Field surveys

Field surveys were undertaken by two ecologists between 10 and 19 September 2024. The survey timing coincided with the mid to late dry season, when water availability is limited, thereby concentrating SBTF drinking at fewer, permanent, water sources. Weather conditions during the survey were fine and warm, with average minimum and maximum temperatures of 18.0°C and 31.6°C, respectively (BoM, 2024). No rainfall was recorded during the survey and rainfall over the preceding three months was well below average (BoM, 2024).

Surveys were undertaken over one to three days on each property and consisted of habitat assessments, active searches, water source watches and deployment of bioacoustics recorders (BARs) (Table 7.2-2; Figure 7.2-2). Survey effort and techniques was dependent on habitat suitability, property size and the availability and condition of access tracks. Active searches were conducted to search for the presence of SBTF within areas of modelled habitat. Searches consisted of random meander transects for 30 min per site through suitable SBTF habitat in the early morning and/or late afternoon when SBTF are typically more active. Water source watches were typically conducted over 1 hour at each waterhole during the warmer part of the day (late morning to mid-afternoon) when SBTF often travel to water to drink and bathe (Section 3.1). Other water sources were also watched for shorter periods of time whilst travelling through the properties.

Table 7.2-3: Properties accessed

Property name	Lotplan	Date/s accessed	Survey methods
Widgeman	3814PH116	10/9/2024	Habitat assessments, active searches
Maynard	1SP246199	11/9/2024	Habitat assessments, active searches
Wambiana	22DV770	14/9/2024 – 17/9/2024	Habitat assessments, water source watches, active searches, BAR deployment
Doongara	576SP274508	16/9/2024	Habitat assessments, active searches
Longton	2FX8	17/9/2024 - 19/0/2024	Habitat assessments, water source watches, active searches



Legend

- Properties Surveyed
- BTFRP Project Area
- Modelled BTF Habitat (>0.6)

Biogeographic Subregion

- Alice Tableland
- Cape - Campaspe Plains
- Jericho
- Prairie - Torrens Creeks Alluvials

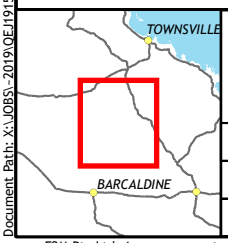
N

Scale 1:1,240,000 (A4)

0 21 42 63

Kilometers

Coordinate System: GCS GDA 1994



Notes:
Aerial Imagery: © ESRI 2021
Cadastre: © DNRME 2021
Watercourse: © Geoscience Australia 2012
Road: © DNRME 2021

Rev	Description	Drawn	Approved	Date
2	Issued for Use	JY	JMC	30/10/2025
0	Issued for Use	GO	JMC	08/08/2025

FIGURE 7.2-2: PROPERTIES SURVEYED

BTFRP Final Report
Bravus

Map Number	Job Number	Rev
1 of 1	QEJ19152	2

Document Path: X:\JOBS\-2019\QEJ19152\GIS\Final Research Program Report\PRO\QEJ19152\FinalReasearchProgram.aprx

E2M Pty Ltd gives no warranty in relation to the data (including accuracy, reliability, completeness or suitability) and accepts no liability for any loss, damage or costs (including consequential damage) relating to any use of the data in this map.

7.2.2.5 Desktop assessment of SBTF habitat clearing rates

Due to the large size of the study area, only a small portion could be surveyed during the ten day period. Due to this, a desktop assessment was undertaken to determine if the survey findings generally aligned with the condition of habitat throughout the entire study area. This assessment aimed to determine the proportion of SBTF habitat within the Desert Uplands that has been cleared since European colonisation and whether certain subregions have been affected more than others due to their proximity to townships and generally smaller property sizes.

DETSI remnant Regional Ecosystem (RE) and pre-clearing RE mapping (Version 13.1) (Department of Natural Resources and Mines, Manufacturing and Regional and Rural Development, 2025) was used as a basis to map SBTF habitat throughout the study area. Historical records were used to determine in which mapped pre-clear REs SBTF were commonly recorded to define SBTF habitat for the purpose of this assessment. Records were primarily sourced from BirdLife (BirdLife Australia, 2025) and environmental impact statements from projects within the study area. Records associated with the BTFRP were excluded to avoid bias towards REs present in the Project Area. Regional ecosystems that contained three or more historical records were used in the analysis as SBTF habitat. These included the following REs: 10.5.5a, 10.3.6, 10.7.11, 10.3.28, 10.5.4d, 10.5.2a, 10.7.4a, 10.5.1a, 10.3.15g, 10.5.1b, 10.3.12a, 10.3.14a, 10.3.5, 10.5.5x2, 10.7.7a and 10.7.9. The total area of SBTF habitat in the pre-clear mapping and current remnant mapping was calculated to determine the change from pre-European colonisation habitat to current habitat.

7.2.3 Results and discussion

No SBTF were recorded during the surveys. These results align with a reduction in SBTF records throughout the Desert Uplands. The survey identified that habitat quality for SBTF across the region was consistently poor, with most of the properties that were surveyed and habitat that could be observed from the roadside affected by one or more key threatening processes to SBTF, including:

- Habitat clearing and fragmentation
- Habitat degradation from grazing and alteration of fire regimes; and
- Invasion of exotic species.

These threats and the condition of habitat are discussed in more detail below.

7.2.3.1 Habitat clearing and fragmentation

Habitat clearing is the primary cause of SBTF decline to date (Black-throated Finch Recovery Team, 2004; Reside *et al.*, 2019). Black-throated finch are sensitive to clearing as they strongly avoid non-remnant vegetation (van Osta *et al.*, 2024b). Canopy trees, that cleared areas lack, are an important feature of SBTF habitat as they provide habitat for roosting and nesting as well as protection from predators (Zann, 1972). Due to this avoidance, clearing not only reduces the area of available habitat but can also fragment populations of SBTF. Genetic studies of the SBTF populations on Moray Downs (Section 3.2) and the Townsville Plains sub-bioregion (Tang, 2016) have found that landscape barriers of unsuitable habitat, such as cleared vegetation, can isolate SBTF populations over relatively small distances (less than 10 km). Fragmentation of populations can lead to reduced gene flow and a loss of genetic diversity that is vital for supporting the long-term persistence of species (Allendorf and Luikart, 2009; Lowe and Allendorf, 2010).

During field surveys, large tracts of cleared vegetation were recorded throughout the survey area, as depicted in Figure 7.2-3. Historical clearing of formerly suitable SBTF habitat was evident across the study area and was particularly notable in the Jericho subregion, where expansive clearing of *Eucalyptus melanophloia* and *Eucalyptus brownii* woodlands has occurred.



Figure 7.2-3: Regrowth vegetation on Fleetwood (left) and Jerico subregion (right)

Results of the desktop assessment (Table 7.2-4), demonstrate that SBTF habitat clearing since European colonisation within the Desert Uplands is generally lower than across the historical range of SBTF (Reside *et al.*, 2019). This aligns with findings that higher clearing rates in the Townsville Plains population has potentially resulted in the Deserts Uplands containing the larger portion of the remaining SBTF population and habitat (Reside *et al.*, 2019; Laguna *et al.*, 2019).

Clearing rates of SBTF habitat are broadly similar to the clearing rates of all remnant vegetation, indicating that SBTF habitat is not being disproportionally cleared within the Desert Uplands. Clearing rates are relatively consistent between subregions, with the exception of Jericho, in which close to 40% of remnant SBTF habitat has been cleared (Table 7.2-4).

Table 7.2-4: Percentage decrease in SBTF habitat within the Desert Uplands Bioregion

Subregion	Proportion of total pre-clear SBTF habitat within each subregion (%)	Decrease in mapped REs (%)	
		All remnant vegetation	SBTF habitat
Alice Tableland	43.28	14.09	13.48
Cape-Campaspe Plains	18.63	15.06	15.10
Jericho	21.71	40.21	39.29
Prairie-Torrens Creeks Alluvials	16.38	11.81	10.16
Total (Desert Uplands Bioregion)	**100.00**	**18.86**	**18.84**

7.2.3.2 Habitat degradation from grazing and alteration of fire regimes

In addition to the loss of habitat through clearing, there is also evidence of widespread impacts to habitat quality due to increased grazing pressure and altered fire regimes which are known to impact on granivores such as SBTF (Kutt and Woinarski, 2007; Woinarski and Legge, 2013). Despite the presence of a suitable tree layer, the ground layer of much of the potential SBTF habitat that was surveyed was heavily

impacted by cattle grazing. Grazing pressure from native and domestic animals plays an important role in the composition and diversity of native vegetation communities (Lunt *et al.*, 2007; Lebbink, Dwyer and Fensham, 2021). As a granivore, SBTF rely on species diversity as well as key foraging grass species to ensure availability of sufficient food sources throughout seasonal fluctuations (Crowley and Garnett, 1999a; Dostine and Franklin, 2002; Franklin *et al.*, 2005). Grazing can shift species composition toward more grazing tolerant species, altering ground layer composition and increasing invasion rates by grazing tolerant exotic species (Lunt *et al.*, 2007; Archibald, Hempson and Lehmann, 2019). As well as affecting ground layer diversity generally, grazing has also been found to negatively impact key SBTF foraging species (Section 6.2). In addition, higher grazing pressures reduce the seed production of grasses and forbs, particularly in high palatable perennial grasses (O'Connor and Pickett, 1992; Ash *et al.*, 1997).The spread of pastoralism and associated land management changes has been implicated in the decline of SBTF among a range of other native fauna species (J. Woinarski *et al.*, 2011; Mula Laguna *et al.*, 2019; D. Franklin, 1999; D. Franklin *et al.*, 2005).

Evidence of high grazing pressure and impacts to the ground layer was recorded in all subregions within the study area but was particularly severe in the Jericho and Cape-Campaspe Plains. Several properties in these areas contained potentially suitable SBTF habitat, however the ground cover had been heavily grazed in areas so that only a stubble of grass tussocks remained.



Figure 7.2-4: Ground layers impacted by overgrazing on Wambiana (left) and Jerico subregion (right)

Furthermore, grazing impacts are likely to be greatest near water sources (James, Landsberg and Morton, 1999; Fensham and Fairfax, 2008), which many granivores are inherently associated with due to their frequent drinking requirements (Czenze *et al.*, 2020). The installation of artificial water sources within the study area has resulted in the formation of noticeable grazing piospheres (Lange, 1969), where the area surrounding the water point is severely degraded due to overgrazing from stock that congregate around water points (James, Landsberg and Morton, 1999; Fensham and Fairfax, 2008).



Figure 7.2-5: Grazing piospheres surrounding a trough on Longton (left) and a dam on Maynard (right)

In addition to the direct impacts of grazing, fire is often excluded in pastoral areas, either actively or as a result of livestock grazing (Russell-Smith, Start and Woinarski, 2001). Prior to European colonization, fire regimes primarily consisted of regular, low-intensity burns that disturb ground layers dominated by high-biomass native perennial grasses (Russell-Smith, Start and Woinarski, 2001; Yibarbuk *et al.*, 2001). This regime promotes native species diversity by providing a disturbance that creates opportunities for short-lived perennial and annual grass species to establish (Russell-Smith, Start and Woinarski, 2001; Lebbink, Fensham and Cowley, 2018). It also creates a patchy mosaic of bare ground and grass cover that allows granivores access to fallen grass seed on the ground during the dry season. The ground cover structure has been identified as an important factor for habitat selection and foraging sites for SBTF (Section 4.2).

Fire exclusion without managing biomass through grazing can also lead to the build-up of fuel sources and increases the risk of high-intensity fires occurring during dry periods (August to November) which can pass through large swathes of the landscape (Russell-Smith and Edwards, 2006; Murphy and Russell-Smith, 2010). These intense fires negatively impact the structure and diversity of SBTF habitat through the destruction of seed banks, increase in non-native cover and thickening of the woody shrubs (Williams *et al.*, 1999; Liedloff *et al.*, 2001; Yates, Edwards and Russell-Smith, 2008; Miller *et al.*, 2010; Wright and Fensham, 2016). Many of the properties surveyed, had long unburnt areas with low ground layer species diversity and high biomass that are at high risk of a high intensity wildfire such as seen in the Jericho subregion in Figure 7.2-6.

Overgrazing can also lead to fire exclusion through the removal of biomass, which often leads to the thickening of woody shrubs which, in turn, can suppress the grassy understorey (Liedloff *et al.*, 2001; Williams, Woinarski and Andersen, 2003). This was observed on numerous properties that was surveyed, especially in the Cape-Campaspe Plains, Prairie-Torrens Creek Alluvials and Alice Tablelands subregions.- This can also reduce food accessibility as SBTF require bare ground patches between grass tussocks in order to forage seeds from the ground and access seed heads (Buosi, 2011b). Further details on this subject are detailed in Section 4.2.




Figure 7.2-6: Results of fire exclusion recorded during the survey, including build-up of biomass in Jerico subregion (left) and woody shrub encroachment due to overgrazing on Longton (right)

7.2.3.3 Exotic species

Exotic pasture grasses have been introduced to Queensland to increase livestock production (Marshall et al., 2011). The dominant exotic species recorded in the study area were buffel grass (*Cenchrus ciliaris*) and stylo (*Stylosanthes* spp.) which were recorded within most of the surveyed properties.

Buffel grass, a perennial native of south and east Africa and southern Asia, is an invasive drought tolerant grass that poses a serious threat to biodiversity in Australia (Wright *et al.*, 2020). Buffel grass invasions result in competitive displacement of native grasses and reduction in overall plant diversity while providing little foraging value for SBTF (Wright *et al.*, 2020). In addition to competitive effects, invasion by vigorous naturalised grasses can significantly increase fuel loads and therefore increase potential fire intensity and frequency, which in turn may impact negatively on many native species (Butler and Fairfax, 2008).

The second most common exotic species recorded during the surveys is stylo, which has been widely introduced across agricultural areas of northern Australia (Gardener, 1993). Stylo species are resilient to heavy grazing due and can easily spread through seeds dispersed in cattle faeces (Gardener, 1993; Orr, 2010).

Studies within Moray Downs have found that higher cover of buffel grass and stylo negatively influenced the SBTF's selection of foraging habitat (Section 4.2). Buffel grass is widespread across all areas that were surveyed in the Desert Upland region. Stylo was also commonly recorded, but higher covers were observed on properties with evidence of particularly high grazing and clearing including Longton and Widgeman.

7.2.4 Conclusion

Although no SBTF were recorded during the ten-day survey, large areas of superficially suitable habitat, namely *Eucalyptus melanophloia* and *Eucalyptus brownii* woodlands, were observed. However, the condition of the majority of these habitats was highly degraded with only relatively small patches of habitat containing a suitably open and grassy understorey with a high diversity of native grasses. The decline in habitat condition was largely attributed to grazing pressures, which had altered the composition and structure of the ground layer and consequently reduced the availability of important food sources for SBTF. The exclusion of fire has also resulted in a thickening of woody shrubs that also impact on the

condition of the groundcover, rendering these areas unsuitable for SBTF. Combined with the clearing of large areas of habitat, especially in the south of the Desert Uplands, these factors have demonstrably resulted in the decline of SBTF across the region.

7.3 Environmental DNA detection trial

7.3.1 Abstract

Environmental DNA (eDNA) is a promising technology offering potential to detect trace amounts of species' DNA that are deposited in the environment. This study evaluated the suitability of environmental DNA (eDNA) sampling to detect the presence of the Endangered southern black-throated finch (SBTF) at water sources in central Queensland, Australia. Water samples were collected from 17 different water sources, that had varying levels of use by SBTF. Results showed SBTF DNA was detected at two water sources: a small trough and a cattle trough. However, detection was inconsistent and failed to detect SBTF at water sources that were likely to be in use by the species at the time of eDNA sampling. Challenges such as high water turbidity, DNA degradation factors (e.g., UV radiation and temperature), and dilution in larger water sources affected the reliability of eDNA detection. We conclude that while eDNA sampling is feasible for detecting SBTF presence, its current limitations make it unsuitable as a standalone monitoring technique. Alternative methods, including water source watches, remote cameras, and bioacoustic recorders, may be more effective for monitoring SBTF populations. Future development of eDNA methods will need to address environmental influences on DNA stability and filtering to improve reliability.

7.3.2 Introduction

Environmental DNA (eDNA) is a rapidly developing and powerful tool to detect and monitor species presence and abundance (Voros *et al.*, 2017). The development of eDNA methods has focussed on aquatic environments, targeting the DNA of species such as amphibians, reptiles, fish and invertebrates (Beng and Corlett, 2020). However, applications of eDNA to terrestrial environs are being investigated, such as detecting koalas using airborne eDNA (Frere *et al.*, 2023), creating species inventories from genetic material collected from spider webs (Gregorič *et al.*, 2022) and detecting Gouldian finch drinking from water sources using water samples (Day *et al.* 2019). Such development of eDNA methods offers potential to detect the use of water sources by the Endangered southern black-throated finch, potentially providing a tool to monitor the species' habitat use.

While interest in eDNA for species detection is on the rise, several factors must be considered when interpreting eDNA data. The development of eDNA methods is highly species and study area specific due to the need to develop targeted species-specific assays and variability of factors that affect eDNA sampling (Beng and Corlett, 2020). The rates at which eDNA is released and decays are influenced by a variety of biotic and abiotic factors. Biotic factors include life-history traits of the organisms, interactions between species, and the presence of microbial communities, all of which can affect the persistence and breakdown of eDNA in the environment (Stewart, 2019). Abiotic factors encompass environmental conditions such as ultra violet radiation, temperature, and salinity, which can substantially affect the stability and degradation of eDNA. (Beng and Corlett, 2020). For example, higher temperatures directly accelerate DNA degradation by denaturing DNA molecules and also indirectly degrade eDNA by enhancing enzyme kinetics and microbial metabolism (Barnes *et al.*, 2014). Another major challenge to the application of eDNA is the difficulty of quantifying species abundance and biomass, as this method has most often been used for presence/absence studies. False negatives can also occur during eDNA sampling as PCR inhibitors can directly interact and bind with DNA, which can prevent amplification or block enzyme activity (Beng and Corlett, 2020). Day *et al.* (2019) also found that highly turbid waters can negatively affect eDNA detection and the volume of water in the water source can influence the probability of species detection.

In this study, we assessed the suitability of eDNA sampling to detect the presence of the Endangered southern black-throated finch within central Queensland, Australia. This was an exploratory study to test whether this method is a viable survey technique for ongoing monitoring of the species and if it can be

used as a standalone survey technique (Beng and Corlett, 2020). We compare the results of eDNA sampling to the utilisation of water sources by SBTF that was recorded from previous monitoring surveys, such as water source watches and remote camera deployment.

7.3.3 Methods

Water samples were taken at 17 water sources within the study area on the 13 and 14 February 2024. The sampling design was chosen to include a diversity of water source types (cattle troughs and ephemeral surface water) and rates of use by SBTF (Table 7.3-1). The rate of SBTF use of water sources was categorised using data from multiple sources, which include water source watches, remote camera deployments, radio tracking, incidental records and mapping of SBTF habitat within the study area. Water sources with a 'consistent use' include those with regular SBTF records from water source watches, active searches and 1 km transects that were recorded within a year prior to the eDNA sampling date. These included four cattle troughs and one ephemeral surface water source that holds water for approximately 1–2 months following rain. 'Inconsistent' use water sources have had SBTF recorded using, albeit irregularly and infrequently. Inconsistently used water included one cattle trough and two ephemeral scrapes at which SBTF have been observed drinking on three or fewer occasions per water source within a year prior to the eDNA sampling date. Three 'known negative' water sources were selected, which were located at least 4 km from the nearest SBTF record and area of known SBTF habitat. All known negative water sources were in non-remnant vegetation, which is avoided by SBTF (van Osta *et al.*, 2024b). An additional six cattle troughs were sampled at which the presence of SBTF was uncertain. These troughs were included within the study to provide additional water source use information on SBTF if the eDNA analyses were successful.

Duplicate samples were collected at eight of these water sources to investigate sampling variation and repeatability of and detections. Each water sample was collected by passing water through a 1.2 µm PES syringe filter (Figure 7.3-1). An average of 208 mL (range = 10–450 mL) of water was passed through the filter for each sample. Preservative was then added to the filter to stabilise the DNA for transport. All samples were transferred to a cool-box upon their collection and stored in a refrigerator for 1–2 days prior to being posted to the laboratory. All sampling kits were supplied by EnviroDNA.

A qPCR assay was developed by EnviroDNA to target the control region for SBTF using sequences from GenBank and SBTF blood samples collected from our study area. Initially, two probes were created that performed similarly for the target species. However, Probe 1 amplified all local off-target species at PCR cycles similar to those of the target species, leading to its elimination. In contrast, Probe 2 did not amplify the plum-headed finch but did amplify the zebra finch and double-barred finch at approximately 10 cycles higher than the target species, meaning that it would require approximately 1024 times higher DNA concentrations of zebra finch and double-barred finch DNA than SBTF DNA to cause off-target amplification issues. Given the low concentration that eDNA samples typically yield, there was therefore minimal risk of off-target amplification. The resultant limit of detectability for this probe was determined to be 2 copies per microlitre, limit of reliable quantification of at 10 copies per microlitre (Thalinger *et al.*, 2021).

Table 7.3-1: Number of tater sources sampled for environmental DNA across two water source types (surface water and troughs) and differing frequency of use by southern black-throated finch.

Estimated SBTF use	Surface water	Trough
Consistent use	1	4
Inconsistent use	2	1
Known-negative	0	3
Uncertain	0	6



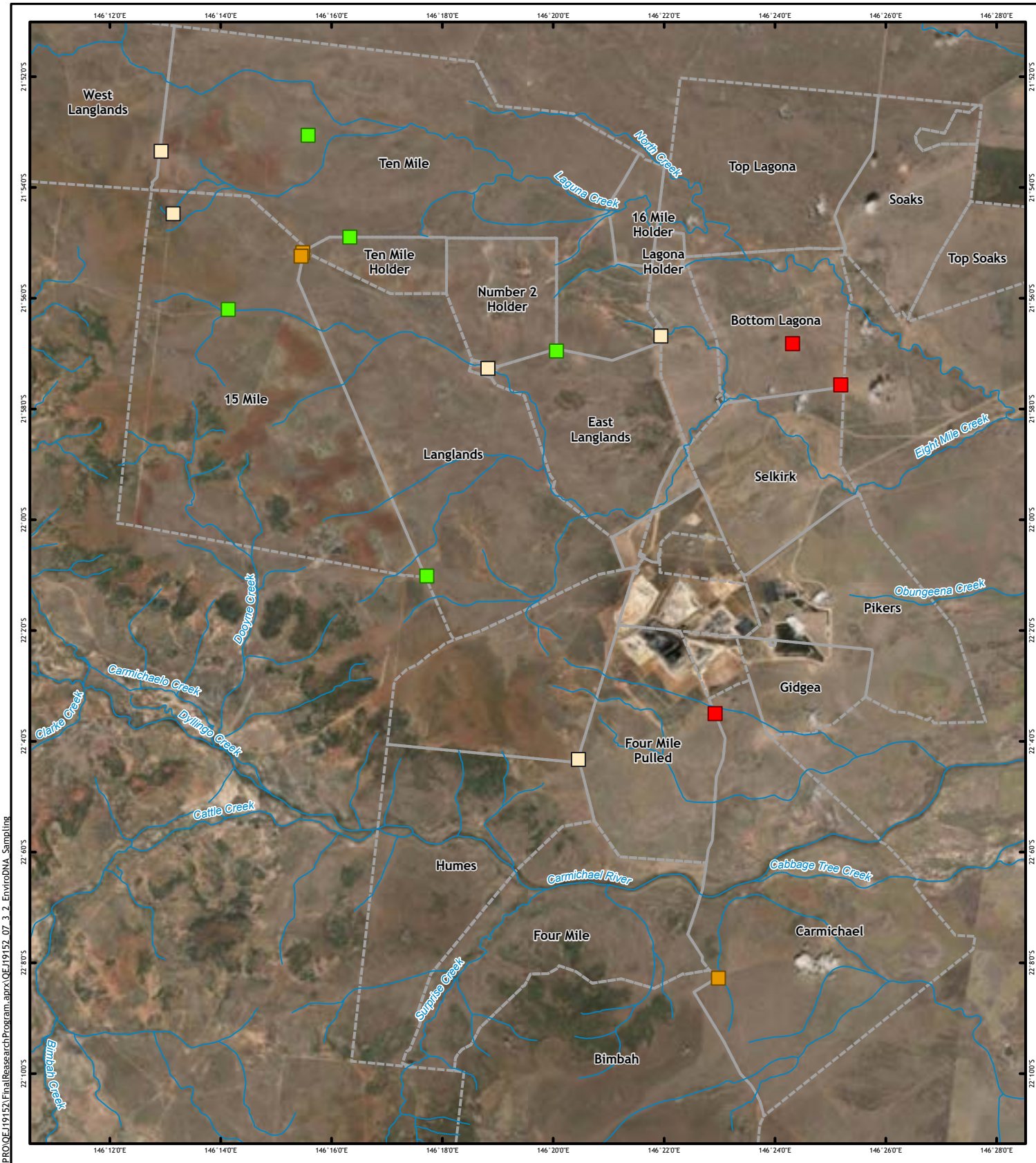
Figure 7.3-1: Water sample collection method at CWAT99

7.3.4 Results

Southern black-throated finch DNA was detected at two water sources (TT01 and CWAT99; Table 7.3-2, Figure 7.3-2). The detection at TT01 comprised a strong detection with all assays (n = 3) returning a positive result. In contrast the detection at CWAT99 was equivocal, with only 1 of the three assays returning a positive result. A duplicate sample taken at CWAT99 returned a negative result. All other samples from the remaining 15 water sources returned a negative result, including three water sources in consistent use by SBTF at the time of survey (Table 7.3-2).

Table 7.3-2: Number of tater sources sampled for environmental DNA across two water source types (surface water and troughs) and differing frequency of use by southern black-throated finch.

Water source	Water source characteristics	Estimated SBTF use	eDNA detection
CWAT01A	Dam	Inconsistent use	Negative
CWAT01B	Ephemeral	Inconsistent use	Negative
CWAT102	Trough	Uncertain	Negative
CWAT105	Trough	Uncertain	Negative
CWAT109	Trough	Uncertain	Negative
CWAT117	Trough	Uncertain	Negative
CWAT118	Trough	Inconsistent use	Negative
CWAT301	Trough	Consistent use	Negative
CWAT302	Trough	Consistent use	Negative
CWAT400	Trough	Known-negative	Negative
CWAT401	Ephemeral	Consistent use	Negative
CWAT402	Trough	Uncertain	Negative
CWAT404	Trough	Known-negative	Negative
CWAT405	Trough	Known-negative	Negative
CWAT99	Trough	Consistent use	Equivocal
TT01	Trough	Consistent use	Positive



Legend

Estimated BTF use

- Consistent use
- Inconsistent use
- Known-negative
- Uncertain

- Watercourse
- Moray Downs Paddock

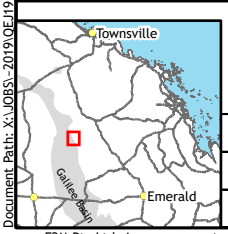
N

Scale 1:175,000 (A4)

0 5

Kilometres

Coordinate System: GCS GDA 1994



Notes:
Aerial Imagery: © ESRI 2024
Watercourse: © Geoscience Australia 2012

Rev	Description	Drawn	Approved	Date
2	Issued for Use	JY	JMC	30/10/2025
0	Issued for Use	GO	JMC	11/08/2025

FIGURE 7.3-2: ENVIRONMENTAL DNA SAMPLING LOCATIONS

BTFRP Final Report
Bravus

Map Number	Job Number	Rev
1 of 1	QEJ19152	2

Document Path: X:\JOBS\-2019\QEJ19152\GIS\Final Research Program Report\PRO\QEJ19152\FinalReasearchProgram.aprx\QEJ19152_07_3_2_EnviroDNA_Sampling

E2M Pty Ltd gives no warranty in relation to the data (including accuracy, reliability, completeness or suitability) and accepts no liability for any loss, damage or costs (including consequential damage) relating to any use of the data in this map.



Figure 7.3-3:Water sources where eDNA sampling detected use by southern black-throated finch, including TT01 (left) and CWAT99 (right)



Figure 7.3-4:Water sources that were in consistent use by southern black-throated finch but had a negative detection result using eDNA, including CWAT301 (left) and CWAT302 (right)

7.3.5 Discussion

In this study, we assessed the suitability of eDNA for detecting SBTF use of water sources. While eDNA successfully detected SBTF presence at two water sources, our results indicate significant limitations in applying this method for SBTF monitoring. The only water source with unequivocal evidence of SBTF use was a small (approx. 3-litre) 'tree trough' (TT01, fabricated by Wildsip Pty Ltd, Figure 7.3-3). This type of water source is highly atypical across the SBTF range and was installed to trial alternative locations in the study area. The other positive detection occurred at a steel cattle trough (CWAT99, Figure 7.3-3), where SBTF DNA was identified in one out of three assays from a single sample but was not found in a duplicate sample. This water source has been used consistently by SBTF since monitoring began in 2014, though usage can lapse when alternative surface waters are available during the wet season. During the survey, ephemeral water was largely unavailable, making use of CWAT99 at the time of sampling likely.

Several factors may have limited the detection of SBTF DNA at water sources. High turbidity reduces the volume of water that can be passed through the filter plate, thus limiting the amount of DNA collected (Beng and Corlett, 2020). Turbidity was particularly noted at surface water sources like dams and ephemeral pools. Additionally, factors such as ultraviolet radiation, bacterial activity, and heat can accelerate DNA degradation, reducing the amount recoverable from samples (Barnes *et al.*, 2014). The quantity of DNA in the water also depends on the size of the water source, with the concentration of SBTF DNA within large water sources such as earthen dams likely to be lower than at smaller stock watering troughs (Beng and Corlett, 2020). Furthermore, water overflow from the CWAT302 trough was noted during eDNA sampling, which may have reduced SBTF DNA concentrations within the water and hampered eDNA detection at this site that is consistently used by SBTF. In contrast, the ‘tree trough’ (TT01) at which SBTF DNA was detected, was filled with clean drinking quality water, located under tree shade and was not accessible to cattle. These conditions likely resulted in a concentration of SBTF DNA within the small volume of water in the ‘tree trough’, affording more reliable eDNA detection.

The application of eDNA to detect water source use by SBTF must be assessed against readily available alternative methods. Such alternative methods include water source watches by a human observer (DEWHA, 2009b), remote cameras for smaller water sources such as cattle troughs, or bioacoustic recorders can be placed within treelines where SBTFs gather before drinking (Section 7.4; van Osta *et al.*, 2023). Considering SBTF are likely to use water sources on a near daily basis, particularly in high ambient temperatures (Mula Laguna *et al.*, 2019; Withers, Cooper and Larcombe, 2022), we consider that such alternative methods are more likely to detect the presence of SBTF than compared to eDNA sampling. These alternative methods are also benefitied by their ability to estimate the relative abundance of SBTF, which while not impossible is technically constrained for eDNA sampling (Beng and Corlett, 2020).

While our results indicate that detecting SBTF presence using eDNA is feasible, the practical application remains limited with the methods currently available. We therefore conclude that eDNA surveys are currently not a suitable standalone method to effectively monitor SBTF populations. Future development of the eDNA techniques will require mitigation of environmental factors that affect eDNA persistence and detection, such as turbidity, temperature, and DNA dilution in large and overflowing water sources.

7.4 Development of an automated SBTF call detection model

This section has been published as a scientific paper in the journal *Ecological Informatics*. Bibliographic details of this scientific paper are:

> van Osta, J. M., Dreis, B., Meyer, E., Grogan, L. F., & Castley, J. G. (2023). An active learning framework and assessment of inter-annotator agreement facilitate automated recogniser development for vocalisations of a rare species, the southern black-throated finch (*Poephila cincta cincta*). *Ecological Informatics*, 77, 102233. https://doi.org/10.1016/j.ecoinf.2023.102233

7.4.1 Abstract

The application of machine learning methods has led to major advances in the development of automated recognisers used to analyse bioacoustics data. To further improve the performance of automated call recognisers, we investigated the development of efficient data annotation strategies and how best to address uncertainty around ambiguous vocalisations. These challenges present a particular problem for species whose vocalisations are rare in field recordings, where collecting enough training data can be problematic and a species' vocalisations may be poorly documented.

We provide an open access solution to address these challenges using two strategies. First, we applied an active learning framework to iteratively improve a convolutional neural network (CNN) model able to automate call identification for a target rare bird species, the southern black-throated finch (*Poephila cincta cincta*). We collected 9,098 hours of unlabelled audio recordings from a field study in the Desert Uplands Bioregion of Queensland, Australia, and used active learning to prioritise human annotation effort towards data that would best improve model fit. Second, we progressed methods for managing ambiguous vocalisations by applying machine learning methods more commonly used in medical image analysis and natural language processing. Specifically, we assessed agreement among human annotators and the CNN model (i.e. inter-annotator agreement) and used this to determine realistic performance outcomes for the CNN model and to identify areas where inter-annotator agreement may be improved. We also applied a classification approach that allowed the CNN model to classify sounds into an 'uncertain' category, which replicated a requirement of human-annotation and facilitated the comparison of human-model annotation performance.

We found that active learning was an efficient strategy to build a CNN model where there was limited labelled training data available, and target calls were extremely rare in the unlabelled data. As few as five active learning iterations, generating a final labelled dataset of 1,073 target calls and 5,786 non-target sounds, were required to train a model to identify the target species with comparable performance to experts in the field.

Assessment of inter-annotator agreement identified a bias in our model to align predictions most closely with those of the primary annotator and identified significant differences in inter-annotator agreement among subsets of our acoustic data. Our results highlight the use of inter-annotator agreement to understand model performance and identify areas for improvement in data annotation. We also show that excluding ambiguous vocalisations during data annotation results in an overestimation of model performance, an important consideration for datasets with inter-annotator disagreement.

7.4.2 Introduction

The use of low-cost acoustic sensors in ecological research has rapidly expanded, enabling new approaches to the detection and monitoring of species (Hervás, Rosa Ma Alsina-Pagès, *et al.*, 2017; Sugai, Silva, Ribeiro Jr, *et al.*, 2019; S. R. Ross *et al.*, 2023). The increased adoption of acoustic sensors has led to an exponential increase in the volume of data produced, thereby positioning bioacoustics within the domain of 'big data' (Sugai, Silva, Ribeiro Jr, *et al.*, 2019).

Accurate automated recognition of bird calls in real-world acoustic data remains a key challenge limiting practical applications of passive acoustic monitoring (Priyadarshani, Marsland and Castro, 2018; Stowell, 2022). Acoustic analyses of bird species and communities typically rely on the development of individual species call recognisers, particularly for threatened species (LeBien *et al.*, 2020; Ruff *et al.*, 2021; Teixeira *et al.*, 2022), or use soundscape indices to draw comparisons among sites (Colonna, Carvalho and Rosso, 2020; Dema *et al.*, 2020). Classification of bird species from real-world environmental data has proven challenging due to unavoidable environmental noise overlap with target bird calls, attenuation of bird calls at distance, overlapping calls, and intra-specific call variation (Priyadarshani, Marsland and Castro, 2018), but also the lack of sufficient training data to build species-specific recognisers (Gibb *et al.*, 2019).

Deep learning methods, such as convolutional neural networks (CNNs) and recurrent neural networks, have proven highly effective in the classification of individual bird species in complex acoustic data (Stowell, 2022) and have been used in a variety of automated call recognition studies (Allen et al., 2021; Fairbrass et al., 2019; Kahl et al., 2021; Ruff et al., 2021; Stowell et al., 2019). Deep learning methods have outperformed alternative detection and classification methods for the identification of specific bird species provided sufficient training data are available to avoid model overfitting, which is a common constraint of deep learning (Gibb *et al.*, 2019).

While the application of machine learning (ML) methods to detect wildlife calls is a well-established and active field of research (Stowell *et al.*, 2019), the majority of existing studies have used large, labelled datasets to train classifiers using supervised learning approaches. Examples include the use of publicly available wildlife call datasets (Sankupellay and Konovalov, 2018; Stowell *et al.*, 2019), annotation by citizen scientists (Mac Aodha *et al.*, 2018) or existing labelled datasets (Towsey *et al.*, 2012; Bergler *et al.*, 2019). However, these approaches may not be suitable for the detection of a species where labelled training data are not available and are expensive to obtain due to the cost of human-generated labels. The expense of creating labelled datasets limits the ability to train state-of-the-art deep learning models (Van Engelen and Hoos, 2020), and provides a barrier to developing recognisers for rare or cryptic species, as well as more common species that vocalise infrequently.

Rare species are more sensitive to human-induced impacts such as habitat loss and climate change than common species and have a higher risk of extinction (Davies, Margules and Lawrence, 2004; Sekercioglu *et al.*, 2008; Loiseau *et al.*, 2020). Environmental legislation and policies have broadly struggled to reverse the decline of rare species, in part due to the difficulty gaining knowledge on these species' distributions, habitat use and ecological interactions (Bland *et al.*, 2015). At the problem's most severe, data deficient species may not be recognised in threatened species lists that drive conservation actions, such as International Union for Conservation of Nature's red list (Howard and Bickford, 2014) and the Australian Government's *Environment and Biodiversity Protection Act 1999* threat categories (Braby, 2018). Furthermore, when these species are recognised, the data to guide conservation actions may not be available (Howard and Bickford, 2014; Braby, 2018). Extending bioacoustic methods to improve the application for rare and difficult to survey species may be a powerful tool to limit data deficiencies and thus improve species' conservation outcomes (Wood *et al.*, 2023).

One such method that has the potential to improve the applicability of bioacoustics to rare and difficult to survey species is active learning. Active learning is a ML paradigm that incorporates a human-machine interface to optimise the annotation of training data (Van Engelen and Hoos, 2020; Monarch, 2021). In a recent review and roadmap of computational bioacoustics, Stowell (2022) identified the development of active learning techniques as highly important to the future of bioacoustics. Active learning is particularly suitable for applications where limited labelled data exist and the cost of obtaining a large labelled dataset is high (Van Engelen and Hoos, 2020). For example, labelled data may be unavailable due to the high cost of human-annotation (Zhu and Goldberg, 2009), or due to the rarity of an event/record making the target signal difficult to find in unlabelled data, thus reducing sample size or increasing the cost of human-annotation (Shi *et al.*, 2019).

The collection of labels for bioacoustics research is often conducted through manual annotation by experts and trained annotators with expertise in the species of interest (Stowell, 2022). The goal is to achieve near 100% accuracy in the annotation of target calls. However, in practice, even highly-trained and experienced experts can have difficulty achieving high annotation reliability from real-world recordings. In two studies of auditory detection at avian point counts, both Simons et al. (2007) and Alldredge, Simons and Pollock (2007) found annotators' abilities to detect species' calls varied by 28% to 42% among varying treatments of ambient noise and distance. Furthermore, Mortimer and Greene (2017) found that agreement among annotators processing audio recordings of New Zealand avifauna ranged between 23.5% and 85.1%, depending on the species being annotated. All three studies identified that the distinctiveness of the bird species call compared to other calls in the environment was a key factor influencing the reliability of call detection and annotation.

Assessment of inter-annotator agreement, also known as inter-rater reliability, is widely recommended to determine the reliability of manual annotation (Reidsma and Carletta, 2008; Gwet, 2014; Duc *et al.*, 2021). Inter-annotator agreement is a measure of the consistency and agreement among multiple annotators in their annotation of a dataset (Gwet, 2014). While inter-annotator disagreement is a well-documented issue for bioacoustics data (Duc *et al.*, 2021), relatively few studies incorporate methods to account for imperfect inter-annotator agreement into the design and evaluation of resulting ML models. Instead, all annotations regardless of annotator or species are typically treated as certain, which Cabitza et al. (2020) and Campagner et al. (2021) describe as the 'elephant in the machine' in the context of medical ML applications. Cabitza et al. (2020) also argue that considering uncertainty in the annotation process is important to develop models that generalise to real-world scenarios.

We present here an open access, novel method to develop a call detection model for a threatened bird species, the southern black-throated finch (SBTF), *Poephila cincta cincta*, that lacks an existing labelled call dataset. Our method applies an active learning framework to prioritise scarce human annotation resources towards data that are the most likely to improve the model. Our work builds on the 'standard recipe' for bioacoustic classification described by Stowell (2022) and applies an active learning framework to iteratively improve the model. We also apply a novel method to include inter-annotator agreement and label uncertainty, which are rarely considered in bioacoustics, into the development and evaluation of call detection models. This work has broad application for many species that have limited existing training data available and/or for species whose calls may not be annotated with certainty.

7.4.3 Methods

7.4.3.1 Target species call

The southern subspecies of the black-throated finch (SBTF), *Poephila cincta cincta*, is a grassfinch (family Estrildidae) endemic to north-eastern Australia typically inhabiting open grassy woodlands (Higgins, Peter and Cowling, 2006; Shephard, Pridham and Forshaw, 2012). The SBTF is listed as 'Endangered' under the

Australian *Environment Protection and Biodiversity Conservation Act 1999* since its extent of occurrence declined by up to 59% in the ten years prior to 2005 (Commonwealth of Australia, 2023).

Black-throated finch have a complex vocal repertoire, with 12 calls described (Shephard, Pridham and Forshaw, 2012). Of these, the 'long'/'pew' call is the farthest carrying and most frequently heard call (Higgins, Peter and Cowling, 2006; Shephard, Pridham and Forshaw, 2012) and as such is the priority for automated detection. Other calls described for the species include a complex of short-range communication and alarm calls, which are heard less frequently and are not as far carrying as the 'long'/'pew' call (Shephard, Pridham and Forshaw, 2012).

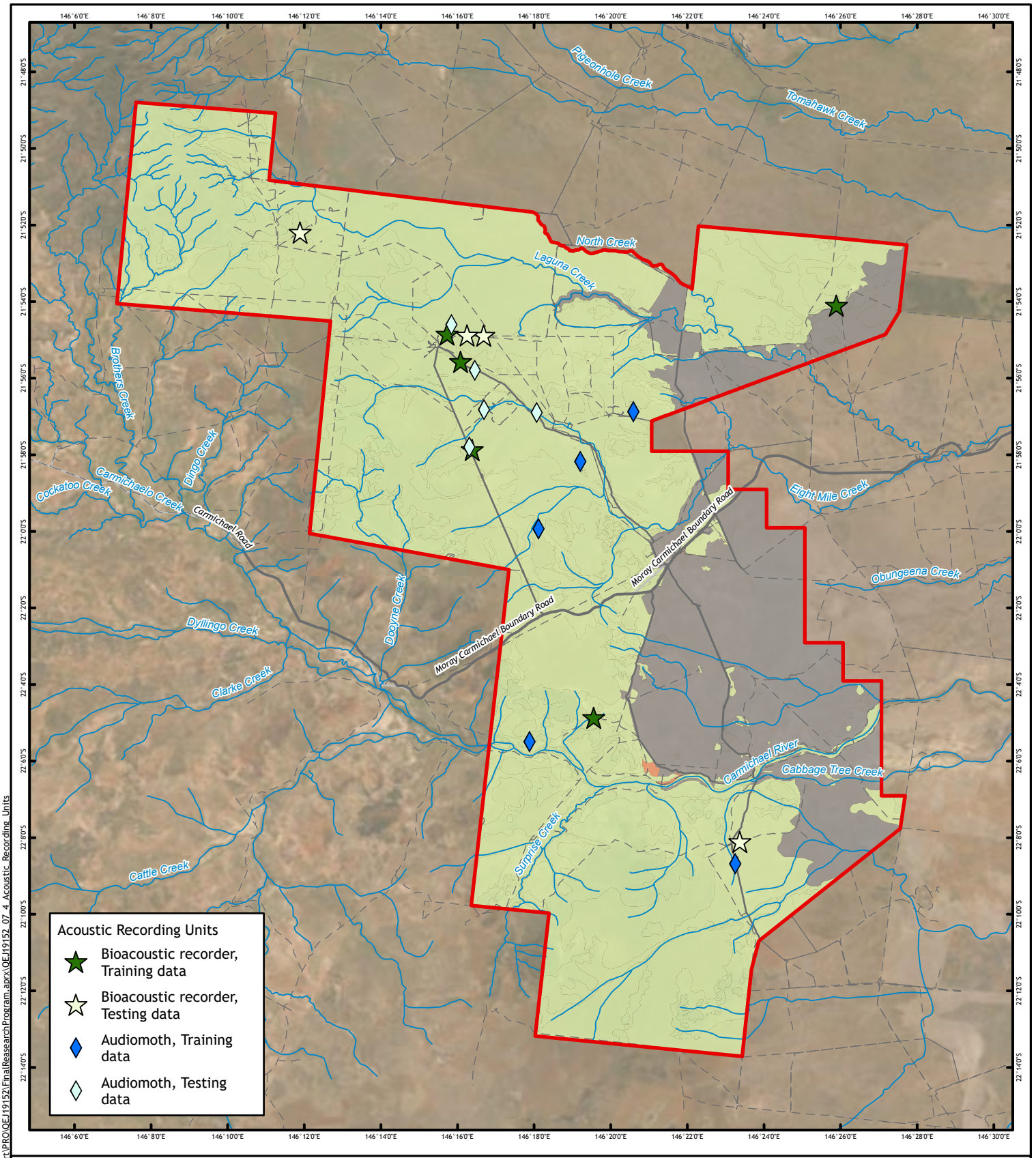
7.4.3.2 Audio data and study area

Recordings of the SBTF were made in in the Desert Uplands Bioregion of Queensland, Australia, near the southern limit of the species current distribution, from 25 August 2020 to 24 April 2021. Recordings were collected as part of an ongoing SBTF research program for Bravus Mining and Resources (Bravus) on a 75,000 ha section of the Moray Downs property, which includes areas managed as environmental offset by Bravus. The study area is largely vegetated, with remnant vegetation, comprising 24 discrete vegetation communities, occurring over 79% of the study area (DES, 2022) (Figure 7.4-1). The study area contains a diverse assemblage of woodland birds, with a total of 119 bird species recorded in the SBTF habitat within the study area (Author, unpublished data, 2023). Among the local avifauna are other Estrildid finches, such as the double-barred finch (*Taeniopygia bichenovi*i), plum-headed finch (*Neochmia modesta*), and zebra finch (*Taeniopygia castanotis*). Of these, the double-barred finch has a call that is most similar to the target SBTF call.

Two types of acoustic recording unit were used within this study. Solar-powered bioacoustic recorders (BARs; Frontier Labs; Roe et al., 2021) were installed within known and suspected SBTF habitat for periods of 6 to 12 months per unit. Solar-powered BARs are suited to long-term deployments, with a weight of 4 kg, no battery limitations, 4 SD card slots and a high-fidelity microphone (signal-to-noise ratio [SNR] = 80 dB; Open Acoustic Devices, n.d.). We also installed Audiomoth recorders (Hill *et al.*, 2019) near known and suspected SBTF nests for a duration of 2 to 4 weeks per unit (Figure 7.4-1). Audiomoths are suited to short-term and opportunistic deployments due to their light-weight (80 g, including 3 AAA batteries) and low-cost (Hill *et al.*, 2019). Recording quality is comparatively lower on Audiomoths compared to the solar-powered BARs, with a SNR of 44.2 dB (Hill *et al.*, 2019).

A total of 9,098 hours of audio data, comprising 8,266 hours from solar-powered bioacoustic recorders from nine independent sites, and 832 hours from Audiomoth recorders from ten known and suspected nest sites, were used for this study. Audio data were separated into training/validation (Section 7.4.3.3) and test datasets (Section 7.4.3.4) prior to the commencement of CNN development. The training/validation dataset comprised 6,363 hours of audio from ten independent sites (five solar-powered bioacoustic recorders and five Audiomoth recorders) and the test dataset comprised 2,735 hours of audio from nine independent sites (four solar-powered bioacoustic recorders and five Audiomoth recorders) that were different from the training/validation dataset. All audio data were unlabelled at the commencement of the study.

All recorders were set to record 10-minute files continuously between 6 am and 6 pm daily, which reflected the diurnal activity patterns of SBTF. We stored audio data as FLAC files at a sampling rate of 44.1 kHz for the solar-powered bioacoustic recorders and WAV files at a 32 kHz sampling rate for the Audiomoth recorders.



Legend

- Watercourse
- Major Road
- Minor Road
- Track
- Street/Local - only provides property access
- Project Area

Vegetation Status

- Remnant
- HVR
- Non-remnant

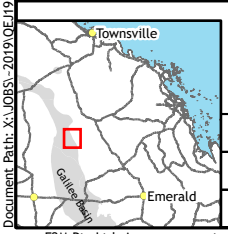
N

Scale 1:250,000 (A4)

0 5 10

Kilometres

Coordinate System: GDA2020



Notes:
Aerial Imagery: © ESRI 2024
Watercourse: © Geoscience Australia 2012

Rev	Description	Drawn	Approved	Date
2	Issued for Use	JY	JMC	30/10/2025
0	Issued for Use	GO	JMC	08/08/2025

FIGURE 7.4-1: SPATIAL DISTRIBUTION OF ACOUSTIC RECORDING UNITS WITHIN THE STUDY AREA

BTFRP Final Report
Bravus

Map Number	Job Number	Rev
1 of 1	QEJ19152	2

Document Path: X:\JOBS\-2019\QEJ19152\GIS\Final Research Program Report\PRO\QEJ19152\FinalReaseachProgram.aprx\QEJ19152_07_4 Acoustic_Recording_Units

E2M Pty Ltd gives no warranty in relation to the data (including accuracy, reliability, completeness or suitability) and accepts no liability for any loss, damage or costs (including consequential damage) relating to any use of the data in this map.

7.4.3.3 CNN development

7.4.3.3.1 Data pre-processing

We split audio data into 1.8 second audio frames, which is approximately double the maximum length of the target call (Higgins, Peter and Cowling, 2006). We used a 50% overlap between frames, i.e. a sliding window approach, which ensured target calls would be entirely included within at least one audio frame (Kahl *et al.*, 2021). We then transformed audio frames into mel-scaled spectrogram images using a short-time Fourier transform, with a Hann window length of 2048, a 50% overlap between segments and 128 mel filter banks, following standard methods such as those applied by LeBien et al. (2020).

We chose a frequency bandwidth of 1.5 MHz to 5 MHz, which ensured the dominant frequency of the target call was included within the frame, while avoiding excess noise at lower frequencies and higher frequencies (e.g. cicadas). We found that this frequency bandwidth typically included a fundamental harmonic and/or a higher harmonic of the dominant frequency; however, these harmonics were not the focus of the bandwidth selection due to their rapid attenuation at lower sound pressure levels (Koehler *et al.*, 2017). An example frame used as input into the CNN, following data pre-processing steps, is depicted within Figure 7.4-2.

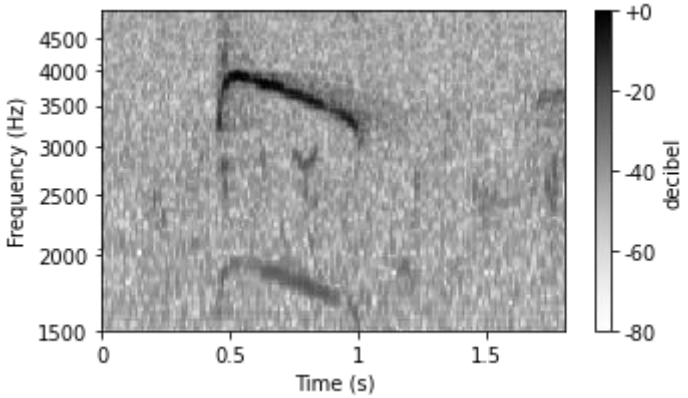


Figure 7.4-2: Example audio frame following data pre-processing steps that was used as input into the convolutional neural network. The audio frame shows a SBTF call with a dominant frequency (stronger signal) and a fundamental harmonic (weaker signal) included in the frame.

7.4.3.3.2 CNN architecture

Application of CNNs to audio recognition tasks is well established, with CNNs forming part of the 'standard recipe' for bioacoustic classification (Allen et al., 2021; LeBien et al., 2020; Stowell, 2022). We used a data pre-processing pipeline and CNN architecture that have been widely applied for bioacoustic classification tasks (Christin, Hervet and Lecomte, 2019; Stowell *et al.*, 2019; Stowell, 2022).

We compiled a CNN using Python (version 3.6.9, Python Foundation), within Google Colaboratory, to interface with Pytorch (version 1.10), an open source ML library (Paszke *et al.*, 2019). We used a ResNet-34 model with pre-trained weights for the CNN architecture. ResNet models typically achieve high-performance in image and audio recognition tasks (He *et al.*, 2016; Stowell *et al.*, 2019; Bergler *et al.*, 2022) and have been widely applied for automated wildlife image and call recognition (Sankupellay and Konovalov, 2018; Kahl *et al.*, 2021; Stowell, 2022). Our training dataset was imbalanced, with a lower number of audio frames containing a SBTF call than no-SBTF call. We followed the recommendations of Buda, Maki and Mazurowski (2018) and oversampled the SBTF audio frames with the

WeightedRandomSampler function in Pytorch. We used an Adam optimiser algorithm with an exponential learning rate decay function, which is a common method of learning rate optimisation used for CNN training (Kingma and Ba, 2014). We then applied a sigmoid activation to the output layer of the CNN to generate predictions between 0 and 1.

Active learning iterations used a batch size of 64, ten epochs and learning rate of 0.001. We used a grid search technique (Mohri, Rostamizadeh and Talwalkar, 2018) to tune hyperparameters of the final model including the number of epochs, batch size and learning rate.

7.4.3.3.3 Active learning framework

We applied an active learning approach to iteratively train and improve the CNN model. The active learning approach is depicted within Figure 7.4-3 and described below.

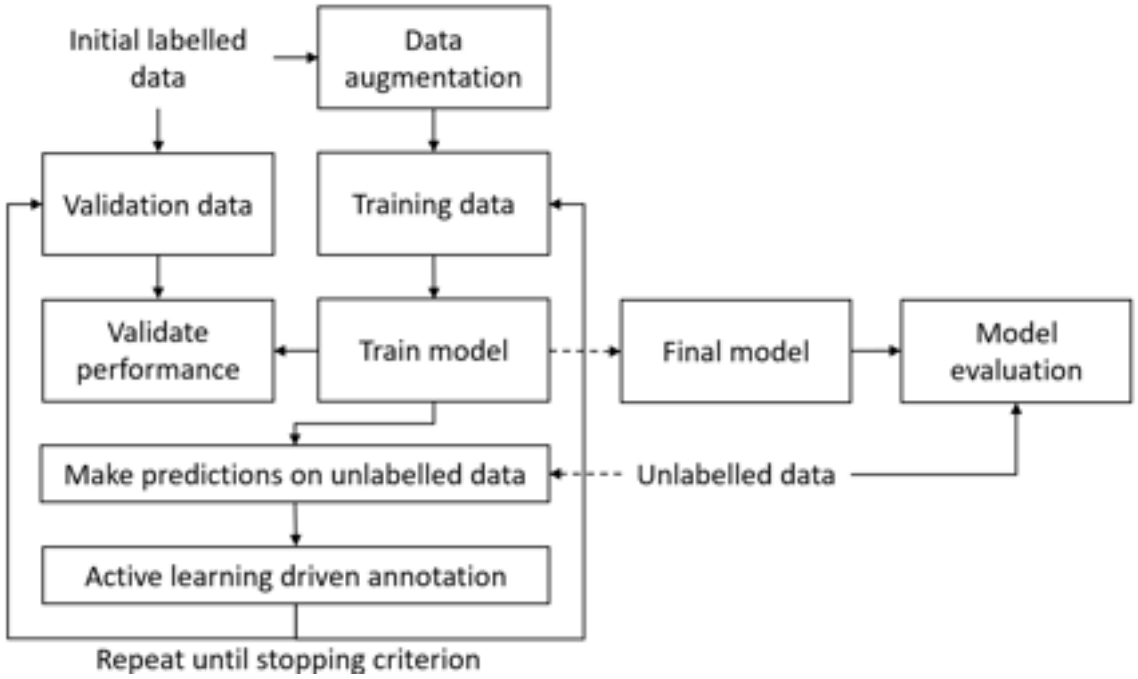


Figure 7.4-3: CNN model development pipeline that incorporates active learning iterations to develop a model for identifying calls of the southern black-throated finch.

We created an initial labelled dataset using unlabelled audio data from one solar-powered bioacoustic recorder (841 hours) and one Audiomoth recorder (108 hours). We clustered similar audio signals within these unlabelled data using the cluster analysis feature of Kaleidoscope Pro, which applied a hidden Markov model (Wildlife Acoustics, 2019). Clusters were manually reviewed and all identified target calls were labelled (n = 254). We also labelled non-target sounds that were distributed throughout the clusters (n = 661). We augmented the initial training data by creating triplicate versions of each target call shifted at random horizontal (time domain) positions, which is a common technique to artificially increase the size of a training set (Stowell, 2022). An initial model was trained from this dataset.

Following the creation of an initial labelled dataset we developed the model using a series of active learning iterations. For each iteration we used the model to make predictions on a new set of unlabelled data, which comprised audio data from one solar-powered bioacoustic recorder (between 833 and 1402 hours) and one Audiomoth recorder (between 80 and 133 hours). Model predictions were in the form of a

logit, on a scale of 0 to 1 (Monarch, 2021), where 0 represented the lowest probability of being the target call and 1 represented the greatest probability. Human annotators then labelled all predictions with a logit of greater than 0.5. The 0.5 logit cut-off was selected to prioritise the annotation of data likely to include target calls, which are rare within the unlabelled data, while also annotating signals that the model identified with the least certainty and would gain the greatest amount of new information, i.e. around the 0.5 logit; (Roh, Heo and Whang, 2019; Monarch, 2021). Through this active learning driven annotation process we grew the number of audio frames in the training data to 1,073 audio frames containing target calls and 5,786 non-target audio frames within five iterations (Table 7.4-1). All labels were reviewed by a primary annotator (J.V.O.) with 5 years' experience with SBTF prior to the next iteration of model training.

Each successive model iteration was trained on 70% of the training data and validated against 30% of the training data. The train/validation split approach minimises potential overfitting of the model on small samples sizes, when compared to a cross-validation approach (Vabalas *et al.*, 2019). We chose F_1 score as our validation metric, which is a standard performance metric that includes information on both model precision and recall (Mohri, Rostamizadeh and Talwalkar, 2018,; Stowell, 2022). Iterative model training stopped when the stopping criterion was reached, which was when successive model iterations did not reduce the F_1 score. A final model was then trained using the combined training and validation data. A separate test dataset was used for final testing/evaluation of the model, which is described in Section 7.4.3.4.1.

Table 7.4-1:The number of target and non-target audio frames used within each model iteration

Iteration	Number of target audio frames	Number of non-target audio frames	Train:validation split
1	254	661	70:30
2	314	2642	70:30
3	1044	4,877	70:30
4	1073	5786	70:30
Final model	1073	5786	100:0

7.4.3.4 Model evaluation

7.4.3.4.1 Test dataset

Evaluation of models trained on data labelled through an active learning framework require evaluation designs beyond the standard method of testing the model on labelled data that are reserved from the initial labelled dataset, i.e. the 'hold-out test set' (Mohri, Rostamizadeh and Talwalkar, 2018; Stowell, 2022). This is because all labelled data available were identified using either hidden Markov model clustering in the Kaleidoscope Pro software (Wildlife Acoustics, 2019), or through the active learning process, which biases data to the model and may not be representative of the broader unlabelled data (Settles and Craven, 2008; Roh, Heo and Whang, 2019).

Therefore, to evaluate the final model, we used an unlabelled dataset that contained 2,735 hours of audio recorded from four sites with solar-powered bioacoustic recorders and five sites with Audiomoth recorders. These sites were independent of those used for the model training. The final model was run over the entire test dataset to predict if each audio frame (1.8 seconds in duration, with a 50% overlap between frames) contained a target call, resulting in c. 10.9 million predictions. We created a test dataset using a stratified random sampling approach. We selected a random sample of 500 audio frames from each 0.01 increment of prediction scores. Prediction scores were in the form of logits on the scale of

0 to 1 (refer to Section 7.4.3.3.2). When 0.01 logit increments had less than 500 audio frames, we included all audio frames within that logit increment. In total, the test dataset included 12,278 audio frames. The primary annotator manually labelled all audio frames in the test dataset.

7.4.3.4.2 Managing uncertain labels

Automated call recognition relies on the accurate identification of calls within the sample data. As noted previously, even highly trained experts can have difficulty achieving high annotation reliability from real-world recordings. In our study the accuracy of annotations is limited by factors such as similar vocalisations of co-occurring species, attenuation of the target calls at distance and overlap with other environmental noise.

Experience of annotators within our study suggested certain sounds cannot be definitively labelled as either 'SBTF' or 'Not SBTF'. These include calls that were substantially attenuated by distance from the microphone or were obscured by other sounds in the same frequency range, as well as short contact calls of SBTF, which are acoustically similar to other estrildid finches within the study area. We therefore allowed annotators to label audio frames into three categories, 'SBTF', 'Uncertain' and 'Not SBTF', following rules supplied to each annotator (Table 7.4-2).

For our model, we used a 'classification with a reject option approach', which allowed audio frames to not be classified (i.e. labelled as uncertain) when the model predictions were ambiguous (Bishop, 2006; Thakur et al., 2019). Classification with rejection uses a threshold value that determines the threshold above which the model's predictions are labelled as 'uncertain'. For the model output of our CNN architecture, prediction scores closest to the logit of 0.5 have the highest uncertainty. The threshold value determined the distance away from the logit 0.5 that we included in the 'uncertain' category. We chose this threshold value to match the proportion of uncertain audio frames in the final model's output to the proportion of uncertain audio frames labelled by the primary annotator. Thus, the final model achieved the same likelihood of classifying an audio frame as uncertain (the reject option) as an annotator of labelling an audio frame as uncertain. This approach allowed us to map the CNN predictions into the same three categories used by the annotators and facilitate a direct evaluation of annotation performance among the model and annotators.

Table 7.4-2: Audio annotation rules provided to annotators

Label	Description
SBTF (southern black-throated finch)	You have evaluated the detection to contain a SBTF call. This may include any type of SBTF call. You need to be certain, or confident, that a call in the detection was made by one (or multiple) SBTF.
Not SBTF	You have evaluated the detection to not contain a SBTF call. You need to be certain, or confident, that no calls in the detection were made by SBTF.
Uncertain	You are not confident that one (or multiple) call/s within the detection were made by a SBTF or were made by another species.

7.4.3.4.3 Model performance

We evaluated the performance of the final model and all preceding active learning iterations against the test dataset. We converted all labels and model predictions to an ordinal scale, with 'Not SBTF' assigned a value of 1, 'Uncertain' a value of 2, and 'SBTF' a value of 3. We evaluated model performance using macro-averaged mean absolute error (MMAE). MMAE is a common evaluation metric for ordinal classification (Cardoso and Sousa, 2011) and accounts for imbalanced datasets (Baccianella, Esuli and Sebastiani, 2009). We did not use the 'standard metrics' of performance recommended by Stowell (2022),

which include precision, recall, F-score and area under the curve metrics, as MMAE accounts for the ordinal nature of our model predictions and labels compared to categorical data that is assumed by the 'standard metrics' (Cardoso and Sousa, 2011). We chose an analysis suitable for ordinal data as the 'uncertain' label is considered closer to the label 'SBTF' and 'Not SBTF' than 'SBTF' and 'Not SBTF' are to each other. We estimated confidence intervals of the MMAE using bootstrapping (n = 10,000). We conducted all analyses using the imblearn version 0.8 package in Python.

7.4.3.4.4 Inter-annotator agreement

Two experts (B.D. and E.M.) independently annotated a random subset of 9.5% (n = 1,165) of the test dataset as secondary annotators. Both experts had over 10 years of experience working on the species. The annotation process was blind, with each expert not receiving annotations from the other expert or primary annotator (J.V.O.).

We assessed inter-annotator agreements using the agreement coefficient Krippendorff's alpha (α) (Krippendorff, 2011; Monarch, 2021). We then bootstrapped (n = 10,000) the distribution of each agreement coefficient and calculated 95% confidence intervals, as recommended by Hayes and Krippendorff (2007). It is generally regarded that values above 0.8 represent good agreement among annotators, while values within the range of 0.667 to 0.8 represent tolerable agreement among annotators (Reidsma and Carletta 2008).

We assessed agreement among the final model, the primary annotator and expert secondary annotators using pairwise combinations of α and tested for differences among the pairwise combinations using a Kruskal-Wallis Test. We tested the effect of excluding 'uncertain' labels during model evaluation by calculating agreement coefficients with 'uncertain' labels retained and removed using a Wilcoxon rank-sum test. We also tested for a difference between the agreement coefficients of data recorded on solar-powered bioacoustic recorders compared to Audiomoth devices using a Wilcoxon rank-sum test. We undertook all significance tests using Scipy version 1.10.0.

7.4.4 Results

7.4.4.1 CNN model development with active learning

The CNN model performance stabilised within five active learning iterations (Figure 7.4-4). The majority of model performance improvement was achieved within the first two iterations, with a 34.3% decrease in macro-averaged MAE between iterations one and three, compared to a 12.1% decrease in macro-averaged MAE between iterations three and the final model (Figure 7.4-4). Most of the training data added between iterations one and three consisted of non-target calls (Table 7.4-1), which led to rapid improvements in the model's ability to differentiate non-target sounds, resulting in a reduction in false positives.

Out of a total of 10,920,849 audio frames of unlabelled data used for testing, the final model predicted that 3,024 audio frames contained a SBTF call (0.028%), 4,581 audio frames were uncertain (0.042%), while the remaining 10,913,244 audio frames (99.9%) did not contain a SBTF call. Southern black-throated finch calls were detected from all locations; however, the frequency of SBTF calls within the audio data varied among recorder locations, ranging from 0.002% to 0.060% of the audio frames.

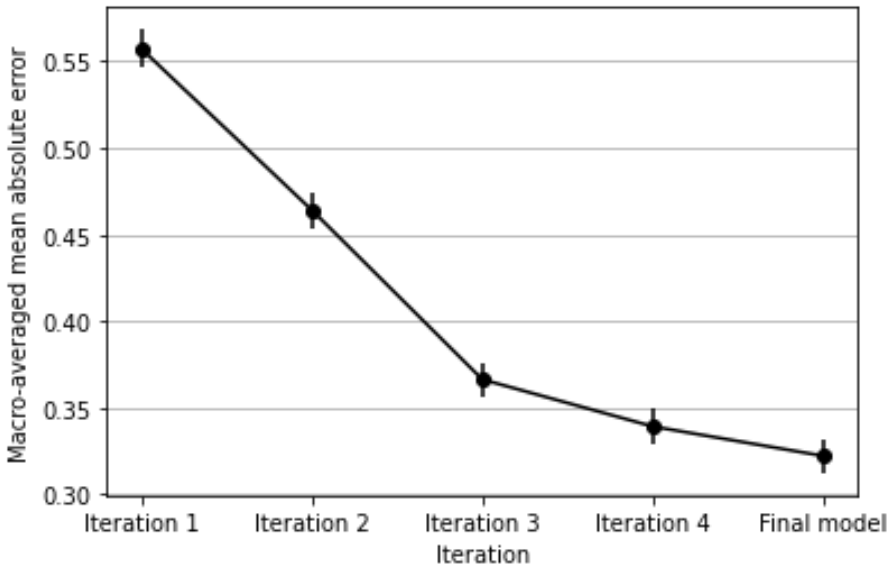


Figure 7.4-4: Evaluation performance of each model iteration. Error bars show 95% confidence intervals.

7.4.4.2 Inter-annotator agreement

Overall, the results showed good agreement among the model and human-annotators, with agreement coefficients ranging from 0.760 α to 0.878 α (Figure 7.4-5). The highest agreement coefficients values were observed between the primary annotator and expert two (0.878 α) and between the primary annotator and the final model (0.859 α). The agreement coefficients were similar between the final model and the two experts (0.760 α and 0.795 α respectively for experts 1 and 2) and between the two experts (0.808 α). Overall, the agreement coefficients show that the final model labelled data within the range of annotation uncertainty that was present among the primary annotator and experts. The final model was more consistent with the primary annotator, who annotated the training data, compared to the experts who independently annotated test data only.

The removal of audio frames labelled as 'uncertain' by the primary annotator during model evaluation caused a small but significant increase in the agreement between the final model and the primary annotator (mean increase 0.016 α, $p<0.01$, Figure 7.4-6). There was reasonable agreement between the audio frames labelled by both the final model and the primary annotator as 'uncertain, with 33.4% of the audio frames labelled 'uncertain' by the primary annotator also being labelled 'uncertain' by the final model. This is compared to the final model assigning an 'uncertain' label to 12.3% and 7.55% of the audio frames which the primary annotator labelled as 'SBTF' and 'not SBTF', respectively (Figure 7.4-6).

Inter-annotator agreement was greater for data recorded on solar-powered bioacoustic recorders than Audiomoths (0.824 compared to 0.639; Figure 7.4-7).

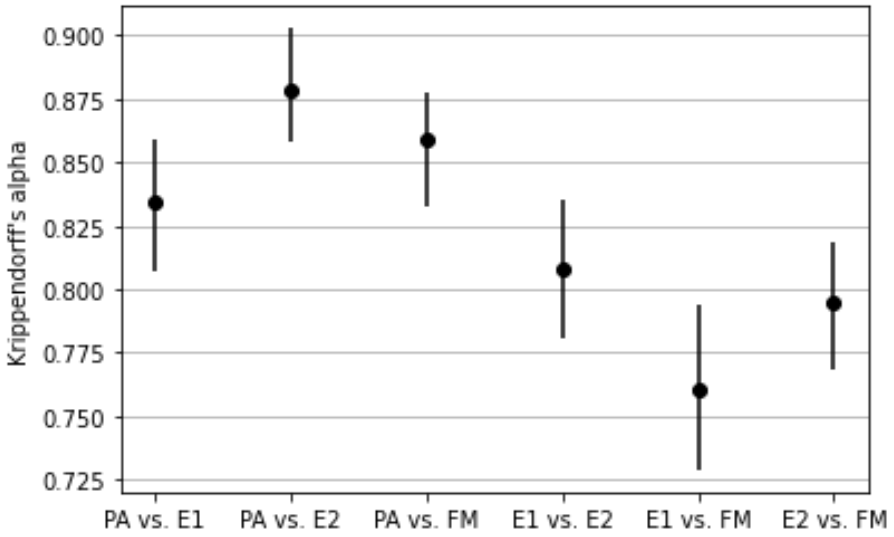


Figure 7.4-5: Pairwise comparison of inter-annotator agreement among the primary annotator (PA), experts one and two (E1 and E2) and the final model (FM). Error bars show 95% confidence intervals.

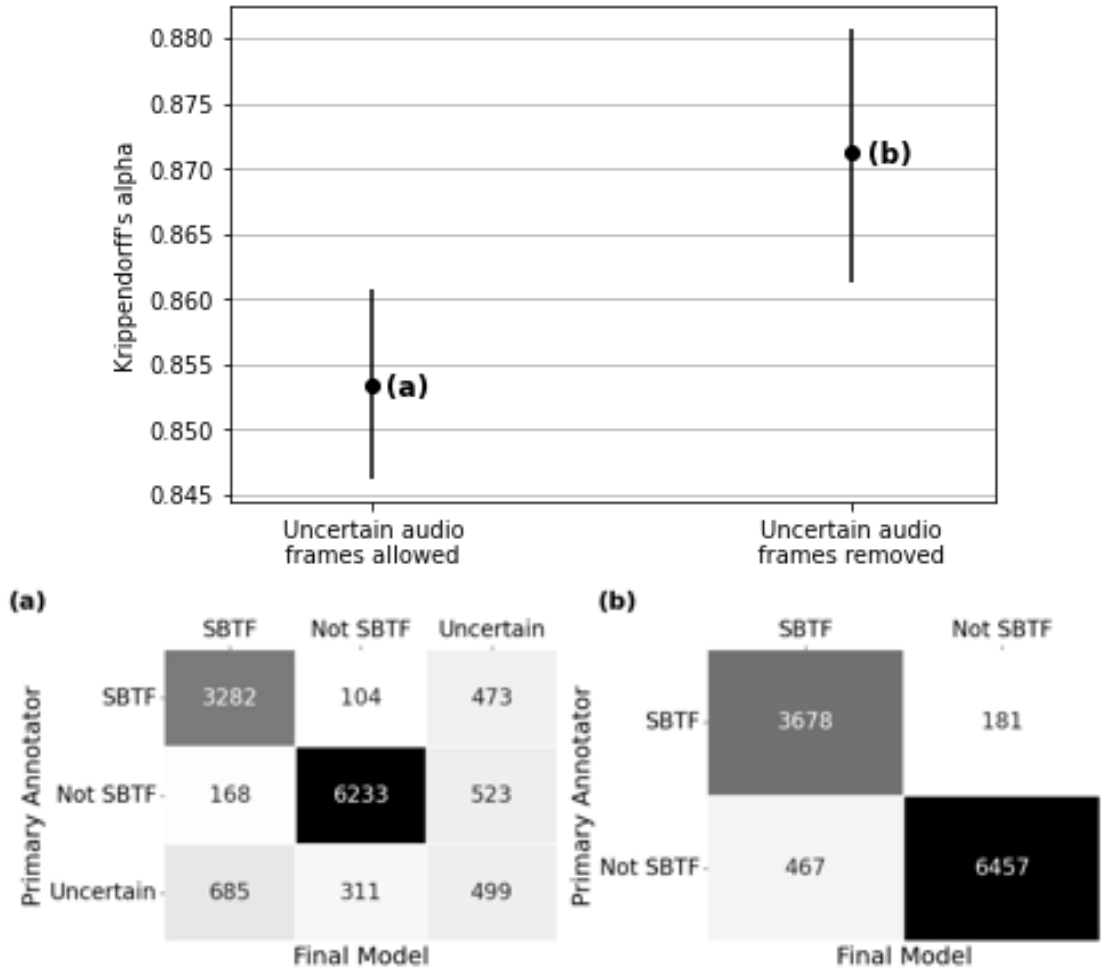


Figure 7.4-6: Comparison of agreement between the primary annotator and the final model when the evaluation process has allowed 'uncertain' audio frames (a), compared to when 'uncertain' audio frames, as assessed by the primary annotator, have been excluded (b). Error bars show 95% confidence intervals. Confusion matrixes are presented below each option as heat plots. The heat plot shading and numbers represent the count of audio frames labelled in each class by the primary annotator compared to the final model.

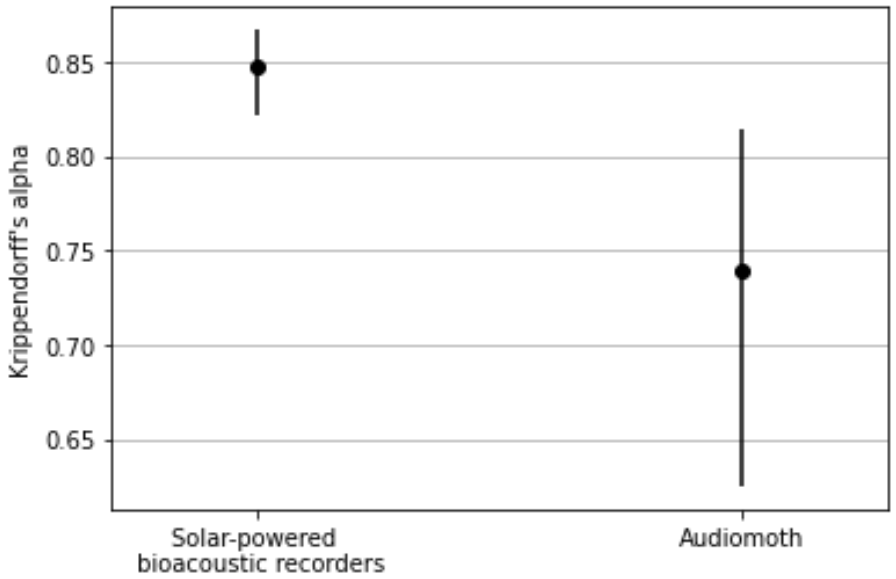


Figure 7.4-7: Agreement among all annotators, including the primary annotator and two experts, for audio frames captured using solar-powered bioacoustic recorders (Frontier Labs) and Audiomoths (Hill *et al.*, 2019). Error bars show 95% confidence intervals.

7.4.5 Discussion

Our study demonstrates that active learning is an effective strategy for building machine learning (ML) classifiers for species with limited labelled training data. Using as few as five active learning iterations, we generated a final labelled dataset of 1,073 target calls and 5,786 non-target sounds. This was sufficient to create a model with classification abilities comparable to experts familiar with the species. Active learning focusses scarce human resources to annotate data that are most valuable for model performance (Van Engelen and Hoos, 2020), primarily records with greater uncertainty. In our study, we found that the active learning framework selected audio frames for annotation that included non-target calls with similarity to the target call, as well as sounds that were dissimilar to the target call due to over-fitting in early model iterations. Over-fitting occurs when the model learns noise in the training data instead of the underlying pattern of the target call (Bishop, 2006). Annotation of these uncertain calls had a measurable impact on model performance. As such, active learning can reduce the time and effort required to develop labelled training data for building classifiers (Priyadarshani, Marsland and Castro, 2018; Stowell, 2022).

Active learning reduces the cost of developing labelled datasets, which form the foundation of successful call detection models (Van Engelen and Hoos, 2020). We estimate that 0.028% of our unlabelled data comprises the target call, making manual screening of our unlabelled data resource intensive. Manually obtaining an equivalent number of target calls to our final dataset (n = 1,073) by reviewing our unlabelled audio data would have required the review of approximately 1,916 hours of audio data. Active learning provided an approach to overcome this barrier to developing a call detection model for the target species. As many wildlife species do not have existing labelled datasets, an active learning approach has general applicability to the field of bioacoustics, particularly for species whose calls are rare in long-term field recordings (Ricci, Rokach and Shapira, 2022). Development of open access methods to improve call recognisers for these rare and difficult to survey species, many of which are also threatened (Loiseau *et al.*, 2020), will contribute to research and monitoring that removes data deficiencies for these species and

thus improves policy and conservation outcomes (Davies, Margules and Lawrence, 2004; Sekercioglu *et al.*, 2008; Loiseau *et al.*, 2020).

Active learning is a powerful method for bioacoustic deep learning, but it poses certain challenges that deviate from the 'standard recipe' of bioacoustic deep learning described by Stowell (2022). One challenge is that the active learning process biases labelled data to the model since the model's predictions guide the annotation process. Resulting labels therefore cannot be used in the model's evaluation (Roh, Heo and Whang, 2019; Ricci, Rokach and Shapira, 2022). Chambert et al. (2018) and Ruff et al. (2020) addressed this issue by selecting audio frames for review post-processing based on their model's predictions. We extended this approach to account for a highly imbalanced dataset. Evaluation of a highly imbalanced dataset through random selection alone would require unfeasibly large numbers of audio frames to be manually reviewed to ensure sufficient target calls were captured to give reliable evaluation metrics (Raeder, Forman and Chawla, 2012). For example, where target calls make up 0.028% of the unlabelled data, which is the case for our unlabelled data, approximately 357,000 audio frames would require manual review to capture 100 target calls. To overcome this challenge, we applied a random selection approach that was stratified across the model's predictions (logits). While this approach substantially reduced class imbalance within our evaluation data and allowed for the calculation of reliable evaluation metrics (Raeder, Forman and Chawla, 2012), the nature of this approach alters the distribution of data and prohibits evaluation metrics being generalised to the unlabelled data. Additional research is needed to investigate more appropriate evaluation methods for highly imbalanced and unlabelled test data.

An active learning strategy requires consideration of biases that may be introduced during annotation (Monarch, 2021). While annotation bias is a consideration for all ML datasets, the active learning framework allows ML models to be generated from smaller datasets and potentially developed by fewer annotators compared to, for example, a publicly available call database such as Xeno-canto (Vellinga and Planqué, 2015). In our study, the final model's predictions most closely reflected the primary annotator's labels, compared to the expert's labels, which suggests that the model is replicating biases of the training data (Roh, Heo and Whang, 2019; Koenecke *et al.*, 2020). Performance improvements may therefore be achieved through the inclusion of labelled training data from additional annotators experienced with the species, or further investigation of the vocal repertoire of the target species compared to potential false positives. Broadly, our results highlight the importance of considering annotation biases, particularly for datasets created by a small group of annotators.

The use of multiple annotators to label training data requires consideration of inter-annotator agreement and how this affects ML model performance. Classifying wildlife vocalisations with certainty is unfeasible for many species. While ML models may theoretically be able to outperform inter-annotator agreement, the level of inter-annotator agreement remains a useful benchmark to assess model performance (Boguslav and Cohen, 2017; Richie, Grover and Tsui, 2022). In the field of natural language processing, where labels are often subjective, inter-annotator agreement metrics are a common tool to assess label quality and model performance (Pustejovsky and Stubbs, 2012). Our results support these arguments with the congruence among annotators providing a useful benchmark to assess model performance and identify potential areas of improvement in the training data.

The standard approach to build a bioacoustic classification model with ML often explicitly or implicitly assumes that labels are accurate, with limited scope for the inclusion of ambiguous vocalisations (Cabitza *et al.*, 2020; Otani *et al.*, 2020; Campagner *et al.*, 2021). Our results show that exclusion of ambiguous vocalisations from the evaluation dataset significantly inflated the model's evaluation metrics. Our findings therefore agree with those of Cabitza et al. (2020) from the medical literature, that model performance is overestimated if inter-annotator agreement is not accounted for. While the effect size for our data was small, due to a good agreement among annotators, the overestimation of model performance is theoretically negatively correlated with inter-annotator agreement (Cabitza *et al.*, 2020). We

recommend that the development of models for species with ambiguous vocalisations account for inter-annotator agreement in design. We demonstrated how this can be accomplished using multiple annotators and adoption of a classification with a 'reject' option approach (i.e. the 'uncertain' category), which allows the model results to be directly compared to those of the annotators (Bishop, 2006; Campagner, Cabitza and Ciucci, 2019). Other approaches applied within the natural language processing and medical imaging fields include the development of a 'gold-standard' set of labels through the use of multiple annotators or label cleaning (Pustejovsky and Stubbs, 2012; Karimi *et al.*, 2020), or incorporating annotator or label uncertainty into the machine learning process (Nguyen, Valizadegan and Hauskrecht, 2014; Hüllermeier and Waegeman, 2021). These approaches all require the evaluation of inter-annotator agreement within the study design, which is supported by the results of our study.

Assessment of inter-annotator agreement allows investigation of the sources of disagreement and potential improvements to annotator accuracy and label reliability (Duc *et al.*, 2021; Monarch, 2021). In our study, inter-annotator agreement was generally high for the entire unlabelled dataset, however inter-annotator agreement for data recorded on Audiomoth devices was substantially lower when compared to solar-powered bioacoustic recorders. Two factors may explain this difference: the placement of Audiomoth devices in proximity to nests and differences in the recording quality of Audiomoth and bioacoustics recorders.

Having been placed near active SBTF nests, Audiomoths would likely have captured a broader range of SBTF calls than bioacoustics recorders placed at foraging and drinking sites, including numerous softer calls unlikely to be heard at distance (see Shephard, Pridham and Forshaw, 2012). Annotators would likely differ in their familiarity with these softer calls, resulting in lower inter-annotator agreement on Audiomoth recordings when compared with recordings from bioacoustic recorders (which are more likely to have captured the more commonly heard, louder and distinctive 'long'/'pew' call of SBTF).

Differences in the volume/amplitude and quality of recordings captured by bioacoustic recorders and Audiomoths may also have affected annotators assessment of calls, with bioacoustics recorders having a signal-to-noise ratio (SNR) of 80 dB compared with 42 dB in Audiomoths (Roe *et al.*, 2021; Open Acoustic Devices, n.d.). The higher SNR of bioacoustic recorders would afford greater sound fidelity and less background noise, thereby providing recordings that align more closely with the annotators' field experience (Darras et al., 2020; Turgeon et al., 2017).

7.4.6 Conclusions

Our study investigated the efficacy of active learning as a framework for building deep learning models in a setting where limited training data were available and the cost to obtain training data was high. Using field recordings of a threatened species, the southern black-throated finch, we successfully developed and demonstrated the value of an open access active learning method that considered imbalanced and unlabelled data and ambiguous vocalisations, which are common barriers to constructing call recognition models for rare or cryptic species, and other species that vocalise infrequently. Our results demonstrate the utility of these methods in developing effective call recognisers for rare and difficult to survey species.

Our results also support the assessment of inter-annotator agreement during the development of call recognition models, with the assessment of inter-annotator agreement allowing for more accurate evaluation of model performance and identification of annotator biases, as well helping identify areas for improvement in training data.

7.5 Development of a data pipeline to localise automated radio telemetry tower data

This section has been published as a scientific paper in the journal *Animal Biotelemetry*. Bibliographic details of this scientific paper are:

> van Osta, J. M., Dreis, B., Grogan, L. F., & Castley, J. G. (2024). A location fingerprinting approach for the automated radio telemetry of wildlife and comparison to alternative methods. *Animal Biotelemetry*. 12(1), 24. https://doi.org/10.1186/s40317-024-00379-w

7.5.1 Abstract

Automated radio telemetry (ART) systems enable high-temporal resolution data collection for species unsuited to satellite-based methods. A challenge of ART systems is estimating the location of radio tagged animals from the radio signals received on multiple antennas within an ART array. Localisation methods for ART systems with omni-directional receivers have undergone rapid development in recent years, with the inclusion of machine learning techniques. However, comparable machine learning methods for ART systems with directional antennas are unavailable, despite their potential for improved accuracy and greater versatility. To address this, we introduce an open-source machine learning-based location fingerprinting method for directional antenna-based ART systems. We compare this method to two alternative localisation approaches. Both alternatives use relative signal strengths recorded among multiple antennas to estimate the signal's angle of arrival at each receiver. In the 'biangulation' approach, the location is estimated by finding the intersection of these angles from two receivers. In contrast, the 'linear regression' approach uses a linear regression model to estimate the distance from the receiver along the angle of arrival, providing a location estimate. We evaluate these methods using an ART dataset collected for the southern black-throated finch (*Poephila cincta cincta*), in the Desert Uplands Bioregion of Queensland, Australia.

The location fingerprinting method performed slightly better than the best performing alternative, the linear regression method, with mean positional errors of 308 m (SE = 17.7) and 335 m (SE = 18.5), respectively. The biangulation method performed substantially worse, with a mean positional error of 550 m (SE = 42.9, median = 540 m). Improved accuracy was observed with shorter distances between transmitters and receivers, higher signal strengths, and a greater number of detecting receivers, suggesting that increasing receiver density improves localisation accuracy, albeit with potential trade-offs in system coverage or cost. Furthermore, shorter pulse intervals of transmitters resulted in greater accuracy, highlighting the trade-offs among battery life, transmitter weight and radiative power. The open-source location fingerprinting method offers an improved and versatile localisation approach suitable for a wide variety of ART system designs, addressing the challenge of developing study-specific localisation methods using alternative approaches.

7.5.2 Introduction

The field of movement ecology has seen rapid growth, driven by the development of tracking technologies that enable the monitoring of animal movements across time and space in diverse environments (Allen and Singh, 2016). These technologies encompass a range of methods including radio telemetry, satellite-based systems (e.g., GPS and Argos), accelerometry, and wireless sensor networks (Thomas, Holland and Minot, 2011; Katzner and Arlettaz, 2020; Ripperger *et al.*, 2020). Each method has its inherent limitations, such as tag mass, cost, accuracy, frequency of location estimates, and data retrieval options, with no one-size-fits-all solution (Lindberg and Walker, 2007; Thomas, Holland and Minot, 2011; Zekavat and Buehrer, 2011).

Among these technologies, Global Positioning Systems (GPS) stand out for their capacity to deliver location estimates with high spatiotemporal resolution (Thomas, Holland and Minot, 2011; Zekavat and Buehrer, 2011). The ongoing miniaturisation of GPS transmitters has resulted in multiple manufacturers offering minimum tag weights between 2.6 g and 5 g. Despite this miniaturisation, GPS transmitters remain too large to affix to animals weighing less than approximately 100 g, assuming maximum transmitter weight of no more than 5% of an animal's weight (Kenward, 2000). In comparison, radio transmitters, with minimum weights of around 0.13 g (e.g. Lotek NanoPin), remain an important tool for the telemetry of smaller animals weighing less than 100 g (Griffin et al., 2020; Smetzer, 2018; Thomas et al., 2011).

Traditional manual radio telemetry techniques are often limited by the time and cost constraints of locating and resighting tracked animals (Gottwald *et al.*, 2019). Automated radio telemetry (ART) systems have emerged as a promising development, enabling high-temporal resolution data collection remotely from radio tagged animals (Taylor *et al.*, 2017; Griffin *et al.*, 2020). ART systems can use lightweight and low-cost radio transmitters, making them a versatile approach that has been used to track over 180 species of birds, bats and insects to date (Taylor *et al.*, 2017; Griffin *et al.*, 2020).

Unlike satellite-based methods, ART systems do not directly receive transmitter locations. Instead, transmitter locations are estimated from the relative signal strength (RSS) of the transmitter's signal recorded by one or more receivers (Kays *et al.*, 2011; Ward, Sperry and Weatherhead, 2013; Gottwald *et al.*, 2019; Griffin *et al.*, 2020; Fisher *et al.*, 2021; Paxton *et al.*, 2022). There is substantial variation in the design and objectives of ART systems that result in no versatile approach to localisation (Mendoza-Silva, Torres-Sospedra and Huerta, 2019).

The design of ART systems can be broadly split into omni-directional and directional systems. Omni-directional systems typically use one isotropic antenna per receiver. The antenna receives signals in an approximately uniform pattern, 360 degrees around the antenna, however little information is provided with regards to the angle of arrival (AOA), or bearing, of the signal to the receiver (Paxton *et al.*, 2022; Wallace *et al.*, 2022). In contrast, directional systems typically use multiple antennas (3-6) per receiver, with each antenna orientated in different directions to allow the AOA of the signal to be estimated (Taylor *et al.*, 2017; Griffin *et al.*, 2020; Fisher *et al.*, 2021). In general, directional systems often have a longer range than omni-directional systems but have a higher cost per receiver (Griffin *et al.*, 2020).

A variety of methods have been employed to localise transmitter position from omni-directional and directional ART systems. Lateration techniques (Figure 7.5-1A) are common for omni-direction systems (Wallace *et al.*, 2022). Lateration estimates the distance of a transmitter from multiple receivers using the decline in RSS with distance from transmitter. Lateration requires simultaneous detections on at least three receivers in order to identify a point at which distances to the receivers overlap (Mendoza-Silva, Torres-Sospedra and Huerta, 2019; Wallace *et al.*, 2022). Since lateration does not require an AOA of the signal, it has been widely applied to omni-directional systems (Tyson, Fragueira, *et al.*, 2024). However, lateration is generally not applied to ART designs with directional receivers as data pertaining to the AOA is wasted (Mendoza-Silva, Torres-Sospedra and Huerta, 2019). ART systems with directional receivers therefore employ methods that rely on angulation, which is where the AOA of the transmitter to the receiver is calculated using the relative signal strength recorded among the multiple directional antennas per receiver (Mendoza-Silva, Torres-Sospedra and Huerta, 2019). Once the AOA has been calculated, locations are estimated using biangulation or triangulation, which is the point at which the AOAs from multiple receivers intersect (Figure 7.5-1B) (Kays *et al.*, 2011; Gottwald *et al.*, 2019). Alternatively, the distance between the transmitter and the receiver can be inferred from the RSS, typically based on the assumption of a linear decline in RSS with increasing distance (Figure 7.5-1C) (Fisher *et al.*, 2021).

Location fingerprinting is a method that has been widely employed for indoor positioning systems and has recently been developed for ART systems (Mendoza-Silva, Torres-Sospedra and Huerta, 2019). This method

uses machine learning based techniques to develop a model that relates the RSS of signals received to the locations of known reference points (Alhomayani and Mahoor, 2020). The model is then used to estimate the location of new signals (Figure 7.5-1D). Tyson et al. (2024) and Wallace et al. (2022) have developed location fingerprinting methods for omni-direction systems and achieved accuracies of approximately 10-30 m. However, there is no comparative open-source system available for directional receiver systems, nor studies comparing the effectiveness of location fingerprinting to alternative localisation approaches.

Location fingerprinting offers a highly versatile approach with potential for improved accuracy compared to alternative methods. The method requires the development of study-specific radio fingerprinting models (Figure 7.5-1D), which account for receiver specific variation in topography, vegetation density and radio tower design, as well as study-specific variation in species behaviour, radio transmitter power, receiver spacing and radio pulse interval (Tyson, Fragueira, *et al.*, 2024). All of these factors are difficult to factor into alternative approaches (Gottwald *et al.*, 2019; Mendoza-Silva, Torres-Sospedra and Huerta, 2019). Using study-specific location fingerprint models allows the location fingerprinting approach to generally achieve higher accuracies than alternative approaches (Mendoza-Silva, Torres-Sospedra and Huerta, 2019; Alhomayani and Mahoor, 2020; Tyson, Fragueira, *et al.*, 2024). Furthermore, this method offers a low-code approach to localising wildlife positions from ART data as site-specific training data are used to build location fingerprint models through machine learning techniques, rather than through coding bespoke solutions, which would be required to customise alternative approaches (Kays *et al.*, 2011; Gottwald *et al.*, 2019; Fisher *et al.*, 2021).

A key requirement of location fingerprinting is the collection of reference points, or 'training data', which are known coordinates where the RSS is measured to develop the fingerprint model. In their location fingerprinting systems for omni-direction systems, Tyson et al. (2024) and Wallace et al. (2022) collected training data over 50 ha and 0.4 ha study areas, respectively, to create a whole-of-array radio fingerprint map. However, directional systems are often used to track animals over study areas one to three orders of magnitude larger (Crewe et al., 2019; Kays et al., 2011; Smetzer, 2018; Smetzer et al., 2021), which can make the collection of training data for whole-of array radio fingerprint maps time and cost prohibitive (Mendoza-Silva, Torres-Sospedra and Huerta, 2019). An alternative approach that has not been explored is to train a fingerprint model for each directional receiver and apply the same model to multiple receivers where they have the same design and are in a similar environment (e.g. consistent topography and vegetation structure). While this would come at a cost of accuracy, it would proportionally reduce the training data required and enable the location fingerprinting method to be applied to large-scale ART systems.

In this study, we present an open-source location fingerprinting method to estimate radio transmitter positions from directional receiver-based ART systems. The method provides a versatile approach to location fingerprinting that allows the development of models that pool training data among similar receivers and thus allows the method to be applied to large-scale ART systems where collection of whole-of-array training data is time prohibitive. We demonstrate the application of this method across a 2,680 ha ART array tracking a threatened Australian bird species, the southern black-throated finch (*Poephila cincta cincta*). We compare the performance of the location fingerprinting model (Figure 7.5-1D) to two alternative localisation methods, namely: (1) finding the intersect between AOAs from pairs of two receivers, hereafter 'biangulation' (Figure 7.5-1B); and (2) using a linear regression model to estimate the distance that a transmitter is located along the AOA, given the RSS, hereafter 'linear regression' (Figure 7.5-1C).

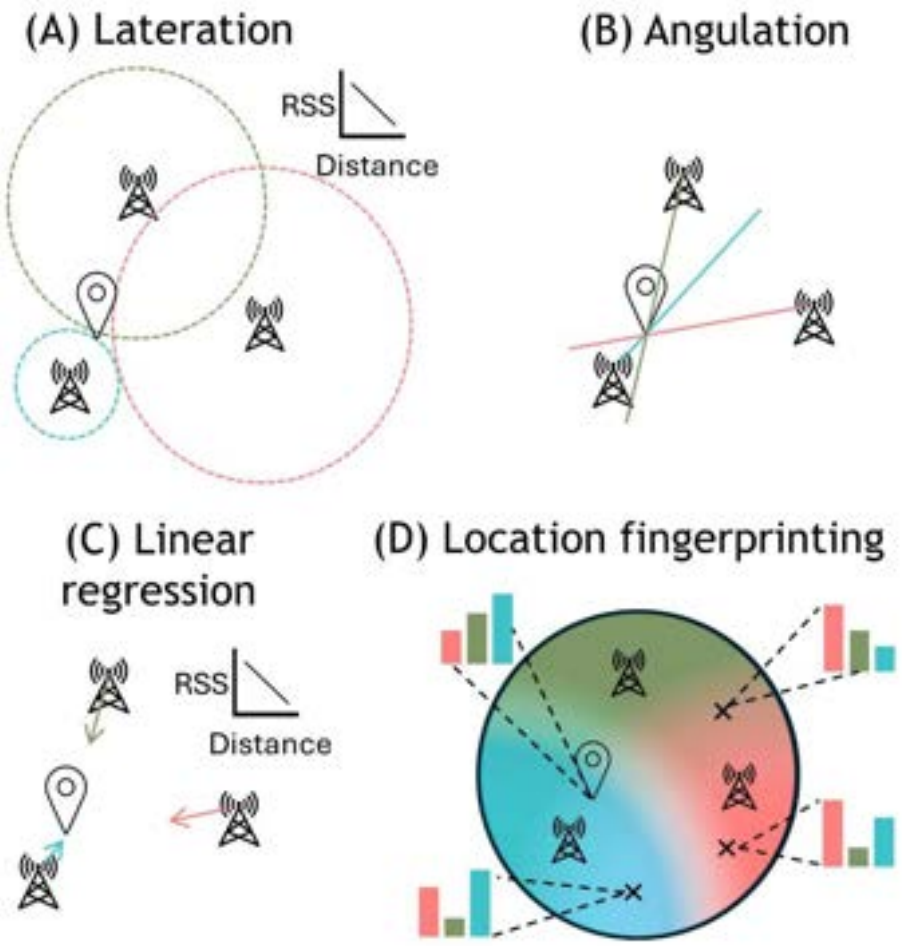


Figure 7.5-1: Localisation methods for automated radio telemetry systems, which estimate location by: A) finding the intersection of distance estimates that are inferred from relative signal strengths (RSSs) on omni-directional receivers; B) finding the intersection of multiple angles of arrival (AOA) produced by directional receivers; C) using the AOA and distance estimates inferred from the RSSs of directional receivers; and D) developing a radio fingerprint map of each receivers' capture area (denoted by the colours red, green and blue), using ground-truthed RSS data.

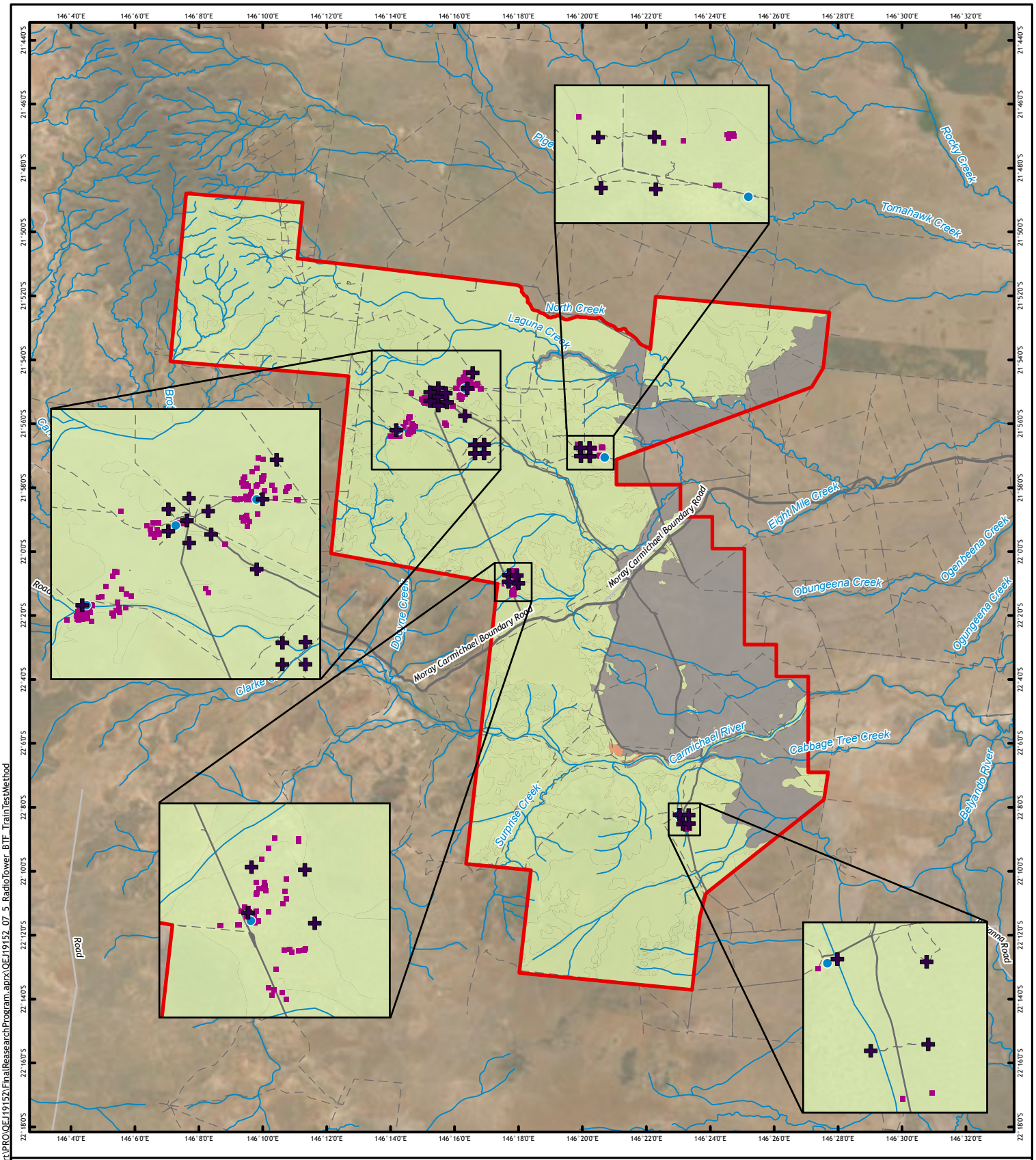
7.5.3 Methods

7.5.3.1 Case study species

The southern black-throated finch (SBTF), *Poephila cincta cincta*, is a small grassfinch (~15 g) in the family Estrildidae. The species is endemic to north-eastern Australia, with a current range that has contracted by over 80% from its historical extent (Commonwealth of Australia, 2023). The species inhabits grassy woodlands and feeds primarily on grass seeds, with movement across the landscape influenced by water and resource availability (Rechetelo *et al.*, 2016a). Movement patterns of SBTF may be characterised as sedentary with a large home range of approximately 319 ha (Section 3.1). On a daily basis, SBTF move through their home range between foraging, drinking and nest locations (Section 3.1). The SBTF also uses all structural layers of the woodlands in which they inhabit, ranging from the ground layer in which they feed, up to the canopy and subcanopy layers in which they perch and nest (Shephard, Pridham and Forshaw, 2012; Rechetelo *et al.*, 2016a).

7.5.3.2 Study area

The study was conducted in a 75,000 ha section of the Moray Downs property (Figure 7.5-2), located within the Desert Uplands Bioregion of Queensland, Australia, which is one of the few remaining strongholds of SBTF (Vanderduys *et al.*, 2016a). Remnant vegetation occurs over 79% of the study area, which is dominated by *Eucalyptus melanophloia* and *Eucalyptus brownii* open woodland ecosystems (DES, 2022). Sub-canopy and shrub layers are generally sparse and commonly include *Acacia* spp. *Bursaria incana* and *Carissa lanceolata*. Other habitats include Acacia woodlands and shrublands and cleared grazing pastures, which are typically not suitable habitat for SBTF (Commonwealth of Australia, 2023).



Legend

- Automted Radio Tower
- Mist-netting site

Training Data Type

- Radio tracked SBTF

- Watercourse
- Major Road
- Minor Road
- Track
- Street/Local - only provides property access

- Project Area

Vegetation Status

- Remnant
- HVR
- Non-remnant

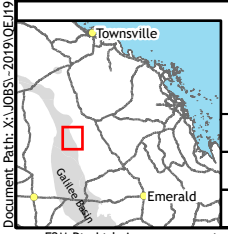
N

Scale 1:300,000 (A4)

0 5 10 15

Kilometres

Coordinate System: GDA2020



Notes:
Aerial Imagery: © ESRI 2021
Cadastre: © DNRME 2021
Watercourse: © Geoscience Australia 2012
Road: © DNRME 2021

Rev	Description	Drawn	Approved	Date
2	Issued for Use	JY	JMC	30/10/2025
0	Issued for Use	GO	JMC	08/08/2025

FIGURE 7.5-2: AUTOMATED RADIO TOWERS AND RADIO TRACKED BTF USED FOR TRAINING AND TESTING LOCALISATION METHODS

BTFRP Final Report
Bravus

Map Number	Job Number	Rev
1 of 1	QEJ19152	2

Document Path: X:\JOBS\-2019\QEJ19152\GIS\Final Research Program Report\PRO\QEJ19152\FinalReasearchProgram.aprx\QEJ19152_07_5_RadioTower_BTF_TrainTestMethod

E2M Pty Ltd gives no warranty in relation to the data (including accuracy, reliability, completeness or suitability) and accepts no liability for any loss, damage or costs (including consequential damage) relating to any use of the data in this map.

7.5.3.3 Automated radio telemetry system

Twenty-seven (27) receiver stations were installed within remnant woodland within the study area in November 2020. Receiver stations were installed as four arrays of four towers, with an additional array of 7 towers surrounded by four individual towers (Figure 7.5-2). Receivers within an array were separated by approximately 500 m. Site-based testing of the receivers found a typical detection distance of approximately 300–800 m depending on whether the SBTF was foraging on the ground or perching (typically 2–10 m above ground height). Thus, foraging SBTF could be reliably detected by at least one receiver when in an array and substantial overlap in the detection ranges among towers was achieved while SBTF were perching. The total area covered by the array after subtracting areas of overlapping coverage among receivers was approximately 2,680 ha, assuming an 800 m detection range of each transmitter. The number of receivers and antennas that simultaneously detected each signal was recorded to model their impact on localisation error.

The receivers remotely and autonomously logged the radio signals received from nearby radio tagged birds. The design of radio telemetry receivers was similar to the Motus Wildlife Tracking System (Taylor *et al.*, 2017). Each receiver comprised four 5-element Yagi-Uda antennas (TDJ-150B 150 MHz) mounted on a 4 m telescopic aluminium tripod (Figure 7.5-3). The antennas were horizontally polarised and oriented to face north, east, south and west. Antennas were spaced ¼ wavelength apart (approximately 0.5 m at 150.6 MHz) and connected to a Lotek SRX800-D1 receiver. Antennas were scanned at 15 second intervals, resulting in a complete scan of all four antennas every minute. Receivers stored the transmitter ID, received signal strength (RSS) measured in decibels, antenna direction and GPS-synced time stamp of the detection. Transmitters were individually coded (Lotek Nanotag NTQB2-2) allowing multiple transmitters to be detected simultaneously while scanning antennas.



Figure 7.5-3: Automated radio telemetry receiver with four directional antennas installed within the study area

7.5.3.4 Radio transmitter attachment

We mist-netted SBTF over eight survey periods, each approximately 12 days in length, between February 2021 and May 2023. A total of 47 SBTF were fitted with Lotek NTQB2-2 transmitters (0.32 g, <2.5% of the bird's weight), across six mist-netting locations that were dispersed among the spatial extent of the ART system (Figure 7.5-2). The number of transmitters attached at each mist-netting location ranged from one to 14. Transmitters were attached by trimming a small patch of feathers on the bird's back and attaching the transmitter to the skin using cyanoacrylate glue (Johnson, Pebworth and Krueger, 1991; Diemer, Wheeler and Nocera, 2014; Rechetelo *et al.*, 2016a). Transmitter function and signal detection were tested using a handheld receiver (Lotek SRX1200 M2) prior to release. Two radio pulse intervals were used within this study, each below the 15 second receiver antenna scan windows used to ensure that at least one antenna would detect a tagged bird when within range of the receiver tower. Transmitters with a 13 second pulse interval (n = 21) and 97 day battery life were initially used based on an estimated transmitter retention time (Diemer, Wheeler and Nocera, 2014). Following initial field surveys and evaluation of tag retention times, we changed to a 3 second pulse interval and 29 day battery life (n = 26) to better reflect the tag retention that was being achieved, which was on average 21.8 days (SD = 22.5 days) and improve manual radio-tracking efficiency.

7.5.3.5 Collection of training and testing data

We collected ART receiver data when the transmitters were at known locations. These data were used to train the localisation models and evaluate the model's accuracy, referred to as 'training' and 'testing' data.

Training data were collected by manually radio tracking tagged SBTF with a 3-element Yagi-Uda antenna connected to a Lotek SRX1200-M2 receiver. We used a hand-held GPS (Trimble Nomad TDC100) to record the location of the bird and the start and end times (GPS-synchronised) of each sighting where the SBTF was in the same location for more than three minutes. We truncated records to a maximum of 20 minutes for balanced data representation among locations. In total, 232 unique locations were recorded across the 47 radio tagged birds, which ranged from one to 27 locations per bird. The average duration of recording was 8.5 minutes (SD = 5.0 minutes; range 3–20 minutes).

Following data collection, we randomly excised 20% (n = 47) of the radio tracked SBTF locations to use as a testing dataset. These testing data were not used for model training, which avoided the leakage of training data into the testing set (Mohri, Rostamizadeh and Talwalkar, 2018). The remaining 80% (n = 185) of the radio tracked SBTF locations were used for the purposes of training the location fingerprinting model and in the linear regression method (described below).

7.5.3.6 Location fingerprinting method

7.5.3.6.1 Location fingerprinting overview

We developed a data processing pipeline to train a location fingerprinting model (Figure 7.5-4A) and then predict locations of radio transmitters using the trained model (Figure 7.5-4B).

Model training (Figure 7.5-4A) is divided into two steps: (1) data pre-processing restructures the raw ART signal data into a format suitable for input to a location fingerprinting model; (2) signals are mapped to their known locations in order to train a location fingerprinting model using an open source automated machine learning platform (LeDell and Poirier, 2020).

Inference (Figure 7.5-4B) combines the signal pre-processing steps (as per the model training) and applies the location fingerprinting model to predict locations of new signal data. Transmitter location is estimated separately for each receiver on which the signal is received. A location averaging function is then applied when transmitters are recorded simultaneously across multiple receivers. The model training and inference methods are described below. All data processing was undertaken using Python (version 3.6.9, Python Foundation).

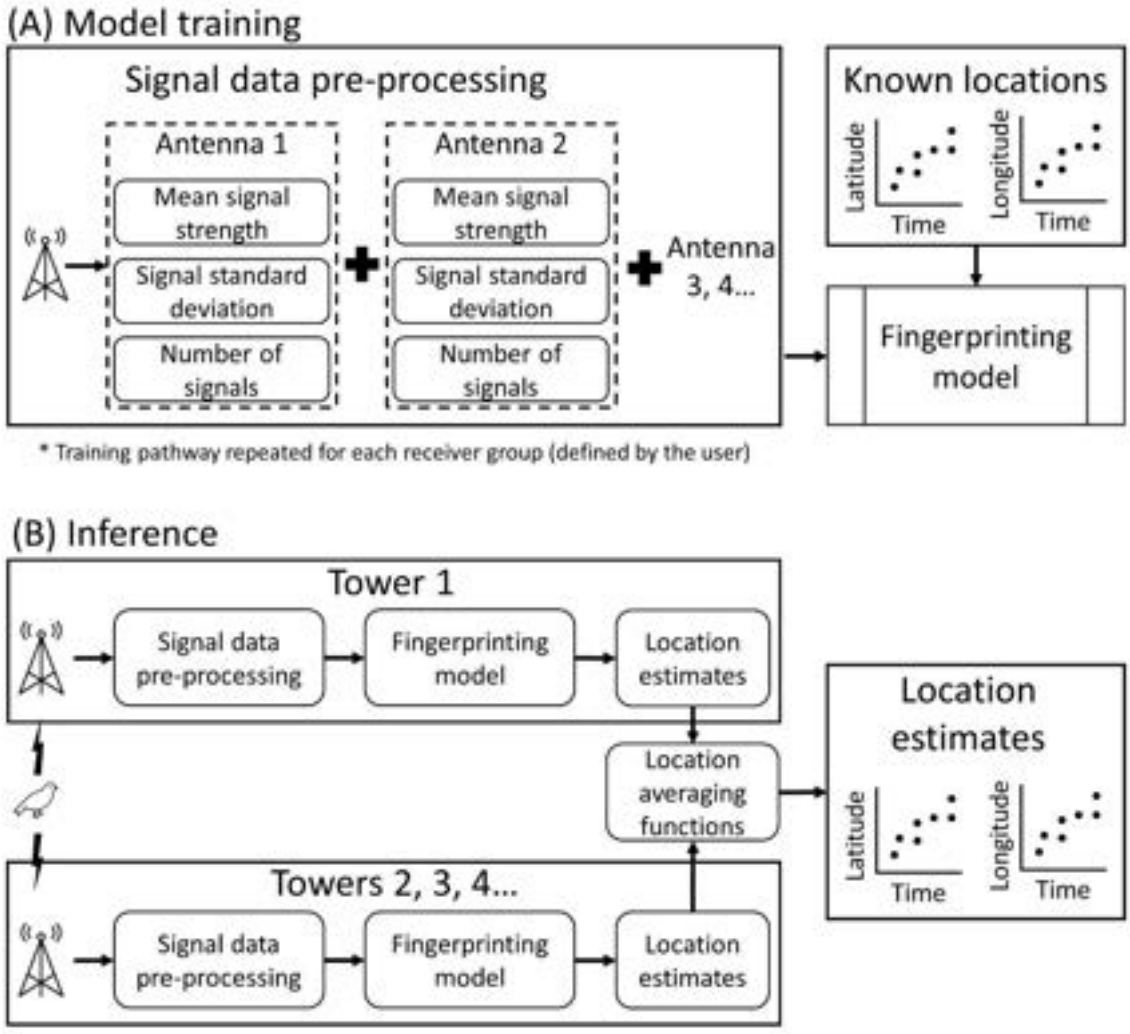


Figure 7.5-4: Data pipelines for the training of the location fingerprinting model (A) and inference of transmitter location (B).

7.5.3.6.2 Receiver groups

The data pipeline uses a receiver-centric fingerprinting approach, whereby a fingerprinting model is trained for each receiver. Fingerprinting models may be grouped among receivers that have similar properties, such as antenna design, topography and vegetation structure. This receiver-centric approach allows users to train and deploy location fingerprinting models for receivers that have limited training data available by pooling training data among receivers (Zhang *et al.*, 2019; Sulaiman *et al.*, 2023). It also provides the versatility to allow receiver-specific fingerprinting models, should sufficient training data be available (Sulaiman *et al.*, 2023). The amount of training data required will be study-specific; however, we provide an assessment of training data requirements in the 'model evaluation' section of the methods.

For this study, we chose to train one location fingerprinting model that was shared among all of the receivers. We did this as no single receiver had training data locations from the breadth of its area of coverage, which was approximately 800 m from the receiver (Figure 7.5-2). Additionally, all receivers within our study used the same design and were located in similar vegetation (eucalypt open woodland) with flat topography.

7.5.3.6.3 Data pre-processing

Manually radio tracked SBTF locations were time-matched to ART receiver data to create a labelled dataset of ART signals received when transmitters were at known locations. Example training data used as input into the data pipeline are provided in the Supplementary Information.

To prepare training data for input to the location fingerprinting model, we calculated the average RSS for each transmitter detection at each antenna within a specified period (*t*). We selected a *t* value of 3-minutes, as the base duration for creating training and testing samples. This time frame was chosen because it corresponds to three complete cycles of the antennas connected to each receiver, balancing the accumulation of more signal data from a SBTF location with the potential of the tracked animal moving. While we used a 3-minute period in this study, the data pipeline accepts any other value of *t*, which may be adjusted based on the frequency and speed of animal movements and pulse interval.

For each period of *t*, the data pipeline creates three variables for each antenna of each receiver. These variables include the mean RSS, a count of the number of signals received and the standard deviation of the signal strength. Thus, for our ART system, which had four antennas per receiver, the data pre-processing produced 12 predictor variables in total. These 12 predictor variables were then used to train the location fingerprinting model to predict the transmitter's easting and northing position, which were the response variables.

To construct the response variables, we normalised the transmitter locations to an X-Y grid, where the origin (0,0) was the location of each receiver and X-Y axes represented the east-west (x) and north-south (y) distance from the receiver. To do this, we converted GPS locations from their geographic coordinate system (represented by latitudes and longitudes) to a projected coordinate system (represented by eastings and northings) using the utm package (version 0.7.0) in Python. We then subtracted the easting and northing coordinate of the receiver from each GPS recorded location to obtain the X and Y distances of the transmitter relative to the receiver. This step was undertaken for each receiver within the ART array independently, allowing one transmitter location to be simultaneously associated with multiple receivers that received the signal within the period *t*. Example data obtained after the post-processing steps are provided in the Supplementary Information.

7.5.3.6.4 Machine learning model

Predictor and response variables were input into the H2O AutoML algorithm (version 3.40.0.4), which is an open source platform to automate the training and optimisation of a wide variety of supervised machine learning models (LeDell and Poirier, 2020). H2O AutoML was chosen due to its open source design, the general high performance of the platform in automated machine learning tests (Truong *et al.*, 2019) and its ability to integrate into our broader automated data processing pipeline.

Separate models were trained to independently predict the x-axis (east-west) and y-axis (north-south) distances of the transmitter from each receiver, resulting in two models per receiver group.

7.5.3.6.5 Location averaging

Location estimates produced for each receiver were an X-Y offset from the centre of the receiver. These locations estimates were converted to an easting-northing Universal Transverse Mercator grid reference by

offsetting the known location of each receiver by the estimated X-Y offset determined by the H2O AutoML model.
Where two or more receivers produced a location estimate for the same transmitter within the same time-period (t), we calculated the geographic midpoint by averaging the easting and northing estimates.

7.5.3.7 Comparative methods

We compared the results of our location fingerprinting method against two commonly used approaches to localise transmitters from directional ART data (Figure 7.5-1), which are described below. Additionally, the code for these methods is accessible in the Supplementary Information.

7.5.3.7.1 Biangulation

For each period of t and for each receiver, we identified the pair of orthogonal antennas with the highest combined RSS. Given that our receivers each had four antennas orientated towards the cardinal directions, this step effectively determined the quadrant where the transmitter was expected to be located relative to the receiver. We then calculated the AOA using the equation:

$$AOA = \frac{90}{\pi} \times \arccos(\Delta g)$$

Where Δg was the difference in RSS between the two antennas ($s_l - s_r$) normalised by the maximum signal strength difference Δm, using the equation (Gottwald *et al.*, 2019; Höchst *et al.*, 2021):

$$\Delta g = \frac{(s_l - s_r)}{\Delta m}$$

The transmitter's location was calculated by biangulating two lines generated from this approach from each pair of receivers that detected the signal (Gottwald *et al.*, 2019; Höchst *et al.*, 2021). This method therefore requires at least two receivers to estimate a position. If more than two receivers detected the signal in period t, each pair of receivers was biangulated separately and the geographic midpoint of the resulting location estimates was calculated. We excluded any location estimates falling outside of the study area, as these would be beyond the ART receivers' potential detection range.

7.5.3.7.2 Linear regression

The linear regression method followed two steps. Firstly, the AOA was calculated following the biangulation method as above. Secondly, the distance between the transmitter and receiver was estimated using a linear regression model that estimated the decline in RSS relative to the increasing distance between the transmitter and receiver. We then calculated the transmitter position by using the distance from the receiver placed along the line of the AOA, with the known receiver position serving as the reference point.

To develop the linear regression model, we calculated the Euclidean distance of each recorded location in the training dataset to the respective receivers. We then fitted a linear regression using the sklearn package version 1.4.1 (Pedregosa *et al.*, 2011) to model the relationship between the RSS and the distance to the receiver (Fisher *et al.*, 2021). Where transmitters were recorded on multiple antennas in the same period of t, we used a linear regression model that averaged the RSS-distance relationship among all antennas (Fisher *et al.*, 2021). If the transmitter was recorded on only one antenna, we used a linear regression that had been fitted only to training data for that antenna and estimated distance from the receiver using the average signal strength for that antenna in period t along the cardinal bearing of the antenna. Where two or more receivers detected the signal in period t, the geographic average of the resulting location estimates was calculated.

7.5.3.8 Model evaluation

We evaluated the performance of the three localisation methods using a test dataset that comprised 47 radio tracked SBTF locations. The error of location predictions for each method was calculated as the Euclidian distance between the predicted and the actual locations, herein referred to as 'positional error'. Four additional variables were calculated for each SBTF location, which were factors that we identified *a priori* as having potential to affect the positional error of localisations. These were the: (1) Mean distance of the transmitter to the receivers on which the signal was detected; (2) Number of receivers that detected the signal; (3) Pulse interval of the radio transmitter; and (4) Mean RSS of the detections.

We examined the effects of localisation method, mean distance to the receiver, number of receivers, mean RSS and pulse interval using a generalised linear mixed effects model with the glmmTMB package version 1.1.7 (Magnusson *et al.*, 2023). A gamma distribution with a log link function was selected to fit the response variable, positional error, which was in the form of continuous data with a positively skewed distribution. Continuous fixed factors were standardised using the sjPlot package version 2.8.15 (Lüdecke, 2023). A unique identifier of test dataset location was included as a random effect, to account for test dataset locations that spanned longer than the period of *t* (3 minutes) having multiple location estimates. All model fits and the distribution of residuals were tested using the DHARMa package version 0.4.6 (Hartig, 2022) (Supplementary Information).

We developed five candidate generalised linear mixed effects models and used an information theoretic approach to select the best performing model (Anderson and Burnham, 2004; Tredennick *et al.*, 2021). We used Akaike Information Criterion corrected for small sample size (AIC_c) to select the best performing model. The six candidate models were developed based on an *a priori* understanding of potential factors and interaction effects likely to impact positional error (Table 7.5-1). All candidate models used the unique identifier of the test dataset location as a random effect.

We estimated the impact of sample size on positional error for each of the two methods that required site-specific training data (location fingerprinting and linear regression). To do this, we iteratively performed each method using randomly chosen subsets of the training data, varying the subset size from 5 to the total number of samples (n = 185) and increasing the number of training data samples by 5 in each iteration. The positional error of these models was then calculated using the complete testing dataset. This process was repeated 10 times to account for bias in positional errors introduced by chance, which was particularly relevant to small sample sizes. All data analyses were undertaken using R statistical language version 4.3.0 (R Core Team, 2023).

7.5.4 Results

7.5.4.1 Positional error

The average positional errors were: 308 m for the location fingerprinting method (SE = 17.7, median = 230 m, Figure 7.5-5), 335 m for the linear regression method (SE = 18.5, median = 280 m, Figure 7.5-5), and 550 m for the biangulation method (SE = 42.9, median = 540 m). Both the location fingerprinting and linear regression methods were able to estimate locations for all of the 47 test locations, while the biangulation method was only able to estimate a location for 19 of the 47 test locations (40%). Of the 28 test locations that could not be estimated using the biangulation method, 23 locations (82%) failed as the transmitter was recorded on only a single receiver for each period of *t*, while 5 locations (18%) failed due to non-intersecting AOA estimates.

Positional error was best predicted by the model that included all main effects and the first order interactions between localisation method and the other fixed effects (Table 7.5-1). The next best performing model had an AIC_c that was 24.1 higher than the best model and as such we solely used the best performing model for interpretations (Anderson and Burnham, 2004).

Table 7.5-1: Candidate models evaluating the effects of localisation method, mean distance to the receiver, number of receivers, pulse interval and mean relative signal strength (RSS) on the positional error of location estimates.

Model	*K*	AIC_c	ΔAIC_c	w_i
Localisation method * (Receiver count + Pulse interval + Mean distance to receiver + Mean RSS)	17	5190.4	0.0	1
Localisation method + Receiver count + Pulse interval + Mean distance to receiver + Mean RSS	9	5214.5	24.1	0
Receiver count + Pulse interval + Mean distance to receiver +Mean RSS	7	5256.3	65.9	0
Localisation method	5	5267.6	77.2	0
Intercept only model	3	5303.0	112.6	0

Compared to the location fingerprinting method, the linear regression method achieved a similar positional error (ß = 1.08, 95% CI: 0.97–1.21, Table 7.5-2, Figure 7.5-5), while the positional error of the biangulation method was substantially greater(ß = 2.44, 95% CI: 1.82–3.28). Model accuracy was affected by all fixed factors assessed (Table 7.5-2, Figure 7.5-6). Signals detected on more receivers and signals with a higher average RSS both decreased positional error (ß = 0.89, 95% CI: 0.80–0.99; ß = 0.72, 95% CI: 0.65–0.80; respectively). In contrast, tags with a longer pulse interval and tags located further from receivers had a higher positional error (ß = 1.45, 95% CI: 1.01—2.06; ß = 1.16, 95% CI: 1.02–1.32; respectively).

While positional error increased with both increasing distance to the receivers and decreasing signal strength, the effect was less for the location fingerprinting method than the other methods. The result of this effect was that location fingerprinting achieved a lower positional error than both alternative methods where transmitters were far from the receivers (Figure 7.5-7). This is demonstrated by small but significant interaction effects between the mean distance to the receiver and the linear regression method (ß = 1.28, 95% CI: 1.13–1.44) as well as between the mean signal strength and both the linear regression and biangulation methods (ß = 1.11, 95% CI: 1.00–1.24; ß = 1.59, 95% CI: 1.12–2.272; respectively; Table 7.5-2).

Table 7.5-2: Relationships between positional error and fixed factors for the top performing model identified in Table 7.5-1. Methods were compared using a generalised linear mixed effects model with a log-link gamma distribution. Model estimates (ß and 95% confidence intervals [CIs]) have been scaled.

Predictor	Estimate (ß)	95% CI	p	Predictor
(Intercept)	256.19	223.87 – 293.17	<0.001	(Intercept)
Biangulation	2.44	1.82 – 3.28	<0.001	Biangulation
Linear regression	1.08	0.97 – 1.21	0.162	Linear regression

Predictor	Estimate (β)	95% CI	p	Predictor
Receiver count	0.89	0.80 – 0.99	0.04	Receiver count
Pulse interval (13 seconds)	1.45	1.01 – 2.06	0.041	Pulse interval (13 seconds)
Mean distance to receiver (km)	1.16	1.02 – 1.32	0.02	Mean distance to receiver (km)
Mean received signal strength	0.72	0.65 – 0.80	<0.001	Mean received signal strength
Biangulation x Receiver count	0.94	0.75 – 1.18	0.596	Biangulation x Receiver count
Linear regression x Receiver count	1.09	0.97 – 1.23	0.143	Linear regression x Receiver count
Biangulation x Pulse interval (13 seconds)	0.58	0.36 – 0.93	0.023	Biangulation x Pulse interval (13 seconds)
Linear regression x Pulse interval (13 seconds)	0.83	0.62 – 1.10	0.195	Linear regression x Pulse interval (13 seconds)
Biangulation x Mean distance to receiver (km)	1.07	0.84 – 1.35	0.598	Biangulation x Mean distance to receiver (km)
Linear regression x Mean distance to receiver (km)	1.28	1.13 – 1.44	<0.001	Linear regression x Mean distance to receiver (km)
Biangulation x Mean received signal strength	1.59	1.12 – 2.27	0.01	Biangulation x Mean received signal strength
Linear regression x Mean received signal strength	1.11	1.00 – 1.24	0.05	Linear regression x Mean received signal strength
Observations	406			Observations
R2 marginal/R2 conditional	0.408/0.614			R2 marginal/R2 conditional

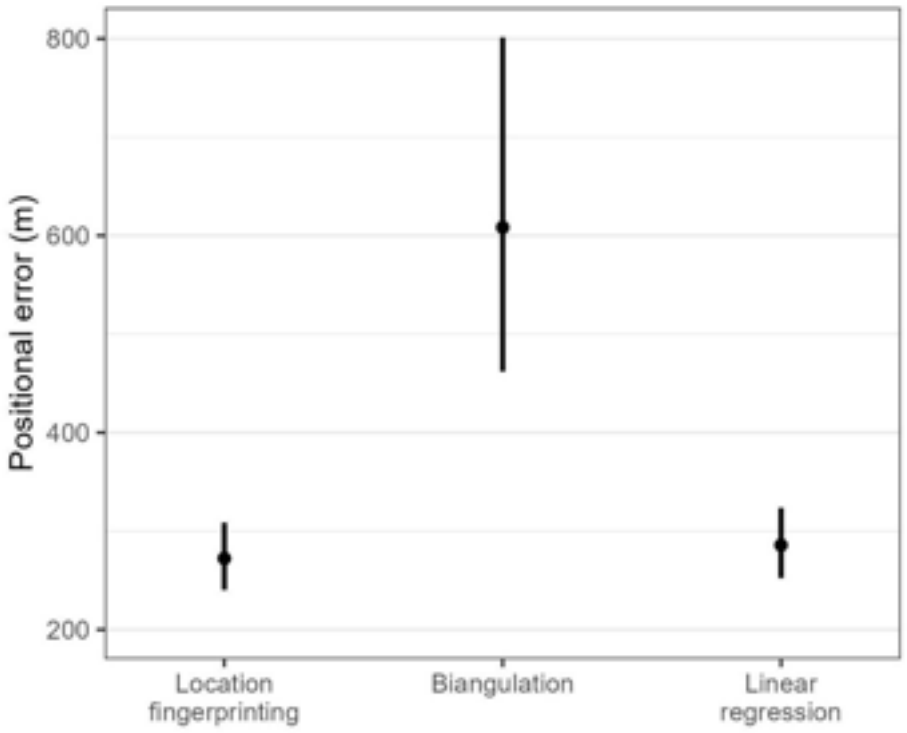


Figure 7.5-5: Comparison of the mean positional errors of each localisation method. Bars show 95% confidence intervals.

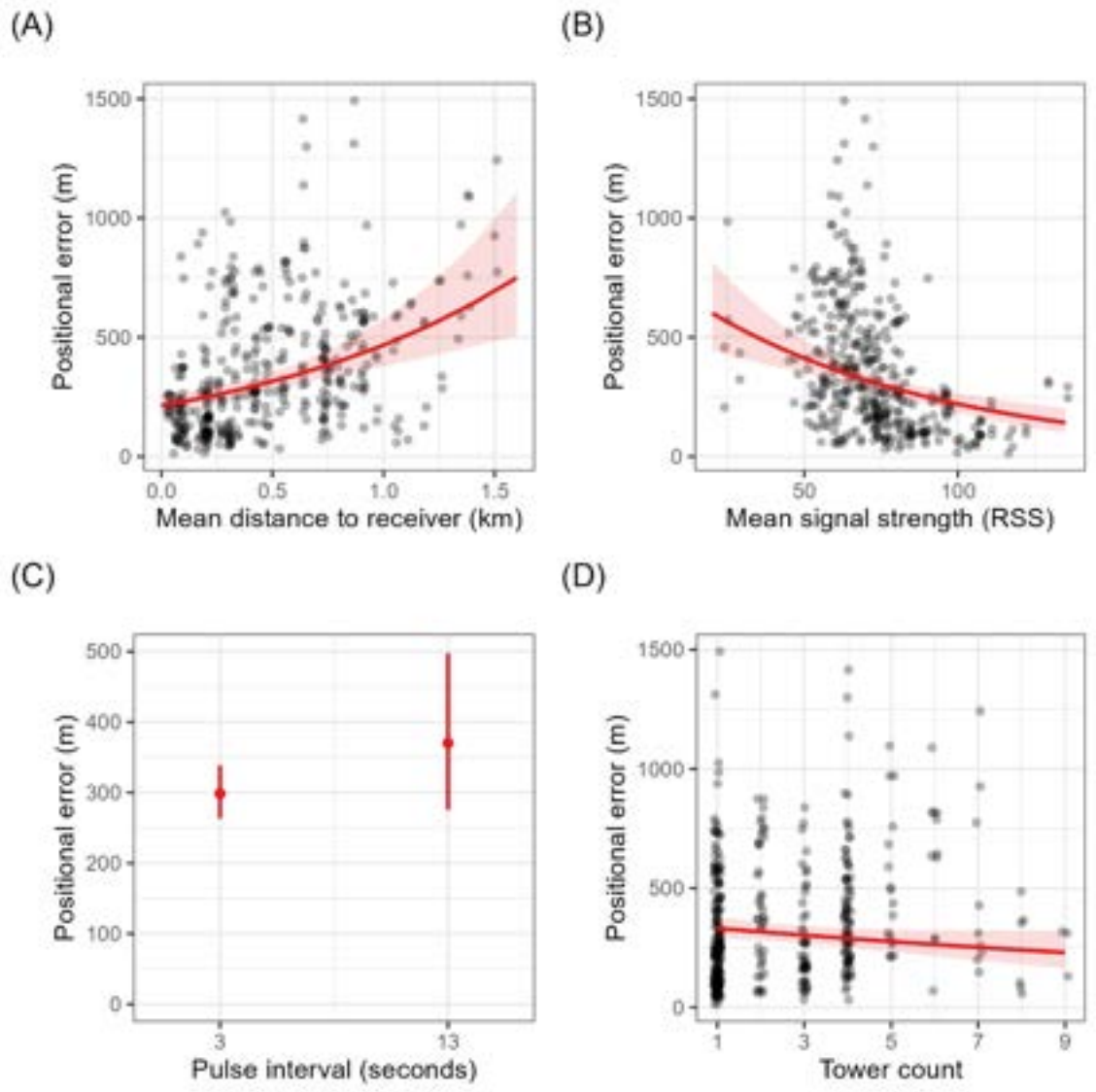


Figure 7.5-6: Effect of four covariates on the positional error of localisations. The relationships depicted include the: (A) mean distance to the receiver; (B) mean relative signal strength (RSS) of the detections; (C) pulse interval of the transmitter; and (D) number of receivers on which the signal was recorded. All figures show the mean estimate with 95% confidence intervals. A random jitter has been applied to the points of (D) to aid data visualisation.

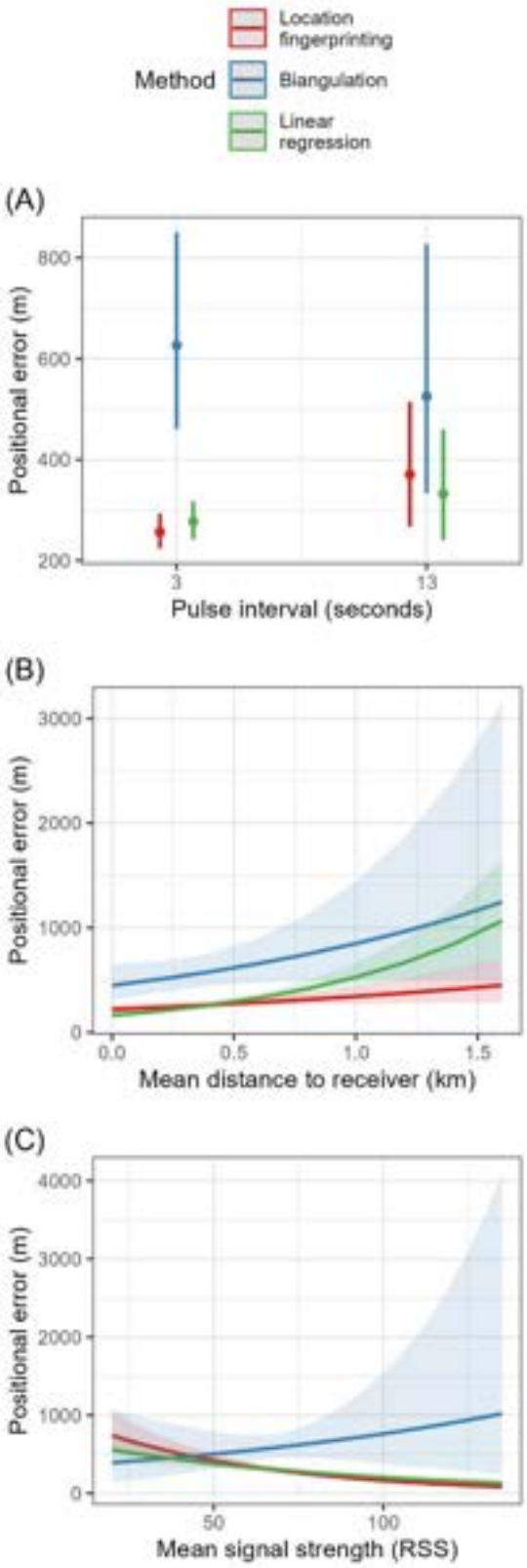


Figure 7.5-7: Interaction effects between localisation method and the: (A) pulse interval; (B) mean distance to the receiver; and (C) mean relative signal strength (RSS) of the detections. All figures show the mean estimate with 95% confidence intervals.

7.5.4.2 Sample size requirements

The relationship between positional error and training data size showed an approximately exponential decay distribution, with an initial rapid decrease in positional error with additional training samples, plateauing as more samples were included. The linear regression method required relatively few training data samples to plateau in performance, achieving a plateau after approximately 25 training locations. In contrast, the location fingerprinting method required over 100 training locations for a plateau to be reached (Figure 7.5-8).

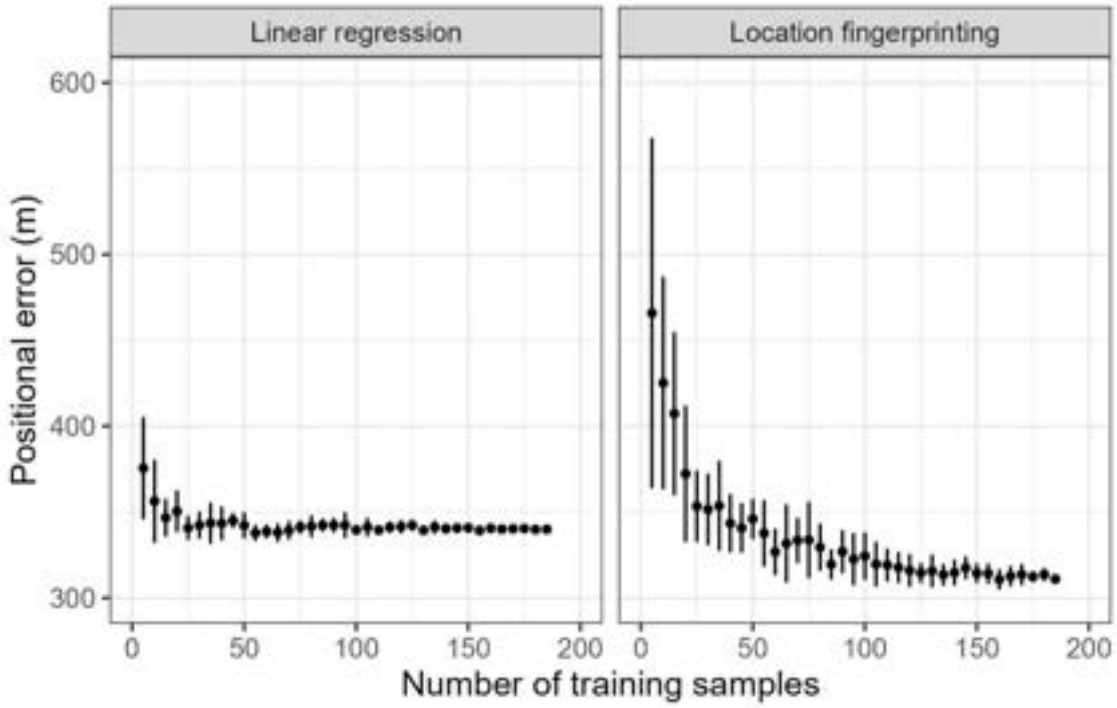


Figure 7.5-8: Positional error of localisation estimates compared to the number of samples used to train the model. Bars show the standard deviation around the mean positional error. The y-axis scale is the same for both plots and does not start at zero.

7.5.5 Discussion

We compared the performance of three localisation methods for ART systems with directional receivers. We found that the location fingerprinting and linear regression methods demonstrated comparable accuracy for estimating transmitter locations, with mean errors of 308 m and 335 m, respectively. Previous studies that have used similar methods to localise ART data have substantially differed in their ART design, which makes direct comparisons of accuracy among studies challenging. Nonetheless, previous studies have included: Harbicht et al. (2017), who achieved sub-meter accuracy using a location fingerprinting method that used six receivers to track Atlantic salmon (*Salmo salar*) along a 295 m linear waterway; Fisher et al. (2021), employed a linear regression method and achieved a median error of 72 m tracking monarch butterflies (*Danaus plexippus*) in the centre of an array of four receivers located 250 m apart, with limited vegetation or topographic variation; and Scardamaglia et al. (2022) who used a location fingerprinting method with relatively few receivers over a 500 ha study area to achieve a mean positional error of 488 m.

The biangulation method was the worst performing method in this study, with an average positional error nearly double that of the best performing methods. Angulation methods, which include biangulation, are the most widely applied localisation method (Gottwald et al., 2019; Höchst et al., 2021; Kays et al., 2011;

Lenske & Nocera, 2018; Smetzer, 2018). Accuracies achieved in previous studies have been wide ranging and include a median error of 250 m for aerial insectivorous birds (Lenske and Nocera, 2018) down to 21 m in an environmentally uniform and short-range trial (Gottwald *et al.*, 2019), albeit noting the aforementioned challenge of directly comparing accuracy estimates among studies. Regardless of the accuracy of the biangulation method, the method is limited by its inability to estimate locations from signals detected on only a single receiver or where the estimated AOAs do not intersect (Gottwald *et al.*, 2019). In our study, which had receivers that were sparsely distributed compared to previous studies, biangulation could localise only 40% of the test data locations. While using triangulation or multiangulation (more than three receivers) may improve accuracy (Gottwald *et al.*, 2019), it is likely to come at a cost of fewer localisations. For many studies, including our SBTF study, reducing the number of location estimates is a key disadvantage of the biangulation method.

The wide range of positional errors achieved in our study and previous studies (Kenward, 2004; Naef-Daenzer *et al.*, 2005; Ward, Sperry and Weatherhead, 2013; Crewe *et al.*, 2019) highlights the practical trade-offs that must be considered when designing an ART system and subsequent localisation methodology. We found that across all methods that we employed, positional error was affected by four fixed factors, all of which may be considered within ART system design. Accuracy was increased when the mean distance to the receiver was lower, signal strengths were higher and the number of receivers that simultaneously detected the signal was higher. Increasing receiver density and having transmitters located inside of an array area, instead of on the periphery, are likely to improve localisation accuracy through changes to these three factors. However, it comes at the cost of either reduced coverage or greater cost to install more receivers (Höchst *et al.*, 2021). Accuracy was also greater for tags with a shorter pulse interval, demonstrating a common trade-off in radio telemetry, which is that among the battery size, battery life, transmitter weight and radiative power of the transmitter (Kenward, 2004; Naef-Daenzer et al., 2005). Positional error is also impacted by animal mobility and behaviour, with species that move quickly, irregularly and among a range of habitats (e.g. height above ground level) producing a more inconsistent signal and result in greater localisation error (Crewe *et al.*, 2019; Fisher *et al.*, 2021; Gottwald *et al.*, 2023).

The trade-off between spatial coverage, accuracy and cost is an important consideration for researchers implementing ART systems. Griffin et al. (2020) estimated that a network of 85 ART receivers would cost approximately USD $500,000 and Birds Canada (2017) suggests a cost per receiver ranging from USD $2,200 to $7,300. Our experience indicates a similar cost of approximately USD $3,500 to $5,000 per receiver. The benefits of ART systems must also be weighed against alternative radio telemetry methods, such as manual radio tracking (Kenward, 2000) and drone-based methods (Ivosevic *et al.*, 2015; Hui *et al.*, 2021; Saunders *et al.*, 2022). Although there is no one-size-fits all solution, ART systems are likely to be most beneficial for studies requiring a high temporal resolution of data capture, as they allow for continuous radio signal detection. Due to their fixed nature, ART systems will also be most applicable to species that remain within the array's coverage area or where regular movements through the array are of research importance, such as studying migration patterns (Taylor *et al.*, 2017; Crewe *et al.*, 2019). In contrast, manual radio tracking and drone-based systems are advantaged by their flexibility of spatial coverage (Ivosevic et al., 2015; Kenward, 2000). Finally, localising animal positions using ART systems has inherent inaccuracies. While small-scale controlled trials have reported accuracies less than 100 m (Gottwald *et al.*, 2019; Fisher *et al.*, 2021; Tyson, Fragueira, *et al.*, 2024), most applied field studies, including our own, have reported accuracies in the range of 100–500 m (Lenske & Nocera, 2018; Scardamaglia et al., 2022; Smetzer et al., 2021; Ward et al., 2013). Although positional accuracy can be improved by increasing tower density, it is unlikely to achieve the sub-20 m accuracy available through manual radio-tracking with a hand-held GPS (Thomas, Holland and Minot, 2011). Therefore, ART systems may not be suitable for studies requiring highly accurate position estimates for radio-tagged wildlife.

In our study, both top performing methods required the collection of training data. While collecting site-specific training data is labour-intensive, the results of our study and the indoor localisation literature (Mendoza-Silva, Torres-Sospedra and Huerta, 2019) suggest that the most accurate localisation methods require these site specific training data. However, in our study, the location fingerprinting and linear regression methods substantially differed in their training data requirements. The linear regression method required approximately 25 training data locations to achieve maximum performance for our dataset. In contrast, the location fingerprinting method required over 100 training locations before the accuracy plateaued. The linear regression method uses site specific training data to train a simple linear regression model that relates the distance between the transmitter and receiver to the RSS. In contrast, the location fingerprinting method maps locations directly to the signal strengths from multiple antenna on each receiver (Mendoza-Silva, Torres-Sospedra and Huerta, 2019). As such, location fingerprinting requires training data from a broad area around the receiver, indicative of the spatial coverage of the receiver. For ART applications with limited training data, we therefore recommend the use of the linear regression approach; however, where sufficient training data can be collected then either the location fingerprinting or linear regression methods may achieve similar performance.

While we collected training data by manually radio tracking SBTF within the range of an ART array, this method is not feasible for all studies. Manual radio-tracking may be constrained by inaccessible terrain, limited resources and time, or species behaviour (e.g. highly vagile species) (Saunders *et al.*, 2022). An alternative to manual radio-tracking is to use an artificial simulation of the tracked animal to collect training data. Methods include holding the antenna in the air with a low-conductivity medium such as PVC tubing (Fisher *et al.*, 2021; Tyson, Fragueira, *et al.*, 2024), attaching the transmitter to a drone (Collett *et al.*, 2023), or using an artificial surrogate of the species, such as a glove filled with saline water (Naef-Daenzer *et al.*, 2005; Ward, Sperry and Weatherhead, 2013). To train either a location fingerprinting or linear regression model, the simulation method should replicate the target species' signal properties as closely as possible to maximise the model's accuracy when applied to ART data from the target species (Mohri, Rostamizadeh and Talwalkar, 2018). In a small-scale experiment on the effective radiative power from different radio transmitter attachment methods, Naef-Daenzer et al. (2005) found that the transmitter attachment technique and radiative coupling between the transmitter and the animal's body resulted in an almost three-fold difference in effective radiative power. Furthermore, Ward et al. (2013) found that postural changes in radio tagged ratsnakes (*Pantherophis* spp.) resulted in significant variation in received signal strength. The design of an artificial surrogate of the species must therefore be considerate of the species behaviour, transmitter orientation and attachment, body mass and movement patterns (Ward, Sperry and Weatherhead, 2013; Fisher *et al.*, 2021).

The location fingerprinting method, though more data-intensive, offers greater versatility across different ART designs with directional receivers. It accommodates receivers with any number of antennas at any combination of orientation, two elements of ART design that would currently require custom code development to implement a linear regression method. An alternative location fingerprinting approach, which was employed by Tyson et al. (2024) for omni-directional receivers and Harbicht et al. (2017) for an aquatic environment, is to create a whole-of-array radio fingerprinting map, rather than the receiver-centric approach used in this study. The indoor positioning literature suggests that these whole-of-array approaches would provide the highest level of accuracy (Mendoza-Silva, Torres-Sospedra and Huerta, 2019); however, collecting training data for such whole-of-array models would be challenging, potentially prohibitive, in many large scale ART arrays. Nonetheless, should such training data be available, a whole-of-array location fingerprinting method would likely outperform the receiver-centric method we have applied (Mendoza-Silva, Torres-Sospedra and Huerta, 2019). Further research could improve our location fingerprinting method by incorporating time information associated with the RSS data and testing alternative approaches for location averaging to increase the weighting of receivers closer to the SBTF that record a higher RSS (Jia *et al.*, 2022). Additionally, future extensions may integrate activity classification into the location fingerprinting method. Gottwald et al. (2023) and Schofield et al. (2018)

demonstrated that machine learning methods can classify activity states of microchiropteran bats and songbirds by extracting features such as variation in received signal strength from ART data. Leveraging these techniques could improve insights into animal behaviour that are gained from ART systems.

The difficulties of comparing localisation performance among methods and studies underscores the importance of like-for-like methodological comparisons. While literature that compare localisation methods for omni-directional antennas have progressed in recent years (Paxton *et al.*, 2022; Tyson, Fragueira, *et al.*, 2024), our study is the first methodological comparison for ART systems with directional receivers. There remain methodological variations not included in this study, and likely future developments, that will benefit from having their performance benchmarked to alternative methods. To this end, we have made our training and testing datasets and methods open access.

7.5.6 Conclusions

In this study, we introduced a novel machine learning based localisation method for ART systems that use directional receivers and compared it to two alternative approaches. We found that the location fingerprinting method provided a highly versatile approach to localisation for these systems that achieved comparable accuracy to the best performing alternative approach that we tested. The location fingerprinting method can be applied to a wide variety of ART system designs; however, it requires the collection of site-specific training data, an important consideration in designing ART systems using this approach. Additionally, our findings provide insights into the practical trade-offs in ART system design, especially among localisation accuracy, receiver density and cost.

8 A review of land management strategies to support habitat quality for the southern black-throated finch

8.1 Abstract

Land use changes have significantly reduced the availability and quality of resources in woodland habitats, severely impacting SBTF, which has experienced a dramatic range decline and is now largely confined to the Townsville Plains and Desert Uplands regions of Queensland. This study investigates the critical habitat features essential for the conservation of the SBTF, with an emphasis on tailoring land management practices to enhance these features. We identify key ecological requirements that support SBTF populations, such as a species-rich open ground layer, low shrub cover, minimal exotic species, reliable access to water, and large, interconnected habitat areas. Furthermore, we explore how land management strategies, including low to no grazing pressure, low-intensity patch burning, and consistent access to water, can be utilised to preserve and improve these habitat characteristics. Our findings suggest that low intensity grazing regimes generally promote the growth of key forage species and increase seed production. Furthermore, regular low-intensity burning is essential for maintaining appropriate habitat structure, particularly when grazing pressure is removed. Our research and the broader literature highlight the importance of maintaining a good condition ground layer and appropriate vegetation structure to promote long-term conservation and recovery of SBTF populations.

8.2 Introduction

Savannas worldwide have been extensively cleared and degraded (Purvis *et al.*, 2019), with over 80% transformed by human activities by 2000, including 52% converted to rangelands and 25.8% to croplands (Ellis *et al.*, 2010).The savannas of northern Australia, covering 1.9 million km^2 (Russell-Smith and Yates, 2007), remain the largest and least impacted globally, with less than 1% cleared (Murphy et al., 2016). However, increasing grazing pressure, altered fire regimes, woody vegetation thickening, and the spread of exotic species are major threats, leading to significant habitat degradation (O'Connor and Pickett, 1992; Franklin *et al.*, 2005; Kutt and Woinarski, 2007; Legge *et al.*, 2019). These land use changes have reduced the availability and quality of resources in woodland habitats, contributing to the decline of granivorous birds, which are overrepresented among Australia's threatened bird species (Franklin, 1999; Franklin, Woinarski and Noske, 2000; Franklin *et al.*, 2005; Garnett and Baker, 2020). Despite relatively intact vegetation and low-intensity pastoralism in northern Australia, around 25% of granivorous bird species are declining in abundance (Franklin, 1999; Franklin *et al.*, 2005; Woinarski *et al.*, 2011; Edwards, Russell-Smith and Meyer, 2015).

This habitat degradation has severely impacted the southern black-throated finch (SBTF), an endangered estrildid finch that has suffered extensive range decline since European settlement and is now largely confined to the Townsville Plains and Desert Uplands regions in Queensland (Franklin *et al.*, 2005; Vanderduys *et al.*, 2016b; Mula Laguna *et al.*, 2019). As an inhabitant of open grassy woodlands, SBTF primarily feed on the seeds of both native and introduced grass species (Higgins, Peter and Cowling, 2006; Mula Laguna *et al.*, 2019; Williams *et al.*, 2020). Food availability significantly impacts population dynamics (Sutherland, 2004; Peacock, Macleod, Matt Davies, *et al.*, 2024) and wildlife foraging behaviour is shaped by competition, predation pressures, and habitat quality (Wiens, 1986; Southwood, 1988; Block and Brennan, 1993; Kramer, 2001; Abdulwahab, Osinubi and Abalaka, 2019). Due to this, habitat selection occurs at multiple spatial scales, ranging from broader landscape features to site-specific vegetation

characteristics, with habitat heterogeneity providing diverse resources essential for breeding and survival (Pickett and Siriwardena, 2011; Price *et al.*, 2013; Anderle *et al.*, 2023). Effective conservation strategies must account for these multi-scale habitat requirements, including food, nesting, and shelter needs, to support SBTF's survival and breeding success.

This study examines the critical habitat features utilised by the SBTF and explores how land management practices can be tailored to enhance these features. Specifically, we identify the key ecological requirements that support SBTF populations, including key foraging and nesting requirements, access to water, and interconnected habitat. Additionally, we provide recommendations for land management strategies aimed at preserving and improving these habitat characteristics, addressing threats such as grazing pressure, fire and availability of water. By aligning land management practices with the species' habitat needs, we aim to foster the long-term conservation and recovery of SBTF populations.

8.3 Important habitat requirements for the southern black-throated finch

8.3.1 Grass species rich open ground layer

The survival, reproduction, and overall productivity of bird populations relies heavily on the quality and availability of foraging resources (Martin, 1987; Briga *et al.*, 2017; Mishra and Kumar, 2019). Section 6.1 discusses the influence of land management strategies on ground layer species richness; however, in addition to the quantity of native species present, quality of forage species is also a key aspect of SBTF habitat quality (Section 4.2 and 5.1). To optimise energy intake, granivores often choose more profitable grass species with seeds that have high caloric content or are easiest to process (Díaz, 1996; Luis Marone *et al.*, 2022). When these species are sparse, individuals must manage a reduced nutritional intake as well as increased travel or processing time, which in turn leads to elevated risk of predation (Benkman, 1991; King and Marshall, 2022). Within the Project Area, a few key grass species have a high Jacob's Index which indicates they are heavily selected over others, likely due to their high nutritional value and low processing time (Section 5.1). These species include: *Digitaria* spp., *Panicum* spp., *Urochloa* spp., *Paspalidium rarum, Schizachyrium fragile* and *Enneapogon virens*. Although some of the species identified as preferred forage species in Section 5.1 are exotic species, land management should avoid promoting the spread of these species as SBTF has been found to select habitat with low exotic species cover (Section 4.1 and 4.2).

However, it is also important to consider that SBTF diversify their diet in periods of low seed availability (van Osta et al., 2024; Section 5.1). In the tropical savannas of northern Australia, the majority of seeds are produced during the wet season (Crowley and Garnett, 1999a; Franklin *et al.*, 2005), during which SBTF can preferentially feed on high value species. However, seed availability declines through the dry season and may be lowest as summer rains arrive and trigger seed germination, further reducing seed availability in the soil seed bank (Crowley and Garnett, 1999; Dostine and Franklin, 2002). Due to this variation in seed availability, high quality foraging habitat for SBTF also requires a diverse grass layer with temporal variation in seed production to provide a continuous supply of seeds. Late-seeding perennials, such as *Heteropogon contortus* and *Aristida* spp., extend seed availability into the dry season (Section 5.2). Early seeding perennials, such as *Alloteropsis semialata*, *Chrysopogon fallax* and *Themeda triandra*, provide a crucial resource in the early wet season when seed availability is lowest (Section 5.2).

In addition to the seed itself, access to fallen seed is also an important habitat requirement. SBTF require a patchy mosaic of bare ground between grass tussocks in order to forage seeds from the ground and access seed heads (Buosi, 2011b, Section 4.2).

8.3.2 Low shrub cover

Low shrub cover (<2 m) is a characteristic of both nesting and foraging habitat for SBTF (Section 4.1 and 4.2). There may be multiple reasons for the preference that SBTF have for open grassy woodlands with a low shrub cover. The cover of native perennial grasses has a negative relationship with woody vegetation cover (Daly and Hodgkinson, 1996; Brown and Carter, 1998; Fensham, Fairfax and Archer, 2005), and as such, higher shrub cover results in less food available for SBTF. Uncharacteristically among the sympatric finches of the Study Area (double-barred finch, zebra finch and plum-headed finch), SBTF prefer to nest in trees, above 2 m, albeit with a small proportion of exceptions in shrubs such as *Melaleuca nervosa* and *Bursaria incana*. SBTF also seek cover in canopy trees when disturbed, which differs from double-barred finch, which prefer to seek cover in the shrub layer. We therefore posit that SBTF are better adapted to environments with a low shrub layer than sympatric finch species, with which they compete for food resources.

8.3.3 Low exotic species cover

Exotic pasture species have been introduced all over the world for their competitive vigour and resilience in grazing systems (Gardener, 1993; Marshall, Lewis and Ostendorf, 2012), which also make them a key threat to biodiversity in Australia (Eyre *et al.*, 2009; Read *et al.*, 2020). Higher cover of the two dominant exotic species recorded in the Project Area, buffel grass (*Cenchrus ciliaris*) and stylo (*Stylosanthes scabra*), has been found to negatively influence the selection of foraging habitat by SBTF (Section 4.2). This is likely due to the displacement of native grasses and reduction in overall plant diversity that result from exotic pasture invasions (Wright *et al.*, 2020). In addition to competitive effects, invasion by buffel grass can significantly increase fuel loads and therefore increase potential fire intensity and frequency, which in turn may impact negatively on multiple habitat requirements of SBTF, including shrub and canopy cover (Miller *et al.*, 2010; Schlesinger, White and Muldoon, 2013; Ryan-Colton *et al.*, 2024). Although effects of stylo on biodiversity are less known than buffel grass, the species is highly competitive and able to rapidly colonise uninfested areas (Gardener, 1993; Orr, 2010; Kemp and Kutt, 2020).

8.3.4 Reliable access to water

Like many other granivorous birds, SBTF must drink water on a regular basis (NRA, 2007; van Osta *et al.*, 2024b). As such, SBTF show a preference for foraging and nesting habitats closer to permanent water sources to reduce the energy invested in travelling between foraging patches and water sources (Section 4.1 and 4.2). Records of SBTF at newly established water sources in areas of the Project Area without previous records may indicate that the introduction of new water sources can potentially increase the area of occupancy for the species; however, further studies need to be carried out to confirm this (Section 6.2). However, the introduction of new water sources can impact the surrounding environment due to overgrazing from stock that congregate around water points (James, Landsberg and Morton, 1999; Fensham and Fairfax, 2008). The formation of these grazing piospheres can lead to the degradation of foraging habitat surrounding these water sources (James, Landsberg and Morton, 1999; Fensham and Fairfax, 2008), causing SBTF to utilise more energy while travelling to drink. Piospheres can be avoided by designing water sources that exclude cattle. Our research found that SBTF are robust to water source design, being able to use a wide range of water source designs, provided they can reach the water's edge while stationary (Section 6.2). However, preferences have been shown for water sources that allow for protection from predators such as those with surrounding canopy cover (Section 6.2 and 6.3).

8.3.5 Large areas of interconnected habitat

Habitat clearing is the primary cause of SBTF decline to date (Black-throated Finch Recovery Team, 2004; Reside *et al.*, 2019), with SBTF strongly avoiding non-remnant vegetation (van Osta *et al.*, 2024b). Canopy trees, which cleared areas lack, are an important feature of SBTF habitat as they provide habitat for roosting and nesting as well as protection from predators (Zann, 1972). Due to this avoidance, clearing not only reduces the area of available habitat but can also fragment populations of SBTF. Genetic studies of the SBTF population on Moray Downs revealed that what was previously considered one population were two genetically separate subpopulations that have been isolated by areas of unsuitable habitat north of the Carmichael River (Section 3.2). Fragmentation of populations can lead to reduced gene flow and a loss of genetic diversity that is vital for supporting the long-term persistence of species (Allendorf and Luikart, 2009; Lowe and Allendorf, 2010).

8.4 Habitat management

Land management within the Project Area should aim to promote the habitat requirements of SBTF (Section 8.3). Key land management 'levers' that can influence habitat quality at large scales include grazing pressure, fire regimes and water sources. Studies have been carried out as part of the BTFRP to determine how to effectively manage SBTF habitat to promote the habitat values discussed above (Section 6.1 and Section 6.2). A summary of these findings, as well as a review of relevant literature, is presented in Sections 8.4.1 to 8.4.3 to provide a guide to SBTF habitat management. This guide has also been summarised into a conceptual model of the interaction among habitat management factors and habitat attributes for SBTF is shown in Figure 8.4-1.

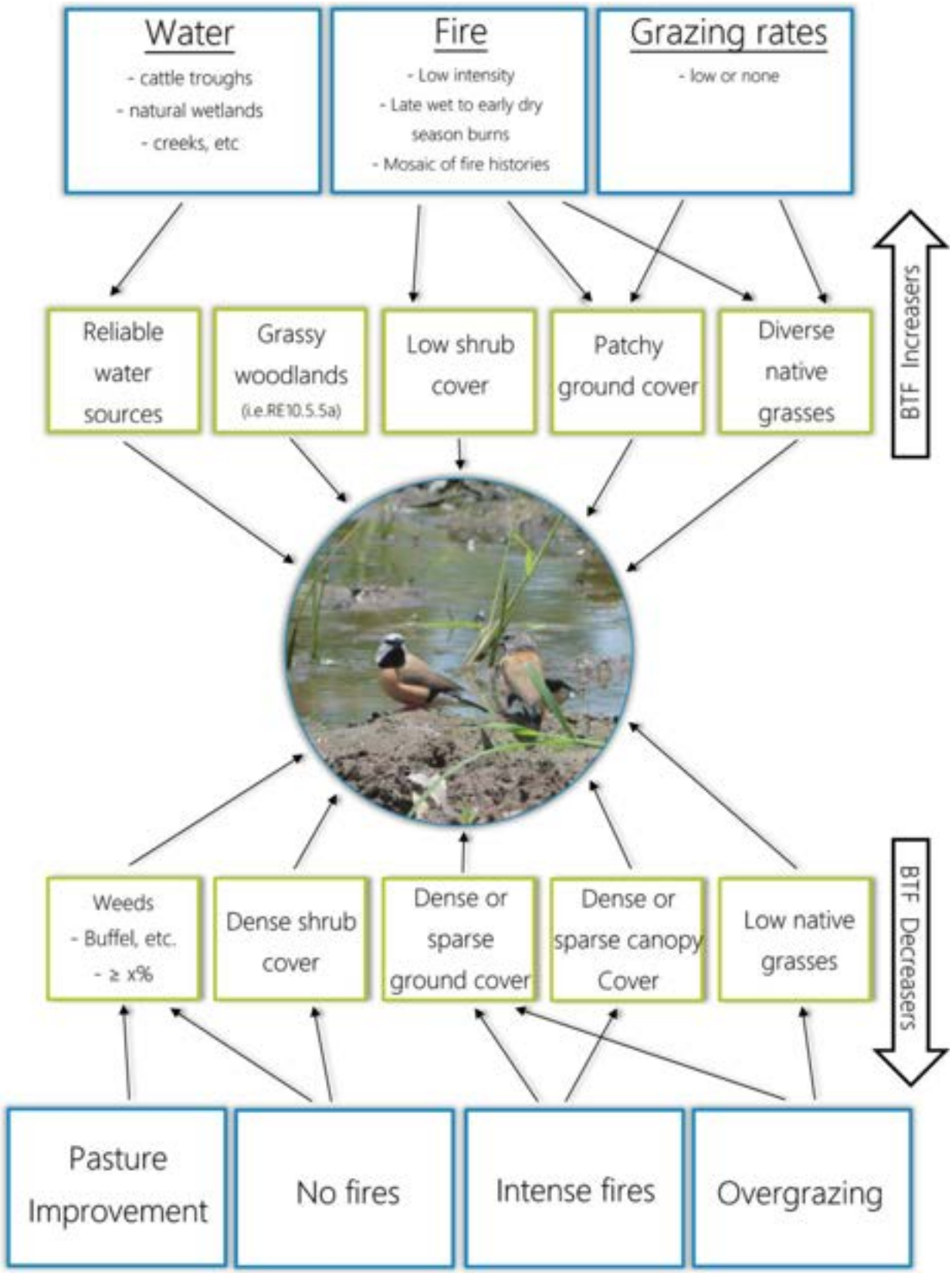


Figure 8.4-1: Conceptual model of SBTF response to habitat management factors (blue) and habitat attributes (green)

8.4.1 Low to no grazing pressure promotes preferred grass foraging species

The Australian savanna, in which the Project Area is situated, is a landscape adapted to disturbance by fire, whereas grazing by large, hooved mammals has been recently introduced to the landscape (Lunt *et al.*, 2007). Due to this, grazing at even low intensities has negative effects on ecosystem structure, function and composition (Kutt and Woinarski, 2007; Eldridge *et al.*, 2016; Lebbink, Dwyer and Fensham, 2021). Section 6.1 discusses the effects of grazing and fire on ground layer species richness and cover more generally; however, land management in the Project Area should aim to promote key forage species specifically. Although native grass species are generally not adapted to grazing, species can have varying responses to disturbance based on factors such as morphology, herbivore selectivity and life history (Seymour *et al.*, 2010; Van Coller *et al.*, 2018; Moore, Camac and Morgan, 2019). To broadly describe the species responses to grazing pressure, the term 'increaser' and 'decreaser' species have been used to describe species that either increase or decrease with grazing pressure (Vesk and Westoby, 2001).

Key forage species for SBTF generally respond negatively to increased grazing pressure (Table 8.4-1). Of the 15 species assessed, seven were found to be decreasers and two were found to be increasers. The remaining six species were found to have inconclusive results due to either lack of data or variable responses between studies. Species responses have been grouped broadly for this review; however, grazing impacts can vary depending on grazing pressure and external influences such as rainfall, which may lead to variation between studies (Mcintyre, Heard and Martin, 2003; Orr and O'Reagain, 2011).

The high number of grazing decreasers is expected as fire-adapted species are often not grazing-resistant, as fire-resistant traits such as flammability and vertical growth differ from the lateral growth and below-ground protection typical of grazing adapted species (Archibald, Hempson and Lehmann, 2019). This is highlighted by the leafy morphology of *Panicum decompositum* and *Digitaria ammophila*, which is highly palatable to cattle and therefore selectively grazed (Bohning and Wilkie, 1999; McIntyre and Lavorel, 2001; Vesk and Westoby, 2001), resulting in the species being a strong decreasers (Table 8.4-1). Grazing also negatively effects fire-adapted species that require fire or conditions created by fire such as heat, smoke or edaphic conditions to germinate (Lunt *et al.*, 2007; Lewandrowski *et al.*, 2018; Pausas and Lamont, 2022). The reduction in biomass caused by grazing excludes fire from the landscape, likely causing species with fire-adapted germination to decrease in response to grazing, such as *Themeda triandra* and *Panicum spp.* (Read and Bellairs, 1999; Hodges *et al.*, 2021).

Two of the 15 species were found to increase in response to grazing (Table 8.4-1). One which is the native species *Enneapogon virens* which is highly resistant to disturbances, including grazing (Kutt and Woinarski, 2007; Hall *et al.*, 2017). The remaining increaser species is *Digitaria ciliaris*, an invasive species that is common in disturbed areas (Jones, Contreras and Everman, 2021). Although this species is a preferred SBTF forage species, the presence of both grazing and exotic, grazing-adapted species within the landscape will likely shift species composition toward a lower diversity ground layer of more grazing tolerant species (Archibald, Hempson and Lehmann, 2019).

Beyond the effects of grazing directly to species cover, higher grazing pressures also reduce the seed production of grasses through grazing of inflorescences and defoliation, reducing plant yield (O'Connor and Pickett, 1992; Ash *et al.*, 1997; Crowley and Garnett, 2001). This alteration to grass seed production has significantly impacted granivorous bird populations in the Australian savanna (Franklin, 1999; Franklin *et al.*, 2005).

Table 8.4-1: Summary of key forage species for SBTF found within the Project Area and their response to grazing

Species	Response to grazing	References
Alloteropsis semialata	Decreaser	Crowley and Garnett (2001)
Chrysopogon fallax	Inconclusive	Section 6.1; McIntyre and Lavorel (2001); Vesk and Westoby (2001); Orr and O'Reagain (2011); Hall *et al.* (2017)
*Digitaria ammophila**	Decreaser	McIntyre and Lavorel (2001); Vesk and Westoby (2001)
*Digitaria brownii**	Inconclusive	Section 6.1; McIntyre and Lavorel (2001); Vesk and Westoby (2001); Mcintyre, Heard and Martin (2003); Fensham, Silcock and Dwyer (2011)
*Digitaria ciliaris**	Increaser	Fensham and Skull (1999); Vesk and Westoby (2001)
*Enneapogon virens**	Increaser	Section 6.1; Kutt and Woinarski (2007); Hall *et al.* (2017)
Heteropogon contortus	Inconclusive	Section 6.1; McIntyre and Lavorel (2001); Vesk and Westoby, (2001); Mcintyre, Heard and Martin (2003); Hall and Douglas (2005); Hall *et al.* (2017)
*Panicum decompositum**	Decreaser	Vesk and Westoby (2001); Mcintyre, Heard and Martin (2003)
*Panicum effusum**	Inconclusive	Section 6.1; Fensham and Skull (1999); Vesk and Westoby (2001); Mcintyre, Heard and Martin (2003); Kutt and Woinarski (2007)
*Paspalidium rarum**	Decreaser	Fensham, Fairfax and Dwyer (2010)
*Schizachyrium fragile**	Decreaser	Section 6.1; Kemp and Kutt (2020)
Sehima nervosum	Decreaser	Section 6.1; Fensham and Skull (1999); McCosker, Rolfe and Fensham (2009)
Themeda triandra	Decreaser	Section 6.1; Vesk and Westoby (2001); Mcintyre, Heard and Martin (2003); Fensham, Silcock and Dwyer (2011); Hall *et al.* (2017)
*Urochloa piligera**	Inconclusive	
*Urochloa holosericea**	Inconclusive	

*Preferred SBTF forage species (Section 5.1)

8.4.2 Low-intensity patch burning in the late wet and early dry seasons

Fire plays an important role in Australian savannas, primarily through changes in species and structural composition that heavily influence the availability of resources and suitability of SBTF habitat (Butler and Fairfax, 2008; Woinarski and Legge, 2013). Although fire has been shown to promote native species diversity within the Project Area (Section 6.1), the response of key forage species to fire were generally less significant than the response to grazing. In addition, the lack of quantitative information of the effects of fire on savanna ecosystems has been identified as a key gap by previous research (Woinarski and Legge, 2013; Fensham, Wang and Kilgour, 2015; Lebbink, Dwyer and Fensham, 2021; Radford *et al.*, 2024). Due to this, despite a similar review being carried out for fire as was for grazing, all key forage species

but four were found to have inconclusive responses to fire, due to either insignificant results, variation between results or lack of research all together. However, of the remaining four species, all four were found to increase in response to fire (Table 8.4-2).

Table 8.4-2: Summary of key forage species for SBTF found within the Project Area and their response to fire

Species	Response to fire	References
Alloteropsis semialata	Increaser	Bateman and Johnson, (2011)
Chrysopogon fallax	Increaser	Section 6.1; Hall and Douglas (2005); Silcock *et al.* (2018)
*Digitaria ammophila**	Inconclusive	
*Digitaria brownii**	Inconclusive	Section 6.1
*Digitaria ciliaris**	Inconclusive	
*Enneapogon virens**	Inconclusive	Section 6.1; Cowley *et al.* (2014); Silcock *et al.* (2018)
Heteropogon contortus	Increaser	Section 6.1; Orr, McKeon and Day (1991); Hall and Douglas (2005); Silcock *et al.* (2018)
*Panicum decompositum**	Inconclusive	
*Panicum effusum**	Inconclusive	Section 6.1; Fensham, Wang and Kilgour (2015); Silcock *et al.* (2018)
*Paspalidium rarum**	Inconclusive	
*Schizachyrium fragile**	Inconclusive	Section 6.1; Crowley and Garnett (1999)
Sehima nervosum	Inconclusive	Section 6.1
Themeda triandra	Increaser	Section 6.1; Morgan and Lunt (1999); Silcock *et al.* (2018)
*Urochloa piligera**	Inconclusive	
*Urochloa holosericea**	Inconclusive	

*Preferred SBTF forage species (Section 5.1)

However, beyond the direct effects of fire on species cover, fire is a vital tool to maintain SBTF habitat values and structure in an Australian savanna landscape. In addition to promoting species diversity, fine-scale burns promote phenological diversity which improves grass seed availability throughout the year and helps to reduce resource bottlenecks (Section 5.2). This is particularly relevant for annual species that are an important food source for granivores due to the high abundance of seeds they produce (Crowley and Garnett, 1999a; Franklin *et al.*, 2005) and often complete their lifecycle in response to disturbance (Friedman, 2020), including land management interventions such as fire (O'Reagain *et al.*, 2018). Recently burnt areas are also preferentially foraged by avian granivores which can lead to seed-bank depletion; however, patchy burns will allow SBTF access to seed while maintaining the grass population across the broader landscape (Crowley and Garnett, 1999).

Fire is often excluded in pastoral areas, either actively or as a result of biomass removal from livestock grazing (Russell-Smith, Start and Woinarski, 2001). This negatively affects SBTF habitat as regular, low-intensity fires disturb dense ground layers dominated by high biomass native perennial grasses, which

allows short-lived perennial and annual species access to resources such as space and light (Russell-Smith, Start and Woinarski, 2001; Lebbink, Fensham and Cowley, 2018). Within the *Eucalyptus melanophloia* woodlands that are the preferred habitat of SBTF within the study area, *Triodia* tends to dominate sections of sandier soils that are likely less fertile than the loamy soils where tussock grasses dominate. However, our data (Section 6.1) suggests that low-intensity burning of *Triodia* can increase the richness of native tussock grass species and promote forage availability for SBTF.

Along with a dense ground layer, fire exclusion also often results in the thickening of woody shrubs, which suppress the grassy understorey and therefore reduce food availability (Liedloff *et al.*, 2001; Williams, Woinarski and Andersen, 2003). *Carissa* species, such as *Carissa lanceolata* within the Project Area, are dense, ground-hugging shrubs that spread across the ground layer in the absence of fire and exclude grass species (Back, 2005). Low-intensity fire can effectively manage shrub encroachment, with low intensity fires preferentially burning shrubs that are less than 2 m, which having minimal impact on the tree layer that is required by SBTF for nesting and cover (Section 4.1; Back, 2005). Although fire will not kill individual *Carissa* plants, it can substantially reduce cover, with *Carissa* taking between two and five years to regrow back to its pre-burning extent following fire (Back, 2005; O'Reagain *et al.*, 2018). Low intensity fires, where scorch heights are kept low (<2 m), may also give SBTF a competitive advantage over other granivorous species that nest closer to the ground, such as double-barred finches and diamond doves.

Late wet season and early dry season burns also provide an opportunity to manage exotic species over large-scale areas where labour-intensive methods, such as herbicide application and manual removal, are impractical. Within the Project Area, non-native graminoids, primarily buffel grass, were found to decrease under a regime of low or no grazing pressure and patchy, low intensity burns (Section 6.1). In cases where exotic species are present in low density, such as within the Project Area, these regimes can promote resilience to exotic species invasion by restoring habitat to conditions favourable to native species rather than buffel grass (Ash *et al.*, 1997; Jackson, 2004; Alba *et al.*, 2015).

It is also important to consider the role that regular, low intensity burns play in maintaining foraging habitat once grazing is reduced or removed. Reductions in grazing pressure can lead to substantial increases in accumulated biomass, which in turn leads to an increase in the risk of uncontrolled bushfire, higher fire intensities and larger patch sizes (Ash *et al.*, 1997; Liedloff *et al.*, 2001; Hunt *et al.*, 2014). Without regular low intensity fires to manage biomass levels, high intensity fires can occur, which can impact ground layer species through the destruction of seed banks, increase in non-native cover and thickening of the woody shrubs (Williams *et al.*, 1999; Liedloff *et al.*, 2001; Yates, Edwards and Russell-Smith, 2008; Miller *et al.*, 2010; Wright and Fensham, 2016). The use of low intensity burns to manage biomass may have indirect benefits to maintaining SBTF forage species diversity that outweigh the direct effects identified in Table 8.4-2.

8.4.3 Reliable access to water

The availability of permanent water sources is a key habitat feature for SBTF and with the spread of pastoralism, artificial water sources are now widespread and relatively easy to install. Based on the balance of evidence, water sources should be spaced approximately 4 km apart (van Osta *et al.*, 2024, Section 6.2). However, the implementation of additional water sources can have unintended deleterious effects, primarily increased grazing pressure (James, Landsberg and Morton, 1999; Fensham and Fairfax, 2008). To mitigate this, the addition of new water points should exclude grazers to prevent the associated deleterious effects on the surrounding habitat. In addition it is also important to ensure that water source design is suited to use by SBTF (Section 6.3), including:

- A suitable perch for the birds to grasp with their feet

- A narrow (<3 cm thick) edge and a distance from the edge of the structure to the water level of less than 5cm
- Located in close proximity <10 m to surrounding woody vegetation (trees or shrubs) to provide cover and reduce predation risk as birds fly to and from the water; and
- The reliability of the water source to provide water year-round, but particularly in the dry season.

8.5 Conclusion

Based on the balance of evidence provided by our research and the broader scientific literature, low intensity grazing regimes generally promote growth of key SBTF forage species as well as seed production of grasses more generally. Although the direct effects of fire on key SBTF forage species are inconclusive, regular low intensity burns are important for managing SBTF habitat structure, particularly once grazing is removed from the landscape. Providing access to permanent water sources is also an important habitat consideration for SBTF, provided new water sources do not result in an increase in grazing pressure.

9 Recommendations

9.1 General habitat management recommendations for the southern black-throated finch

There are three pillars of habitat management through which the majority of SBTF habitat quality maintenance and improvement can be achieved:

- Low to no stocking rates (Section 9.1.1)
- Implement a burning program of low-intensity patchy burns (Section 9.1.1); and
- Provide access to reliable water sources (Section 9.1.2).

These habitat management factors promote habitat attributes preferred by SBTF, namely:

- Accessible water sources
- Open grassy woodlands, with a tussock grass understorey
- A rich diversity of native grass species
- Low shrub cover; and
- Patches within the ground cover that promote foraging efficiency.

Additional interventions may directly target SBTF habitat attributes where fire, grazing and water source regimes are unable to promote SBTF habitat suitability. The primary reason for additional management intervention is in the case of highly degraded areas, which require rehabilitation due to:

- Land clearing
- Dense infestations of exotic plants (e.g. buffel grass or stylo); and
- Heavily modified ground layers that contain few native grass species.

Management of heavily modified habitats are discussed in Section 9.1.3.

9.1.1 Fire and grazing management

Fire and grazing are key habitat modifiers affecting the SBTF by altering ground cover composition and seed availability (Section 6.1). The research findings and literature suggest that optimal management of the Project Area for SBTF conservation should focus on:

- Excluding domestic livestock where possible, AND
- Implementing low-intensity, patchy burns (Section 6.1).

The goal is to restore fire and grazing regimes similar to those prior to European colonisation, characterised by frequent, small-scale traditional burns that created a mosaic of fire histories and prevented large-scale wildfires. A program of low-intensity fire combined with minimal or no livestock grazing is expected to benefit SBTF conservation through:

- **Promoting SBTF forage species**: Low-intensity fires enhance the diversity and cover of preferred forage species (Section 6.1) and minimise nutrient loss compared to high-intensity fires (Agbeshie *et al.*, 2022).

- **Reducing grazing impacts**: Grazing reduces seed production by native grasses (O'Connor and Pickett, 1992; Ash *et al.*, 1997). In contrast, low-intensity fire promotes native grass diversity (Section 6.1) and may increase the nutritional quality of grass seeds (Weier, Radford, *et al.*, 2017).
- **Minimising exotic species encroachment**: Low grazing pressure combined with low-intensity fire affords relatively intact ground layers, such as those within the Project Area, the most resilience to minimise the encroachment of exotic species such as buffel grass and stylo (Section 6.1).
- **Promoting the nest niche**: SBTF prefer nesting in trees taller than 2 m (Section 4.1). SBTF may achieve a competitive advantage over other granivorous species that nest closer to the ground, such as double-barred finches and diamond doves, in a landscape with regular low-intensity fires. These low intensity fires will not destroy the majority of SBTF nests, provided scorch heights are kept low (<2 m).
- **Controlling woody shrub encroachment**: Fire helps prevent woody shrub encroachment over time, promoting the open grassy woodlands preferred by SBTF for foraging and nesting (Section 6.1;Cowley *et al.*, 2014).
- **Optimising foraging habitat**: Low intensity burns create open ground cover without destroying grass seeds that fall to the soil surface, providing an optimal foraging habitat. SBTF have been observed foraging in recently burnt areas as early as one-month post-burn, including where unburnt vegetation is available nearby.

The Queensland Parks and Wildfire Planned Burn Guidelines (Melzer and Hines, 2022), outline practical measure to achieve a low-intensity, patchy burn. Key factors to consider include:

- **Timing:** Late wet season or early dry season burns when soil is moist, and fuel has not yet fully cured will be lower intensity and preferred conditions are more likely.
- **Conditions:** Burning when humidity is high and wind speeds are low will reduce intensity.
- **Ignition:** Spot ignition rather than line ignition increases patchiness and orientating ignition so the fire burns against the wind or downslope will reduce intensity.

Grazing and fire recommendations specific to the Project Area are provided in Section 9.2.1.

9.1.2 Water source management

The results of this research program suggest that strategically placing additional permanent water sources will likely benefit SBTF in areas remote from existing water sources, which are more than approximately 3 km from permanent water. Recommendations for the placement and design of these water sources include:

- **Location**: Install new water sources in areas more than 3 km away from existing permanent water sources, as this is the typical distance that SBTF travel between water, foraging, and nesting areas (Section 3.1).
- **Habitat preference**: Place water sources in or near woodlands dominated by preferred habitat, e.g. silver-leaved ironbark (*Eucalyptus melanophloia*) woodlands (Section 3.1).
- **Proximity to shelter**: Locate water sources within 5 m of mature canopy trees to provide nearby shelter for SBTF and reduce the risk of predation from raptors (e.g., collared sparrowhawk, brown goshawk, Australian hobby, brown falcon).
- **Accessibility**: Ensure that water sources are permanently accessible to SBTF, as they likely require water daily throughout much of the year (Section 6.3 Zann, 1972).

- **Design features**: Include suitable perches for small passerines, allowing access even if water levels drop intermittently and permanent remote camera mounts that minimise variation in results between deployments (Section 6.3).
- **Exclusion of grazers**: Prevent access by cattle and macropods to avoid negative impacts on ground cover composition and grass seed production (Section 6.1) and damage to monitoring equipment such as remote cameras (Section 6.3).
- **Fire resistance**: Design water sources to be resistant to fire damage to minimise the management effort needed to protect them during planned burns or wildfires.

9.1.3 Rehabilitation

The primary goal of rehabilitation for SBTF habitat values is to restore open grassy woodlands with a native grass species rich ground layer, sparse shrubs and minimal exotic plant invasion (Sections 3.1, 4.1 and 4.2). Rehabilitation should aim to form established communities that have similar structure to surrounding high value SBTF habitat, which for our study area in the Desert Uplands comprised a canopy and subcanopy cover of approximately 15% and 2.5%, respectively.

Rehabilitation of the ground layer should aim to promote preferred SBTF forage species of higher nutritional value, as well as species that provide seed during periods of seed shortage. The balance of the available dietary evidence on SBTF to date (this research program; Mitchell, 1996; NRA Environmental Consultants, 2005; Meyer and Agnew, 2012; Rechetelo *et al.*, 2016; Mula Laguna *et al.*, 2019; Williams *et al.*, 2020) suggest the following species are important habitat components for SBTF:

- *Digitaria* spp.
- *Alloteropsis* spp.
- *Paspalidium* spp.
- *Panicum* spp.
- *Urochloa* spp.
- *Enneapogon* spp.
- *Schizachyrium* spp.
- *Cymbopogon* spp.
- *Chrysopogon* spp.
- *Setaria* spp.
- *Bothriochloa* spp.
- *Chloris* spp.
- Forage species indicative of native pastures in good condition, with low grazing pressures, including: *Themeda triandra*, *Sehima nervosum and Heteropogon contortus*.

Rehabilitation should aim to reduce the abundance of exotic species, even if the species are within the genera listed above, as SBTF have been found to select habitat with low exotic species cover (Section 4.1 and 4.2).

As a majority of these species are not commercially available, seeds will likely need to be harvested for propagation from surrounding areas of suitable habitat that are free of exotic species. In addition, some of the species listed above are fire-adapted and therefore either require heat or smoke to germinate or have

a greater chance of germinating when exposed to them (Read and Bellairs, 1999; Clarke and French, 2005; Carthey *et al.*, 2018). It is recommended that seed treatments are investigated and trialled to ensure rehabilitation is successful (Baxter *et al.*, 1994; Lloyd, Dixon and Sivasithamparam, 2000). Rehabilitation should prioritise high species richness to ensure a diversity of seed phenologies. This supports year-round seed availability and helps to fill seasonal resource gaps.

9.1.4 Detection and monitoring

Detection and monitoring methods for SBTF are key components of recovery plans and impact assessment guidelines (DAWE, 2009a; Eyre *et al.*, 2022). This research program has contributed to an understanding that may be used to update detection and monitoring requirements, particularly with regards to the SBTF population occurring within the Desert Uplands Bioregion. We provide general recommendations for the detection and monitoring of SBTF drawn from the cumulative experience of the research program and ongoing monitoring (Table 9.1-1). The chosen methods should align with the survey's objectives, such as detecting the presence or absence of SBTF (e.g. environmental impact assessments), monitoring changes in relative abundance (e.g. long-term monitoring), or estimating population size and health (to inform conservation efforts).

Table 9.1-1: Recommended detection and monitoring methods for the southern black-throated finch (SBTF)

Survey technique	Minimum effort
Determine SBTF presence or absence	
Water source watches	Water source watches should be conducted between 9 am and 3 pm, aligning with the peak drinking period of SBTF. Surveys are best undertaken during the late dry season (September to November), when temperatures are high and ephemeral water is scarce. Although watches may also be effective from December to May, the availability of ephemeral water during the wet season can make it difficult to detect drinking birds. Surveys should be avoided during the winter months (June to August), when cooler temperatures and dew reduce SBTF drinking activity. For the assessment of SBTF presence/absence in previously unassessed habitat, it is recommended that all water sources within the survey area, including ephemeral water sources, be surveyed at least twice during the late dry season.
Remote cameras	Remote cameras may supplement or replace water source watches where the water source is small enough to be fully captured within the camera's field of view and detection range. Suitable examples include troughs and small ephemeral water sources less than 2 m in diameter. Cameras should be positioned within 2 m of the water's edge and set to very high trigger sensitivity to maximise the likelihood of detecting small birds as they approach. Cameras should remain in place for a minimum of 5 days, with extended deployments of up to 28 days recommended where feasible.
Acoustic recorders	For the assessment of SBTF presence/absence in previously unassessed habitat, it is recommended that one acoustic recorder be deployed per 100 hectares of potential foraging habitat. Each recorder should remain in place for a minimum of 28 days. When positioning near a water source, place the recorder at the nearest tree line rather than at the water's edge. SBTF call more frequently at the tree line, before flying to the water's edge.

Survey technique	Minimum effort
Incidental observations	Incidental survey methods, such as driving transects and foot traverses are useful supplements to increase survey effort; however, these methods should not be relied upon to confirm absence.
Monitoring SBTF abundance	
Water source census	An annual water source census conducted in the late dry season (September to November) is recommended as the most efficient and reliable method for monitoring long-term changes in SBTF abundance across most study areas. The census should include as many water sources within SBTF habitat as possible. At small water sources, such as troughs, remote cameras can effectively supplement human-based surveys.
Acoustic recorders	Acoustic recorders are an effective method for monitoring SBTF, particularly within foraging habitats away from water sources. Due to seasonal variation in call activity, likely linked to increased vocalisations during breeding and fledging periods, acoustic monitoring is best suited for detecting inter-annual changes in SBTF abundance. Two publicly available call detection models can support data processing (van Osta *et al.*, 2023; Allen-Ankins, Laguna and Schwarzkopf, 2024); however, all potential detections should be verified by an observer familiar with black-throated finch calls and those of similar species (e.g. double-barred finch and shining bronze-cuckoo).
Assess SBTF population size	
Genetic assessment	Genetic sampling is a valuable tool for tracking long-term changes in SBTF populations through the estimation of effective population size. While not a direct population count, effective population size provides a robust indicator of population trends over time. Collecting genetic samples from approximately 25 SBTF per site is generally sufficient to generate reliable estimates (Adam Stowe pers. comm.). In addition to monitoring population trends, genetic sampling can examine the effects of habitat fragmentation on genetic isolation and can identify historical changes in population size and gene flow (Davis *et al.*, 2025; Hooper *et al.*, 2025). Undertaking genetic work requires mist-netting, collection of genetic material (e.g. blood samples), laboratory analysis and interpretation of genetic data. It must therefore be conducted under appropriate research and animal ethics approvals and in collaboration with specialists in avian genetics.

9.2 Recommended updates to the Project's Black-throated Finch Management Plan

9.2.1 Habitat management

A key principle of effective monitoring is the integration of monitoring results into the decision making framework for management (Robinson *et al.*, 2018). The BTFMP uses an adaptive management and monitoring approach that specifies:

- Management objectives
- Performance criteria

- Management actions
- Specific monitoring requirements
- Triggers for adaptive management; and
- Potential corrective actions should triggers be exceeded.

Recommended updates to the BTFMP following findings of the BTFRP and proposed revision of the SBTF monitoring program are detailed in Table 9.2-1. These recommendations are not intended to replace the existing requirements of the BTFMP entirely but should be considered as part of a comprehensive review of the BTFMP. The proposed adaptive management trigger thresholds are justified in the footnotes of Table 9.2-1.

Table 9.2-1: Recommended updates to the Black-throated Finch Management Plan (BTFMP; EcoLogical Australia, 2023) management requirements (Tables 8 and 9 of the BTFMP)

Item	Performance criteria	Management actions	Monitoring	Trigger for adaptive management and corrective actions[1]	Potential corrective actions
Management objective: Maintain or improve SBTF habitat quality					
1	Key habitat attributes for SBTF within the Project Area remain stable or increase.	Implement habitat management recommendations identified within this table.	• Data sourced from SBTF habitat monitoring transects (Table 9.2-2). • Assess temporal change using a generalised linear mixed effects model that accounts for site-specific variation (as a random effect). Analysis will be carried out across the entire Project Area and at a paddock scale.	A 10% decrease in a habitat suitability metric; however, the decrease must be sustained for a period of two years across the entire Project Area or three years at a paddock scale. The following metrics define a 10% decrease in habitat quality and should be averaged across all SBTF habitat monitoring sites: • Native perennial grass cover below 39% • Native perennial grass species richness below 5.4 per habitat quality site • Non-native ground layer species cover exceeding 18% • Shrub cover exceeding 7%; OR • Tree cover below 20% or above 30%.[5]	• Increased weed control. • Evaluate land management regimes, particularly with respect to fire regimes and grazing pressure. • Additional survey effort may be undertaken within a paddock if the trigger is met to verify the results. • Evaluate response options with specialists in consultation with DETSI and DCCEEW.
2	Implement an ecological/cultural burn programme	• Timing: Burns are to be scheduled during the late wet season and early dry season when grasses are curing but soil moisture is sufficient to maintain low fire intensity (less than 2 m scorch height). Opportunistic 'storm burns' may also be undertaken following out-of-season rainfall events when soil moisture is sufficient to maintain a low fire intensity. Avoid burning in the late dry season or in very dry years if wet season rainfall was substantially less than average. • Burn intensity: Aim for 25–50% of ground cover biomass to be ignited within a larger patch. Fire patch sizes should be less than 50 ha with a minimum of 500 m between patches. Target an average scorch height of less than 2 m. • Fire frequency: Aim for a fire return interval of between 5 and 7 years within black-throated finch habitat. This interval represents the approximate fire return frequency of a single point in the vegetation, rather than intervals between burning large patches. Therefore, each larger fire management block should be burnt approximately every 2–3 years to target a 25–50% patchy burn with an approximate 5-year fire return interval. • Fire planning and monitoring: Strategically plan and monitor burn areas, timing, and intensity. • Fencing maintenance: Ensure fencing is intact to facilitate stock management. Exclude stock from burnt areas until biomass has reached 1,500 kg/ha. • Maintaining firebreaks: Maintain firebreaks to minimise the risk of high-intensity wildfires during the late dry season. • Adaptation based on conditions: Increase the number of burn patches during favourable wet seasons, while reducing burns during extended drought periods to proactively manage biomass and the risk of uncontrolled fire.	• Post-burn monitoring (Table 9.2-2) • Habitat quality monitoring (Table 9.2-2)	• The average fire return interval (of a point in vegetation) exceeds 8 years or is less than 3 years. • More than 10% of a burn is mapped as high or extreme severity, or more than 50% of a patch is burnt. • Key habitat suitability metrics decrease, as defined within Item 1.	• Review fire management procedures, including timing, personnel, planning and resources.

Item	Performance criteria	Management actions	Monitoring	Trigger for adaptive management and corrective actions[1]	Potential corrective actions
3	Grazing management	• Management actions must be responsive to ecological conditions and informed directly by monitoring outcomes. Stocking rates will be capped at a maximum of 1 adult equivalent per 25 ha within SBTF habitat, unless higher stocking rates are necessary to mitigate wildfire risk. • Fire will be the primary tool to manage biomass within 5-years of the revised BTFMP's approval. Habitat management is considerate of a progressive shift from grazing focused biomass management to fire focused biomass management, which requires staff training, resources and experience. • Strategic grazing will be used as a management tool guided by results of habitat monitoring to adapt to seasonal risks in habitat management. For example, during the dry season when conditions are unsuitable for achieving low-intensity burns, adaptive grazing may be applied to manage biomass levels to mitigate wildfire risk. This adaptive approach provides an alternative method of biomass management where fire cannot be safely or effectively applied. • Manage stock in burnt areas via management of water availability to direct stock away from burnt areas or where large areas are burnt, total exclusion until biomass levels exceed 1,500 kg/ha. • Strategic grazing options include but are not limited to: 1. Rotational grazing. 2. Continuous grazing: A near-constant stocking rate within the paddock. 3. Continuous grazing with a wet season spell: Livestock graze continuously but are removed during the wet season. 4. Time-limited grazing: high stocking rate for a short duration 5. Livestock exclusion. • A review of land management actions will be undertaken annually to: 1. Identify the optimal grazing management regime 2. Assess the medium-term (approximately 8 years) vegetation response to the fire and grazing regimes; and 3. Provide recommendations for the 5-yearly review of the BTFMP.	• Grazing logs provided by the Agistee • Biomass monitoring (Table 9.2-2)	• Grazing limits are exceeded. • Across all paddocks that contain SBTF habitat, total standing dry matter (ground layer biomass) is determined to be: – less than 1,500 kg/ha on average; or – great enough to pose a wildfire risk to SBTF habitat • Key habitat suitability metrics decrease, as defined within Item 1.	• Evaluate land management regimes, particularly with respect to fire regimes and grazing pressure. • If triggered by low biomass levels, spell paddocks. • If triggered by high biomass levels, undertake controlled burns or, where burns are not suitable, introduce or increase grazing pressure.
4	Water management	• Five additional artificial water sources will be installed within the Offset Area within one year after the BTFMP is revised. • Water source design and placement will be reflective of the recommendations in Section 9.1.2. Water source placement will also aim to enhance connectivity between the northern and southern SBTF subpopulations, as presented in Figure 9.2-1. • All water sources will be maintained as per the current BTFMP requirements.	• Additional water sources will be added to the ongoing water source monitoring requirements listed in Table 13 of the BTFMP.	• Additional water sources are not installed within one year following the BTFMP revision. • Any water sources become inoperable. • Additional water source grazing and predator exclusion is breached, leading to the water source being available to grazing animals (stock and macropods) and aggressive birds (e.g. corvids).	• Water source installation and repair works

Item	Performance criteria	Management actions	Monitoring	Trigger for adaptive management and corrective actions[1]	Potential corrective actions
Management objective: Progressive habitat restoration of mine-disturbed areas					
5	Disturbed habitat is restored to high value habitat for BTF.	• Areas to be rehabilitated to SBTF habitat will target restoration to the soil and floristic composition of regional ecosystem 10.5.5a (silver-leaved ironbark *Eucalyptus melanophloia*, open woodland with a native tussock grass understorey). • Tree species that will be included within the seed mix are: *Eucalyptus melanophloia* and *Acacia sericophylla*. Rehabilitation should target a canopy cover of 15% *Eucalyptus melanophloia* and 2.5% *Acacia sericophylla*. • The rehabilitation seed mix will prioritise a high species richness of native grasses that afford diverse seed phenologies. The seed mix will be collected/established from a local provenance and include the following species: *Digitaria brownii, Digitaria ammophila, Alloteropsis semialata, Paspalidium rarum, Panicum effusum, Urochloa holosericea, Themeda triandra, Enneapogon virens, Schizachyrium fragile, Sehima nervosum, Cymbopogon bombycinus, Aristida pruinosa, Aristida calycina, Eriachne mucronata, Heteropogon contortus, Chrysopogon fallax*.	• Mine-disturbed rehabilitated areas will be included within annual habitat quality monitoring (Table 9.2-2). • Establish rehabilitation benchmarks that are consistent with the Habitat Quality Monitoring methods (e.g. for rehabilitation years 1, 5, 10, 15 and 20).	Habitat quality metrics that surpass the following criteria, on average across each rehabilitated area: • Native grass species richness is below 90% of the associated regional ecosystem benchmark per BioCondition plot for a period of two years. • After 10 years of rehabilitation, the vegetation includes <5% or >25% tree cover of the associated RE benchmark. • Non-native species cover is >10%.	• Oversewing of additional species • Review rehabilitation methods • Review soil amelioration.
Management objective: Maintain or increase the SBTF population in the Project Area					
6	The population of SBTF within the Project Area remains stable or increases	Implement habitat management recommendations identified within this table.	• Water source census program (Table 9.2-2) • Remote cameras (Table 9.2-2) • Mist-netting mark-recapture (Table 9.2-2) • Genetic analysis (Table 9.2-2) • Bioacoustics (Table 9.2-2) Assess temporal change using a generalised linear mixed effects model that accounts for site-specific variation (as a random effect).	A 10% decrease in SBTF abundance from the baseline; however, the decrease must be sustained for a period of two years. The following metrics define a 10% decrease in SBTF abundance: • Annual water source census population size below 580 SBTF within the Project Area[1] • Population size estimate from mark recapture sampling below 510 SBTF within the Project Area[1] • Effective population size (genetics) of the northern sub-population below 1150[2] • Effective population size (genetics) of the southern sub-population below 260[2]; OR • The average annual SBTF call abundance declines more than 10% from the 2025-2026 reporting year average (baseline levels to be calculated once all bioacoustic recorders are installed)[3]	• Investigate causes of decline, including local habitat changes, land management factors (fire, grazing or water) and genetic effects. • Undertaken genetic studies more frequently than every 5 years (Table 9.2-2). • Evaluate response options with specialists in consultation with DETSI and DCCEEW.
7	Forward indicators of the health of the SBTF population within the Project Area remain stable or improve.	Implement habitat management recommendations identified within this table.	• Mist-netting (Table 9.2-2) • Genetic analysis (Table 9.2-2) Report sex ratio and juvenile to be reported following mist-netting as per Section 3.3.4.	Demographic change in the SBTF population that includes a sex ratio of greater than 0.7 males: females or females:males.[4]	• Evaluate response options with specialists in consultation with DETSI and DCCEEW.

[1] The baseline population size estimated from the water source watch census in October 2024 was 652 SBTF (Section 3.5.3) and for the mark-recapture sampling in October 2023 was 567 SBTF (Section 3.5.1). Population size estimates that are 10% lower than these baseline value were selected to allow for natural fluctuations in the species' population and inter-survey estimation error.

[2] The baseline effective population sizes were between 1280 and 1610 for the northern subpopulation (north of the Carmichael River) and 290–360 for the southern subpopulation (south of the Carmichael River)(Section 3.2.5.2). These included samples collected at Hyde Park Station and Lignum Station, which are outside of Bravus control. The effective population size thresholds that are approximately 20% lower than the baseline values have been selected to account for potential natural fluctuations in the species' population, inter-survey estimation error and effects of land-management on adjacent properties, which is outside of Bravus' control.

[3] Preliminary analysis of bioacoustic data from the study area has identified seasonal variation in the call rates of SBTF, which is correlated with the timing of breeding activities (van Osta, 2025). Annual changes in SBTF call activity is therefore an appropriate measure to monitor population change as seasonal effects are accounted for. A 10% change is proposed as the threshold to allow for natural fluctuations in population size and potentially calling activity should breeding activity be reduced (e.g. during a drought year).

[4] The baseline sex ratio of SBTF is approximately even (Section 3.4). A ratio of 0.7 would indicate a 20% change to the approximately equal ratio. A 20% change was chosen to optimise the balance between detecting a meaningful change in the sex ratio while minimising the chance of a false-positive trigger event. The mist-netting recommendations propose to capture 25 SBTF per sub-population each sampling period, which equates to a ~7:18 ratio (sex A:sex B), which is approximately 2.2% chance of occurring at random in an equal weight population. For reference, a ratio of 0.6 would be a 10% change and equivalent to a 10:15 ratio, which is a 21% chance of occurring at random in one year and 10% chance of occurring at random 2 years in a row.

[5] Baseline values for each metric were calculated using the 2024 wet season habitat quality survey data for all SBTF habitat sites. The baseline metrics were: native perennial grass cover of 44.2%; native grass species richness of 6.74; non-native ground layer species cover of 15.6%; shrub cover of 6.16%; and tree cover of 24.5%. An approximately 20% change in each metric was allowed for to account for habitat variation in response to climatic factors (e.g. droughts) and inter-survey estimation error.

9.2.2 Monitoring program

The BTFMP requires a major review of the monitoring program to be completed after 10 years of implementation (EcoLogical Australia, 2023). This timing coincides with the completion of the BTFRP, allowing the review to consider both the SBTF monitoring program results to date and BTFRP results. The aim of this review is to finalise the long-term monitoring method that will be implemented for the life of the Project as required by the EPBC Act approval conditions.

Monitoring the SBTF population within a large area such as the Carmichael Mining Lease and offsets area will require a variety of methods. Monitoring methods and survey efforts should be designed to meet the objectives of the monitoring program. The following objectives for the SBTF monitoring program are recommended to demonstrate the effectiveness of management measures adopted in the BTFMP and Moray Downs West Offset Area Management Plan (MDW OAMP):

- Track temporal and spatial population change within the Project Area
- Link habitat utilisation to land management factors, specifically: fire, grazing, water and habitat condition (e.g. native grass richness, forage species, weed cover and shrub cover)
- Monitor breeding activities, specifically nesting and the presence of juvenile birds with the Project Area; and
- Provide feedback to enable adaptive management measures to be implemented under the BTFMP and MDW OAMP.

Table 9.2-2 outlines the recommended monitoring methods aimed at tracking the population, demographic structure, and habitat utilisation of SBTF within the Project Area. These methods also provide early indicators of habitat quality. The monitoring methods include:

- **Water source watches, remote cameras, and mist-netting**: These methods focus on monitoring SBTF populations, particularly their use of water sources during the dry season when water is limited to permanent sources.
- **Bioacoustic recorders**: Used to monitor occupancy and relative abundance in foraging and breeding habitats.
- **Routine water source checks or remote monitoring**: These confirm that water remains permanently available.
- **Permanent habitat assessment transects**: Conducted annually, using consistent methods between the BTFMP and the MDWOAMP.
- **Biomass monitoring**: Conducted biannually using remote sensing methodology to inform fire and grazing management.
- **Fire monitoring**: Evaluations to monitor the extent and intensity of burns.

Data from these monitoring methods feeds into a Trigger Action Response Plan (TARP) that links monitoring outcomes to management objectives.

Table 9.2-2: Recommendations for SBTF monitoring methods

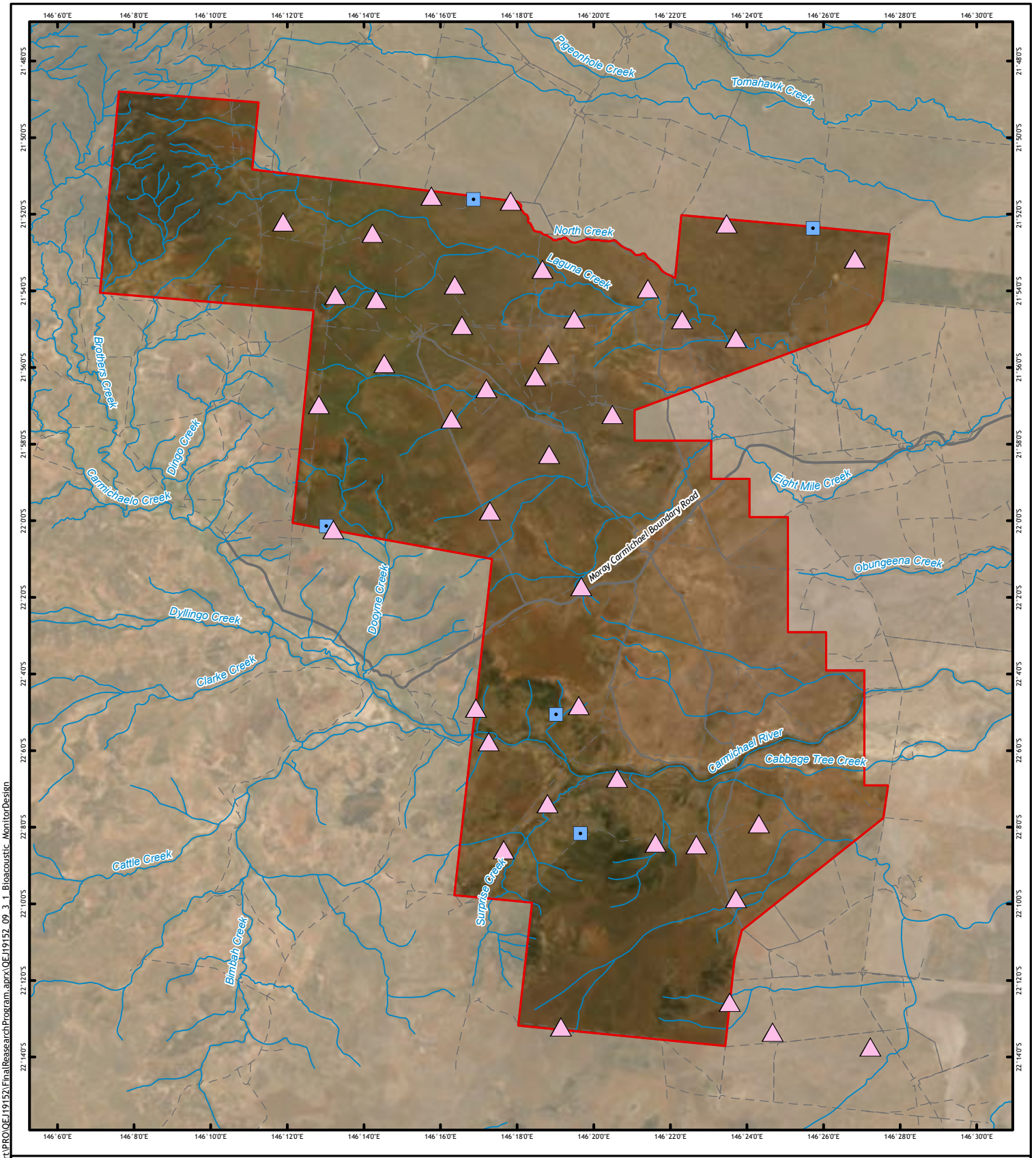
Method	Objectives	Timing	Monitoring requirements
Water source census	A 'census' style survey that records the SBTF population at all permanent water sources within the Project Area. The survey's primary objective is to monitor for long-term changes in the population of SBTF inhabiting the Project Area.	Annual survey in the late dry season (September to October).	Simultaneous remote camera and human-observer water watches that includes: • Dams, turkey's nests, tanks and other large water sources monitored by ecologists from 9 am to 3 pm. • Remote cameras at all troughs, for a period of 5 days per remote camera. An average SBTF use over the 5-day period will be calculated to account for potential variation in detection rates. Remote cameras are to be active across all daylight hours. • Simultaneous remote camera and human-observer water source watches are to be undertaken annually to calibrate remote camera detections to human observations. GLMM to evaluate temporal changes in SBTF detections, with water source included as a random effect to account for site-level variation. • Interpretation of results should be informed by conditions such as rain and temperature that may bias the population estimate. Remote camera detections to be reviewed annually to confirm peak drinking times to determine if the survey period (9am to 3pm) was likely to detect most SBTF.

Method	Objectives	Timing	Monitoring requirements
Mist-netting	• Mark-recapture population estimates coinciding with water source watches. • Genetic estimate of effective population size • Demographics (juvenile presence, sex ratio)	Every 5 years in the dry season (approximately one week prior to the water source census), unless other population monitoring methods present a 10% decline, in which case monitoring should occur more frequently.	• Capture and genetic sampling (blood prick) of 25 SBTF from each subpopulation (north and south of the Carmichael River). • Engage a genetics laboratory to calculate the effective population sizes and sex ratios of the northern and southern sub-populations. • Banding and recapture of birds to estimate population size at each water source using a Cormack Jolly Sieber mark-recapture model (Cooch and White, 2019). • Mist-netting should focus on water sources that have the largest number of SBTF drinking from them. A minimum of two days of mist-netting at each water source will be required, separated by a minimum of two days to allow time for banded birds to mix back into the flocks.
Bioacoustics	• Monitor habitat utilisation • Provide a relative abundance estimate that is independent of water source use. • Response to habitat changes (e.g. fire, weed management, water) • Habitat utilisation	Permanent – maintenance and SD card replacement during habitat monitoring	• An array of 40 bioacoustic recorders installed at each of the 40 habitat monitoring sites, which allows for 2 recorders per paddock. The proposed bioacoustic recorder sites are depicted in (Figure 9.2-1). • Screen all recordings with a call detection model (Section 7.4; van Osta et al., 2023) and detections with a moderate confidence or above (50% confidence of being an SBTF call) will be manually verified by an ecologist experienced with SBTF calls and local bird community. • Spatial and temporal changes in the number of SBTF calls per day will be used to monitor the relative abundance of SBTF within foraging and nesting habitat throughout the Project Area. Tracking temporal changes in SBTF relative abundance requires inclusion of variables to account for potential seasonal changes in SBTF call rates (e.g. due to breeding). Such seasonal changes can be accounted for by including season as a fixed effect in a generalised linear mixed effects model.

Method	Objectives	Timing	Monitoring requirements
Water source inspections	Ongoing regular inspections of water sources to ensure they are accessible to SBTF	• Fortnightly routine water source checks; OR • Remote sensing	The BTFMP version 9c (EcoLogical Australia, 2023) requires monitoring of water levels and availability through fortnightly inspections and weekly inspections at Ten Mile Bore. No change is proposed to the fortnightly checks; however, it is noted that remote monitoring may be used to autonomously monitor water source levels, where such systems are available.
Habitat monitoring	• Track habitat quality for SBTF • Provide early indicators of habitat changes that may benefit or hinder the SBTF population • Provide feedback to inform the fire program.	Annual (wet season)	A total of 104 permanent vegetation monitoring transects have been established in SBTF habitat under the BTFMP (n = 40) and OAMP (n = 64). It is recommended that the habitat monitoring methods within the BTFMP and MDWOAMP be consolidated to a standard method under the BTFMP. Habitat monitoring is to include: • Tree, shrub, grass and forb species • Groundcover quadrats including individual species cover • Number of large trees • Canopy and subcanopy median height • Crown cover of emergent, canopy, subcanopy and shrub layer • Recruitment of canopy species • Abundance of coarse woody debris

Method	Objectives	Timing	Monitoring requirements
Biomass monitoring	Monitor biomass levels to inform fire and grazing management decisions	Biannual in the late wet season (March/April) and late dry season (October)	• It is recommended that biannual remotely sensed biomass measurements (e.g. estimates sourced from Sentinel-2 satellites or drone imagery) are undertaken to assess total standing dry matter. This method provides significant temporal and spatial advantages over on-ground biomass estimates, including: – High-frequency monitoring: Data is captured approximately every five days, enabling near real-time tracking of biomass changes and supporting timely adaptive management. – Broad spatial coverage: Satellite-based monitoring allows for consistent observation across large, remote, or inaccessible areas, overcoming limitations of manual fieldwork. – Consistency and repeatability: Data is sourced from independent third-party satellites, reducing human bias and enabling reliable year-on-year comparisons. – Retrospective analysis: Historical imagery supports long-term trend analysis, enhancing decision-making for land and pasture management. • To field-verify high biomass levels, photo estimates will be undertaken in the wet season during habitat quality monitoring.

Method	Objectives	Timing	Monitoring requirements
Fire monitoring	• Track area and fire severity of all burns that occur within the Project Area. • Provide feedback to inform the fire program.	Within 3 months following burns.	• **Comprehensive burn mapping**: The extent and severity of burns are to be mapped using aerial imagery, such as Sentinel-2 satellite imagery, which is captured approximately every 10 days (European Space Agency, 2023). High resolution aerial or drone imagery may be used as an alternative to Sentinel-2 satellite imagery when available. Burn mapping will be used to identify the area of habitat burnt. Burn severity will be mapped using a method developed by DETSI as part of a Post-fire Assessment Report (Hines *et al.*, 2021). The difference between the pre-fire and post-fire Normalised Burn Ratio will be classified into relative fire severity classes to produce severity mapping across the Project Area. Severity classes will include: – **Unburnt:** Unburnt, canopy and subcanopy unchanged – **Low:** Canopy and subcanopy unscorched, shrubs may be scorched, fire-sensitive low shrubs may be killed – **Moderate:** Partial canopy scorch, subcanopy partially or completely scorched, and/or fire-sensitive tall shrub or small tree layer mostly killed – **High:** Full canopy scorch to partial canopy consumption, subcanopy fully scorched or consumed; and – **Extreme:** Full canopy, subcanopy and understorey consumption. • **Vegetation monitoring**: Vegetation response to burns will be monitored using the existing habitat quality method and sites, as detailed above. If an area of more than 1,000 ha with no existing habitat quality monitoring sites is burnt, a new site will be added. • **On-ground burn assessments:** Assessments will be undertaken during habitat quality monitoring to inform burn intensity mapping. Data collected will include scorch height, percentage of the ground layer burnt, and percentage of the canopy burnt.



Legend

- Proposed BAR Location
- Proposed water source location
- Watercourse
- Major Road
- Minor Road
- Track
- Project Area

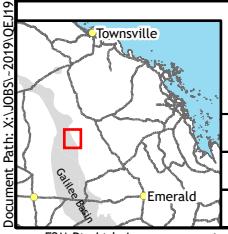
N

Scale 1:250,000 (A4)

0 5 10

Kilometres

Coordinate System: GDA2020



Notes:
Aerial Imagery: © ESRI 2021
Cadastre: © DNRME 2021
Watercourse: © Geoscience Australia 2012
Road: © DNRME 2021

Rev	Description	Drawn	Approved	Date
2	Issued for Use	JY	JMC	30/10/2025
0	Issued for Use	GO	JMC	08/08/2025

FIGURE 9.3-1: BIOACOUSTIC MONITORING DESIGN

BTFRP Final Report
Bravus

Map Number	Job Number	Rev
1 of 1	QEJ19152	2

Document Path: X:\JOBS\-2019\QEJ19152\GIS\Final Research Program Report\PRO\QEJ19152\FinalReasearchProgram.aprx\QEJ19152_09_3_1_Bioacoustic_MonitorDesign

E2M Pty Ltd gives no warranty in relation to the data (including accuracy, reliability, completeness or suitability) and accepts no liability for any loss, damage or costs (including consequential damage) relating to any use of the data in this map.

9.3 Opportunities for additional research

The findings and methodological advancements presented in this report open avenues for further research. These opportunities build on the BTFRP and broader literature and aim to improve the conservation of SBTF within the Project Area and across the species' distribution. The sections below detail opportunities for further research that support the objectives of the BTFMP (Section 9.3.1) and broader conservation objectives for the black-throated finch subspecies across its range (Section 9.3.2).

9.3.1 Research opportunities to support the Black-throated Finch Management Plan

Rehabilitation trial of SBTF foraging grasses in the Desert Uplands: Section 9.1.3 includes a list of recommended grass species to be included within rehabilitation areas of the Carmichael Coal Mine. The species mix is based on our research on the diet selection of SBTF within the Project Area; however, further investigation of rehabilitation strategies for these species is likely to be required. Many of these grass species are not currently commercially available and there is a dearth of literature to inform optimal harvesting, pre-treatment and maintenance methods for these species. Rehabilitation trials may be undertaken at Carmichael Coal Mine, which would support both the implementation of recommendations in Section 9.1.3 and provide valuable information to support other rehabilitation projects in SBTF habitat.

Tracking the use of water sources by SBTF using radio frequency identification (RFID) tags. Estimating population size from water source monitoring and remote camera detections requires an estimate of the frequency at which SBTF visit water sources. While calibration methods have been conducted (see Section 3.5.3), additional and higher temporal resolution data on water source use could be obtained through RFID tags and receivers. These tags are extremely lightweight (<0.2 g) and battery-free but have a limited detection range (<10 cm)(Bonter and Bridge, 2011). RFID technology is commonly used to monitor bird activity at feeders and nest boxes (Bonter and Bridge, 2011). Further research could explore its potential for tracking SBTF at small water sources, such as cattle troughs or newly installed offset-area water sources.

Complementary research projects using Bravus' long-term monitoring data. Bravus' ecological monitoring is producing and will continue to produce high-quality long-term monitoring data that is likely to be equally valuable to research projects as it is to answering the primary monitoring aims. Research topics that may be considered using Bravus' monitoring data may include:

- Investigating the spatiotemporal partitioning of resources by BTF compared to sympatric granivorous birds for which BTF compete for food.
- Response of avian foraging guilds to fire management.
- Long-term fluctuations in the avifauna of the Desert Uplands.

9.3.2 Research opportunities to support conservation of the broader southern black-throated finch population

Breeding success is a key factor that impacts recruitment to the SBTF population. Further research may be undertaken to investigate factors that affect breeding success, which may include:

- Triggers for the timing of breeding (e.g. resource availability)
- Causes of failed nesting, e.g. depredation or starvation; and

- Potential to increase nest success, which has been a strategy implemented for other threatened granivores, including the golden shouldered parrot (Olkola Aboriginal Corporation, 2022) and Gouldian finch (Brazill-Boast *et al.*, 2011).

Engagement of Desert Uplands landholders to investigate conservation of SBTF outside of protected areas: The offset areas associated with the Project are the only protected areas with post-2000 SBTF records within the Desert Uplands (DCCEEW, 2023). While some SBTF sightings have been recorded from Nature Refuges in the Desert Uplands (DCCEEW, 2023), these areas are not subject to the same conservation focussed grazing and fire management practices, nor require a monitoring program (Allen *et al.*, 2018), as are required for the Project Area. Furthermore, the SBTF population within the Project Area may be affected by the health of SBTF populations on surrounding properties, which has an impact on the species' genetic diversity and adaptive capacity (Section 3.2). The key threat that is largely unmanaged is overgrazing. Stocking rates within the Desert Uplands have historically been low (Kutt and Woinarski, 2007); however, there has been a trend of smaller paddock sizes and proliferation of new water sources that have resulted in a more uniform, higher grazing pressure landscape (Juliana McCosker pers. com.; Landsberg *et al.*, 2003) Additional research may be considered on incentive structures to support SBTF conservation outside of protected areas, such as those that potentially extend on the conservation auction system implemented by Windle *et al.* (2009).

Estimate SBTF density using bioacoustics: Section 9.1.4 recommends the use of acoustic recorders to track SBTF call activity, such as the number of calls per day. While this is valuable, estimating species density (e.g., birds per unit area) would enhance the utility of call detection models (Pérez-Granados and Traba, 2021). To achieve this, additional research may be undertaken to establish the relationship between call activity and the number of birds present (Pérez-Granados and Traba, 2021). The research must be considerate of factors that affect individual call rates, including variability in calling behaviour across seasons, habitats, flock size dynamics and environmental conditions (Hutschenreiter et al., 2024). The effective detection radius of SBTF calls from bioacoustic recorders must also be estimated (Pérez-Granados and Traba, 2021). Integrating complementary data sources, such as direct observations, could help calibrate call activity against the number of individuals present (Pérez-Granados and Traba, 2021). Such research would improve the applicability of call detection models for estimating species abundance, thereby expanding the role of bioacoustics in conservation and ecological monitoring.

Conservation genetics assessments of the remaining SBTF subpopulations. Genetic studies of SBTF that have been undertaken for SBTF within and surrounding the Project Area (Section 3.2) and in the Townsville Plains (Tang, 2016) have found that SBTF populations can be isolated by barriers of unsuitable habitat less than 10 km wide. Additional genetic work is currently being undertaken for the Townsville plains SBTF population (Simon Griffith pers. com.); however, a substantial area of potential SBTF habitat in the northern Desert Uplands and Einasleigh Uplands Bioregions remains unstudied from a genetic perspective. Genetic assessments of these under-studied SBTF populations would support the identification of:

- At-risk and isolated sub-populations
- Core populations of higher genetic diversity; and
- Priority populations to target conservation action.

Whole genome sequencing analyses the complete DNA sequence of an organism's genome, providing insights into genetic variation, evolutionary history, and potential functional impacts of mutations. Whole genome sequencing data can be used to inform levels of both 'realised' and 'potential' genetic load within SBTF populations, i.e., the reduction in fitness due to the accumulation of deleterious mutations (Haldane, 1937; Muller, 1950). 'Realised' genetic load is the proportion of total genetic load expressed in the current population and is shaped by short-term demographics (e.g., recent inbreeding), while

'potential' genetic load informs on potential fitness losses in future generations (Mathur and DeWoody, 2021; Mathur *et al.*, 2023). Population declines can increase the level of realised genetic load in wild populations, and thus assessing genetic load can help ensure management plans for the Desert Uplands SBTF population remain adaptive in the face of ongoing environmental change. Whole genome sequencing data can also be used to more accurately reconstruct recent demographic history by examining runs of homozygosity across the genome.

10 References

Abdu, S., Lee, A.T. and Cunningham, S.J. (2018) 'The presence of artificial water points structures an arid-zone avian community over small spatial scales', *Ostrich*, 89(4), pp. 339–346. Available at: https://doi.org/10.2989/00306525.2018.1509904.

Abdulwahab, U.A., Osinubi, S.T. and Abalaka, J. (2019) 'Risk of predation: a critical force driving habitat quality perception and foraging behavior of granivorous birds in a Nigerian forest reserve', *Avian Research*, 10(1), p. 33. Available at: https://doi.org/10.1186/s40657-019-0173-6.

Abraham, J. (1998) 'An Ecological Theory of Sexual Dimorphism in Animals', *Acta Biotheoretica*, 46, pp. 23–35.

Adani (2020) *Black-throated Finch Research Program Project Plan*.

Adelman, J.S. *et al.* (2015) 'Feeder use predicts both acquisition and transmission of a contagious pathogen in a North American songbird', *Proceedings of the Royal Society B*, 282(1815). Available at: http://dx.doi.org/10.1098/rspb.2015.1429.

Agbeshie, A.A. *et al.* (2022) 'A review of the effects of forest fire on soil properties', *Journal of Forestry Research*, 33(5), pp. 1419–1441. Available at: https://doi.org/10.1007/s11676-022-01475-4.

Alba, C. *et al.* (2015) 'Native and exotic plant species respond differently to wildfire and prescribed fire as revealed by meta-analysis', *Journal of Vegetation Science*. Edited by J. Morgan, 26(1), pp. 102–113. Available at: https://doi.org/10.1111/jvs.12212.

Alerstam, T. and Högstedt, G. (1982) 'Bird migration and reproduction in relation to habitats for survival and breeding', *Ornis Scandinavica*, pp. 25–37.

Alhomayani, F. and Mahoor, M.H. (2020) 'Deep learning methods for fingerprint-based indoor positioning: A review', *Journal of Location Based Services*, 14(3), pp. 129–200. Available at: https://doi.org/10.1080/17489725.2020.1817582.

Alldredge, M.W., Simons, T.R. and Pollock, K.H. (2007) 'Factors affecting aural detections of songbirds', *Ecological Applications*, 17(3), pp. 948–955. Available at: https://doi.org/10.1890/06-0685.

Allen, A.M. and Singh, N.J. (2016) 'Linking movement ecology with wildlife management and conservation', *Frontiers in Ecology and Evolution*, 3, p. 155. Available at: https://doi.org/10.3389/fevo.2015.00155.

Allen, A.N. *et al.* (2021) 'A convolutional neural network for automated detection of humpback whale song in a diverse, long-term passive acoustic dataset', *Frontiers in Marine Science*, 8, p. 607321. Available at: https://doi.org/10.3389/fmars.2021.607321.

Allen, L. *et al.* (2018) *Nature Refuges in Queensland: report to The Pew Charitable Trusts and Queensland Trust for Nature*. Available at: https://ourlivingoutback.org.au/wp-content/uploads/2023/10/FINALPew_nature_refuge_report__24102018_1.pdf (Accessed: 12 March 2025).

Allen-Ankins, S., Laguna, J.M. and Schwarzkopf, L. (2024) 'Towards automated detection of the endangered southern black-throated finch (Poephila cincta cincta)', *Wildlife Research*, 51(10).

Allendorf, F.W. and Luikart, G. (2009) 'Conservation and the genetics of populations', *John Wiley & Sons* [Preprint].

Allentoft, M.E. *et al.* (2010) 'Highly skewed sex ratios and biased fossil deposition of moa: ancient DNA provides new insight on New Zealand's extinct megafauna', *Quaternary Science Reviews*, 29(5), pp. 753–762. Available at: https://doi.org/10.1016/j.quascirev.2009.11.022.

Alonso, J.C. *et al.* (1995) 'Patch use in cranes: a field test of optimal foraging predictions', *Animal Behaviour*, 49(5), pp. 1367–1379. Available at: https://doi.org/10.1006/anbe.1995.0167.

Alston, J.M. *et al.* (2023) 'Mitigating pseudoreplication and bias in resource selection functions with autocorrelation-informed weighting', *Methods in Ecology and Evolution*, 14(2), pp. 643–654. Available at: https://doi.org/10.1111/2041-210x.14025.

Amec (2012) *Kevin's Corner Coal Mine supplementary MNES report*. Brisbane: Prepared for Hancock Galilee Pty Ltd.

Anderle, M. *et al.* (2023) 'Habitat heterogeneity promotes bird diversity in agricultural landscapes: Insights from remote sensing data', *Basic and Applied Ecology*, 70, pp. 38–49.

Anderson, D. and Burnham, K. (2004) *Model selection and multi-model inference*. 2nd edn. New York: Springer. Available at: https://doi.org/10.1007/b97636.

Andersson, M. and Simmons, L.W. (2006) 'Sexual selection and mate choice', *Twenty years of TREE - part I*, 21(6), pp. 296–302. Available at: https://doi.org/10.1016/j.tree.2006.03.015.

Andrew, M.H. and Mott, J.J. (1983) 'Annuals with transient seed banks: the population biology of indigenous Sorghum species of tropical north-west Australia', *Australian Journal of Ecology*, 8(3), pp. 265–276.

Archibald, S. *et al.* (2005) 'Shaping the landscape: Fire-grazer interactions in an African savanna', *Ecological Applications*, 15(1), pp. 96–109. Available at: https://doi.org/10.1890/03-5210.

Archibald, S., Hempson, G.P. and Lehmann, C. (2019) 'A unified framework for plant life-history strategies shaped by fire and herbivory', *New Phytologist*, 224(4), pp. 1490–1503.

Arnaiz-Villena, A. *et al.* (2009) 'Estrildinae finches (Aves, Passeriformes) from Africa, South Asia and Australia: a molecular phylogeographic study', *The Open Ornithology Journal*, 2(1).

Arthan, W. *et al.* (2022) 'Heteropogon-Themeda grasses evolve to occupy either tropical grassland or wetland biomes', *Journal of Systematics and Evolution*, 60(3), pp. 653–674.

Ash, A. *et al.* (1997) 'Building grass castles: Integrating ecology and management of Australia's tropical tallgrass rangelands.', *The Rangeland Journal*, 19(2), pp. 123–144.

Austecology (2011) *May 2011 black-throated finch surveys and habitat assessments: mine site Galilee Coal Project (Northern Export Facility)*. Prepared for Waratah Coal Pty Ltd.

Austin, P.C., Lee, D.S. and Leckie, G. (2020) 'Comparing a multivariate response Bayesian random effects logistic regression model with a latent variable item response theory model for provider profiling on multiple binary indicators simultaneously', *Statistics in Medicine*, 39(9), pp. 1390–1406.

Baccianella, S., Esuli, A. and Sebastiani, F. (2009) 'Evaluation measures for ordinal regression', in *2009 Ninth international conference on intelligent systems design and applications*. IEEE, pp. 283–287. Available at: https://doi.org/10.1109/isda.2009.230.

Back, P.V. (2005) 'The impact of fire on population density and canopy area of currant bush (Carissa ovata) in central Queensland and its implications for grazed woodland management', in. Available at: https://api.semanticscholar.org/CorpusID:130105594.

Baker, C. (2014) *Golden Shouldered Parrots In Oz*. Available at: https://goldenshoulderedparrotinoz.weebly.com/.

Baldwin, M. (1976) 'Distribution of the Black-throated Finch', *Australian Birds*, 11, pp. 13–14.

Barbet-Massin, M. *et al.* (2012) 'Selecting pseudo-absences for species distribution models: How, where and how many?', *Methods in Ecology and Evolution*, 3(2), pp. 327–338. Available at: https://doi.org/10.1111/j.2041-210x.2011.00172.x.

Barea, L.P. (2012) 'Habitat influences on nest-site selection by the Painted Honeyeater (Grantiella picta): do food resources matter?', *Emu-Austral Ornithology*, 112(1), pp. 39–45.

Barnes, M.A. *et al.* (2014) 'Environmental conditions influence eDNA persistence in aquatic systems', *Environmental science & technology*, 48(3), pp. 1819–1827.

Barras, A.G. *et al.* (2020) 'The importance of seasonal environmental factors in the foraging habitat selection of Alpine Ring Ouzels Turdus torquatus alpestris', *Ibis*, 162(2), pp. 505–519.

Barrett, G. *et al.* (2003) *The New Atlas of Australian Birds*. Melbourne: Birds Australia.

Barton, K. (2023) *MuMIn: Multi-model inference*. Available at: http://CRAN.R-project.org/package=MuMIn.

Bastin, G.N. *et al.* (2009) 'The Australian Collaborative Rangelands Information System: preparing for a climate of change', *The Rangeland journal*, 31(1), pp. 111–125.

Bateman, B.L. and Johnson, C.N. (2011) 'The influences of climate, habitat and fire on the distribution of cockatoo grass (*Alloteropsis semialata*) (Poaceae) in the Wet Tropics of northern Australia', *Australian Journal of Botany*.

Bates, D. *et al.* (2015) 'Fitting Linear Mixed-Effects Models Using lme4', *Journal of Statistical Software*, 67(1), pp. 1–48. Available at: https://doi.org/10.18637/jss.v067.i01.

Baxter, B.J.M. *et al.* (1994) 'Plant-derived smoke and smoke extracts stimulate seed germination of the fire-climax grass Themeda triandra', *Environmental and Experimental Botany*, 34(2), pp. 217–223. Available at: https://doi.org/10.1016/0098-8472(94)90042-6.

Beauchamp, G. (2022) 'Flocking in birds is associated with diet, foraging substrate, timing of activity, and life history', *Behavioral Ecology and Sociobiology*, 76(6), p. 74. Available at: https://doi.org/10.1007/s00265-022-03183-9.

Bedoya-Perez, M.A. *et al.* (2013) 'A practical guide to avoid giving up on giving-up densities', *Behavioral Ecology and Sociobiology*, 67(10), pp. 1541–1553. Available at: https://doi.org/10.1007/s00265-013-1609-3.

Beerling, D.J. and Osborne, C.P. (2006) 'The origin of the savanna biome', *Global change biology*, 12(11), pp. 2023–2031. Available at: https://doi.org/10.1111/j.1365-2486.2006.01239.x.

Beery, S., Morris, D. and Yang, S. (2019) 'Efficient pipeline for camera trap image review', *arXiv preprint arXiv:1907.06772* [Preprint].

Beggs, R. *et al.* (2020) 'An empirical test of the mechanistic underpinnings of interference competition', *Oikos*, 129(1), pp. 93–105.

Beng, K.C. and Corlett, R.T. (2020) 'Applications of environmental DNA (eDNA) in ecology and conservation: opportunities, challenges and prospects', *Biodiversity and conservation*, 29(7), pp. 2089–2121.

Benkman, C.W. (1991) 'Predation, seed size partitioning and the evolution of body size in seed-eating finches', *Evolutionary Ecology*, 5(2), pp. 118–127. Available at: https://doi.org/10.1007/BF02270828.

Bennett, A.T.D. and Cuthill, I.C. (1994) 'Ultraviolet vision in birds: What is its function?', *The Biology of Ultraviolet Reception*, 34(11), pp. 1471–1478. Available at: https://doi.org/10.1016/0042-6989(94)90149-X.

Bennett, L.T., Judd, T.S. and Adams, M.A. (2003) 'Growth and nutrient content of perennial grasslands following burning in semi-arid, sub-tropical Australia', *Plant Ecology*, 164(2), pp. 185–199. Available at: https://doi.org/10.1023/A:1021253600712.

Bergeron, L.A. *et al.* (2023) 'Evolution of the germline mutation rate across vertebrates.', *Nature*, 615(7951), pp. 285–291.

Bergler, C. *et al.* (2019) 'ORCA-SPOT: An automatic killer whale sound detection toolkit using deep learning', *Scientific Reports*, 9(1), pp. 1–17. Available at: https://doi.org/10.1038/s41598-019-47335-w.

Bergler, C. *et al.* (2022) 'ANIMAL-SPOT enables animal-independent signal detection and classification using deep learning', *Scientific Reports*, 12(1), p. 21966. Available at: https://doi.org/10.1038/s41598-022-26429-y.

Berlusconi, A. *et al.* (2024) 'Habitat partitioning and spatial segregation at multiple scales promotes year-round coexistence in a guild of forest songbirds', *bioRxiv*, pp. 2024–08.

Beumer, L.T. *et al.* (2020) 'An application of upscaled optimal foraging theory using hidden Markov modelling: year-round behavioural variation in a large arctic herbivore', *Movement ecology*, 8, pp. 1–16.

Bijlsma, R. and Loeschcke, V. (2012) 'Genetic erosion impedes adaptive responses to stressful environments.', *Evolutionary Applications*, 5(2), pp. 117–129.

Bird, J.P. *et al.* (2020) 'Generation lengths of the world's birds and their implications for extinction risk.', *Conservation biology*, 34(5), pp. 1252–1261.

BirdLife Australia (2025) *Birdata by BirdLife Australia*. Available at: https://birdata.birdlife.org.au/explore#map=-22.5083100_136.0786120_4.

BirdLife International (2016) *Poephila cincta. The IUCN Red List of Threatened Species 2016: e.T22719692A94639584*.

Birds Canada (2017) *Motus Wildlife Tracking System, Motus Landowner Information*. Available at: https://motus.org/wp-content/uploads/2017/08/Motus_LandownerInformation_April2017.pdf.

Bishop, C.M. (2006) *Pattern recognition and machine learning*. New York: Springer. Available at: https://doi.org/10.1007/978-0-387-45528-0.

Black-throated Finch Recovery Team (2004) *Recovery Plan for the Black-throated Finch Southern Subspecies Poephila cincta cincta*. Hurstville and Brisbane: Department of Environment and Conservation and Queensland Parks and Wildlife Service.

Black-throated Finch Recovery Team (2007a) 'National recovery plan for the black-throated finch southern subpecies (Poephila cincta cincta)'. Department of Environment and Conservation.

Black-throated Finch Recovery Team (2007b) *Recovery Plan for the Black-throated Finch Southern Subspecies Poephila cincta cincta*. Hurstville and Brisbane: Department of Environment and Conservation and Queensland Parks and Wildlife Service.

Black-throated Finch Recovery Team (2020) *Conversation action plan summary for the black-throated finch southern subspecies*.

Bland, L.M. *et al.* (2015) 'Predicting the conservation status of data-deficient species', *Conservation Biology*, 29(1), pp. 250–259. Available at: https://doi.org/10.1111/cobi.12372.

Bliege Bird, R. *et al.* (2020) 'Fire mosaics and habitat choice in nomadic foragers', *Proceedings of the National Academy of Sciences*, 117(23), pp. 12904–12914.

Block, W. and Brennan, L. (1993) 'The habitat concept in ornithology', in *Current Ornithology*. Springer, pp. 35–91.

Block, W.M. and Brennan, L.A. (1993) 'The habitat concept in ornithology', in *Current ornithology*. Springer, pp. 35–91.

Bogdziewicz, M., Zwolak, R. and Crone, E.E. (2016) 'How do vertebrates respond to mast seeding?', *Oikos*, 125(3), pp. 300–307.

Boguslav, M. and Cohen, K.B. (2017) 'Inter-annotator agreement and the upper limit on machine performance: Evidence from biomedical natural language processing.', *Studies in health technology and informatics*, 245, pp. 298–302.

Bohning, G. and Wilkie, A. (1999) 'Palatability scoring of forage plants in central Australia.', *Technote (Darwin)*, (No. 106), p. 7 pp.

Bolton, P.E. *et al.* (2016) 'Three molecular markers show no evidence of population genetic structure in the Gouldian finch (*Erythrura gouldiae*)', *PLoS One*, 11(12), p. e0167723. Available at: https://doi.org/10.1371/journal.pone.0167723.

Bonter, D.N. and Bridge, E.S. (2011) 'Applications of radio frequency identification (RFID) in ornithological research: a review: RFID Applications in Ornithology', *Journal of Field Ornithology*, 82(1), pp. 1–10. Available at: https://doi.org/10.1111/j.1557-9263.2010.00302.x.

Bonthoux, S. *et al.* (2018) 'Spatial and temporal dependency of NDVI satellite imagery in predicting bird diversity over France', *Remote Sensing*, 10(7), p. 1136.

Bowman, D.M. *et al.* (2010) 'Biogeography of the Australian monsoon tropics', *Journal of Biogeography*, 37(2), pp. 201–216.

Boyce, M.S. *et al.* (2002) 'Evaluating resource selection functions', *Ecological Modelling*, 157(2–3), pp. 281–300. Available at: https://doi.org/10.1016/s0304-3800(02)00200-4.

Braby, M.F. (2018) 'Threatened species conservation of invertebrates in Australia: an overview', *Austral Entomology*, 57(2), pp. 173–181. Available at: https://doi.org/10.1111/aen.12324.

Bradie, J. and Leung, B. (2017) 'A quantitative synthesis of the importance of variables used in MaxEnt species distribution models', *Journal of Biogeography*, 44(6), pp. 1344–1361.

Brandt, M.J. and Cresswell, W. (2008) 'Breeding behaviour, home range and habitat selection in Rock Firefinches *Lagonosticta sanguinodorsalis* in the wet and dry season in central Nigeria', *Ibis*, 150(3), pp. 495–507. Available at: https://doi.org/10.1111/j.1474-919x.2008.00811.x.

Brandt, M.J. and Cresswell, W. (2009) 'Diurnal foraging routines in a tropical bird, the rock finch *Lagonosticta sanguinodorsalis*: how important is predation risk?', *Journal of Avian Biology*, 40(1), pp. 90–94. Available at: https://doi.org/10.1111/j.1600-048x.2008.04389.x.

Bravery, J.A. (1970) 'The birds of Atherton shire, Queensland', *Emu-Austral Ornithology*, 70(2), pp. 49–63.

Bravo, C. *et al.* (2016) 'Dietary divergence in the most sexually size-dimorphic bird', *The Auk*, 133(2), pp. 178–197. Available at: https://doi.org/10.1642/AUK-15-206.1.

Brawata, R. and Neeman, T. (2011) 'Is water the key? Dingo management, intraguild interactions and 379 predator distribution around water points in arid Australia', *Wildlife Research*, 38, pp. 426–436. Available at: https://doi.org/10.1071/WR10169.

Brazill-Boast, J. *et al.* (2011) 'Interference from long-tailed finches constrains reproduction in the endangered Gouldian finch', *Journal of Animal Ecology*, 80(1), pp. 39–48.

Brennan, K. (1996) *Flowering and fruiting phenology of native plants in the Alligator Rivers Region with particular reference to the Ranger uranium mine lease area*. Supervising Scientist.

Briga, M. *et al.* (2017) 'Food availability affects adult survival trajectories depending on early developmental conditions', *Proceedings of the Royal Society B: Biological Sciences*, 284(1846), p. 20162287. Available at: https://doi.org/10.1098/rspb.2016.2287.

Brown, J.R. and Carter, J. (1998) 'Spatial and temporal patterns of exotic shrub invasion in an Australian tropical grassland', *Landscape Ecology*, 13, pp. 93–102.

Brown, L.J. and Nocera, J.J. (2017) 'Conservation of breeding grassland birds requires local management strategies when hay maturation and nutritional quality differ among regions', *Agriculture, Ecosystems & Environment*, 237, pp. 242–249. Available at: https://doi.org/10.1016/j.agee.2016.11.004.

Brzęk, P. *et al.* (2010) 'Low plasticity in digestive physiology constrains feeding ecology in diet specialist, zebra finch (Taeniopygia guttata)', *Journal of Experimental Biology*, 213(5), pp. 798–807.

BTFRP (unpublished) *Draft national recovery plan for the southern black-throated finch:* Poephila cincta cincta.

Buckland, S. *et al.* (2015) *Distance Sampling: Methods and Applications*. Cham: Springer International Publishing. Available at: https://doi.org/10.1007/978-3-319-19219-2.

Buda, M., Maki, A. and Mazurowski, M.A. (2018) 'A systematic study of the class imbalance problem in convolutional neural networks', *Neural networks*, 106, pp. 249–259. Available at: https://doi.org/10.1016/j.neunet.2018.07.011.

Bugoni, L., Griffiths, K. and Furness, R. (2011) 'Sex-biased incidental mortality of albatrosses and petrels in longline fisheries: differential distributions at sea or differential access to baits mediated by sexual size dimorphism?', *Journal of Ornithology*, 152(2), pp. 261–268.

Buosi, P. (2011a) *Habitat Management Guidelines for the Black-Throated Finch (Poephila Cincta Cincta) in the Brigalow Belt North Bioregion*. NRA Environmental Consultants and Black-throated Finch Recovery Team.

Buosi, P. (2011b) *Management Guidelines: How to look after the Black-throated Finch*. NRA Environmental Consultants and Black-throated Finch Recovery Team.

Buosi, P. *et al.* (2021) *Southern Black-throated Finch* Poephila cincta cincta. *In 'Action Plan for Australian Birds 2020*. Melbourne: CSIRO.

Bureau of Meteorology (2025) *Climate Data Online: Daily Rainfall Ulcanbah*. Available at: http://www.bom.gov.au/jsp/ncc/cdio/weatherData/av?p_nccObsCode=136&p_display_type=dailyDataFile&p_startYear=&p_c=&p_stn_num=036050 (Accessed: 17 January 2024).

Bureau of Meteorology (BoM) (2025) *Climate Data Online*. Available at: http://www.bom.gov.au/climate/data/.

Bürkner, P.-C. (2017) 'brms: An R package for Bayesian multilevel models using Stan', *Journal of statistical software*, 80, pp. 1–28.

Bürkner, P.-C. and Charpentier, E. (2020) 'Modelling monotonic effects of ordinal predictors in Bayesian regression models', *British Journal of Mathematical and Statistical Psychology*, 73(3), pp. 420–451.

Burley, N. (1981) 'Sex ratio manipulation and selection for attractiveness', *Science*, 211(4483), pp. 721–722.

Burley, N. (1986) 'Sexual Selection for Aesthetic Traits in Species with Biparental Care', *The American Naturalist*, 127(4), pp. 415–445.

Burnham, K.P. and Anderson, D.R. (2004) 'Multimodel Inference: Understanding AIC and BIC in Model Selection', *SOCIOLOGICAL METHODS AND RESEARCH*, 33(2), pp. 261–304.

Burrows, G.E. *et al.* (2010) 'A wide diversity of epicormic structures is present in Myrtaceae species in the northern Australian savanna biome–implications for adaptation to fire', *Australian Journal of Botany*, 58(6), pp. 493–507.

Butler, D.W. and Fairfax, R.J. (2008) 'Buffel Grass and fire in a Gidgee and Brigalow woodland: A case study from central Queensland', *Ecological Management & Restoration*, 4(2). Available at: https://onlinelibrary.wiley.com/doi/full/10.1046/j.1442-8903.2003.00146.x.

Cabitza, F. *et al.* (2020) 'The elephant in the machine: Proposing a new metric of data reliability and its application to a medical case to assess classification reliability', *Applied Sciences*, 10(11), p. 4014. Available at: https://doi.org/10.3390/app10114014.

Calabrese, J.M., Fleming, C.H. and Gurarie, E. (2016) 'ctmm: An R package for analyzing animal relocation data as a continuous-time stochastic process', *Methods in Ecology and Evolution*, 7(9), pp. 1124–1132. Available at: https://doi.org/10.1111/2041-210x.12559.

Camaclang, A.E. *et al.* (2015) 'Current practices in the identification of critical habitat for threatened species', *Conservation Biology*, 29(2), pp. 482–492. Available at: https://doi.org/10.1111/cobi.12428.

Camín, S.R. *et al.* (2015) 'Exploring food preferences and the limits of feeding flexibility of seed-eating desert birds', *Emu-Austral Ornithology*, 115(3), pp. 261–269.

Campagner, A. *et al.* (2021) 'Ground truthing from multi-rater labeling with three-way decision and possibility theory', *Information Sciences*, 545, pp. 771–790. Available at: https://doi.org/10.1016/j.ins.2020.09.049.

Campagner, A., Cabitza, F. and Ciucci, D. (2019) 'Three-way classification: Ambiguity and abstention in machine learning', in. *Rough Sets: International Joint Conference, IJCRS 2019, Debrecen, Hungary, June 17-21, 2019, Proceedings*, Springer, pp. 280–294. Available at: https://doi.org/10.1007/978-3-030-22815-6_22.

Cardoso, J.S. and Sousa, R. (2011) 'Measuring the performance of ordinal classification', *International Journal of Pattern Recognition and Artificial Intelligence*, 25(08), pp. 1173–1195. Available at: https://doi.org/10.1142/s0218001411009093.

Carrillo, C.M. *et al.* (2007) 'Seed selection by the trumpeter finch, Bucanetes githagineus. What currency does this arid-land species value?', in. *Annales Zoologici Fennici*, JSTOR, pp. 377–386.

Carthey, A.J.R. *et al.* (2018) 'Broad-scale patterns in smoke-responsive germination from the south-eastern Australian flora', *Journal of Vegetation Science*, 29(4), pp. 737–745. Available at: https://doi.org/10.1111/jvs.12657.

Catry, P., Phillips, R. and Croxall, J. (2006) 'Sexual segregation in birds: Patterns, processes and implications for conservation', *Sexual Segregation in Vertebrates: Ecology of the Two Sexes*, pp. 351–378. Available at: https://doi.org/10.1017/CBO9780511525629.019.

Cerit, H. and Avanus, K. (2007) 'Sex identification in avian species using DNA typing methods', *World's Poultry Science Journal*, 63(1), pp. 91–100. Available at: https://doi.org/10.1017/S0043933907001316.

Chambert, T. *et al.* (2018) 'A new framework for analysing automated acoustic species detection data: Occupancy estimation and optimization of recordings post-processing', *Methods in Ecology and Evolution*, 9(3), pp. 560–570. Available at: https://doi.org/10.1111/2041-210x.12910.

Chase, M.K. (2002) 'Nest site selection and nest success in a song sparrow population: the significance of spatial variation', *The Condor*, 104(1), pp. 103–116.

Christin, S., Hervet, É. and Lecomte, N. (2019) 'Applications for deep learning in ecology', *Methods in Ecology and Evolution*, 10(10), pp. 1632–1644. Available at: https://doi.org/10.1111/2041-210x.13256.

Clark, R.G. and Shutler, D. (1999) 'Avian Habitat Selection: Pattern from Process in Nest-Site Use by Ducks?', *Ecology*, 80(1), pp. 272–287.

Clarke, S. and French, K. (2005) 'Germination response to heat and smoke of 22 Poaceae species from grassy woodlands', *Australian Journal of Botany*, 53(5), pp. 445–454.

CO2 Australia (2019) *Moray Downs West Offset Area Management Plan*. Brisbane: Prepared for Adani Mining Pty Ltd.

Coates, D. (1996) 'Diet selection by cattle grazing Stylosanthes-grass pastures in the seasonally dry tropics: effect of year, season, stylo species and botanical composition', *Australian Journal of Experimental Agriculture*, 36(7), pp. 781–789.

Collett, Sydney J *et al.* (2022a) 'Sympatric finches differ in visitation patterns to watering holes: implications for site-focused bird counts', *Emu-Austral Ornithology*, 122(3–4), pp. 1–7. Available at: https://doi.org/10.1080/01584197.2022.2104734.

Collett, Sydney J *et al.* (2022b) 'Sympatric finches increase trophic and spatiotemporal partitioning to enable coexistence through periods of low resource availability', *Functional Ecology*, 36(9), pp. 2138–2151. Available at: https://doi.org/10.1111/1365-2435.14111.

Collett, Sydney J. *et al.* (2022) 'Sympatric finches increase trophic and spatiotemporal partitioning to enable coexistence through periods of low resource availability', *Functional Ecology*, 36(9), pp. 2138–2151. Available at: https://doi.org/10.1111/1365-2435.14111.

Collett, S.J. *et al.* (2023) 'The threatened Gouldian finch preferentially forages in prescribed burnt savannas', *International Journal of Wildland Fire*, 32(9), pp. 1277–1290. Available at: https://doi.org/10.1071/WF22185.

Colonna, J.G., Carvalho, J.R. and Rosso, O.A. (2020) 'Estimating ecoacoustic activity in the Amazon rainforest through Information Theory quantifiers', *PLoS One*, 15(7), p. e0229425. Available at: https://doi.org/10.1371/journal.pone.0229425.

Commonwealth of Australia (2023) *National recovery plan for the southern black-throated finch* Poephila cincta cincta *(Draft)*. Canberra: Department of Climate Change, Energy, the Environment and Water.

Cooch, E. and White, G. (2019) 'Program MARK," A gentle introduction"', *http://www. phidot. org/software/mark/docs/book/* [Preprint].

Cowie, R.J. (1977) 'Optimal foraging in great tits (Parus major)', *Nature*, 268(5616), pp. 137–139.

Cowley, R. *et al.* (2014) 'How hot? How often? Getting the fire frequency and timing right for optimal management of woody cover and pasture composition in northern Australian grazed tropical savannas. Kidman Springs Fire Experiment 1993–2013', *The Rangeland Journal*, 36, p. 323. Available at: https://doi.org/10.1071/RJ14030.

Crewe, T.L. *et al.* (2019) 'Detection range of songbirds using a stopover site by automated radio-telemetry', *Journal of Field Ornithology*, 90(2), pp. 176–189. Available at: https://doi.org/10.1111/jofo.12291.

Cropper, S. (1993) *Management of Endangered Plants*. East Melbourne: CSIRO Publishing.

Crowley, G. (2008) 'Cockatoo grass *Alloteropsis semialata* as a keystone species in northern Australia', *Northern Territory Naturalist*, (20), pp. 58–63.

Crowley, G. and Garnett, S. (1999) 'Seeds of the annual grasses *Schizachyrium* spp. as a food resource for tropical granivorous birds', *Australian Journal of Ecology*, 24(3), pp. 208–220. Available at: https://doi.org/10.1046/j.1442-9993.1999.00964.x.

Crowley, G. and Garnett, S. (2001) 'Growth, seed production and effect of defoliation in an early flowering perennial grass, *Alloteropsis semialata* (Poaceae), on Cape York Peninsula, Australia', *Australian Journal of Botany*, 49(6), pp. 735–743. Available at: https://doi.org/10.1071/BT00090.

Crowley, G., Garnett, S. and Shephard, S. (2003) *Management guidelines for golden-shouldered parrot conservation*. Environmental Protection Agency, Queensland Parks and Wildlife Service.

Czenze, Z.J. *et al.* (2020) 'Regularly drinking desert birds have greater evaporative cooling capacity and higher heat tolerance limits than non-drinking species', *Functional Ecology*, 34(8), pp. 1589–1600.

Daly, R.L. and Hodgkinson, K.C. (1996) 'Relationships between grass, shrub and tree cover on four landforms of semi-arid eastern Australia, and prospects for change by burning.', *The Rangeland Journal*, 18(1), pp. 104–117.

Darras, K.F. *et al.* (2020) 'High microphone signal-to-noise ratio enhances acoustic sampling of wildlife', *PeerJ*, 8, p. e9955. Available at: https://doi.org/10.7717/peerj.9955.

Davies, D.S.J.J.F. (1977) 'Man's Activities and Birds' Distribution in the Arid Zone', *Emu - Austral Ornithology* [Preprint]. Available at: https://doi.org/10.1071/MU9770169.

Davies, K.F., Margules, C.R. and Lawrence, J.F. (2004) 'A synergistic effect puts rare, specialized species at greater risk of extinction', *Ecology*, 85(1), pp. 265–271. Available at: https://doi.org/10.1890/03-0110.

Davis, S. *et al.* (2025) 'Recent historical bottlenecks and restricted gene flow in one of the last remaining stronghold populations of the southern black-throated finch (Poephila cincta cincta)', *pre-print* [Preprint]. Available at: https://www.researchsquare.com/article/rs-6780931/latest (Accessed: 28 June 2025).

Davis, S.K. (2005) 'Nest-site selection patterns and the influence of vegetation on nest survival of mixed-grass prairie passerines', *The Condor*, 107(3), pp. 605–616.

Davison, C.W. *et al.* (2023) 'Vegetation structure from LiDAR explains the local richness of birds across Denmark', *Journal of Animal Ecology*, 92(7), pp. 1332–1344.

Day K *et al.* (2019) 'Development and validation of an environmental DNA test for the endangered Gouldian finch', *Endangered Species Research*, 40, pp. 171–182.

De León, L. *et al.* (2014) 'Darwin's finches and their diet niches: the sympatric coexistence of imperfect generalists', *Journal of evolutionary biology*, 27(6), pp. 1093–1104.

Debataraja, I.B., Pradana, D.H. and Winarni, N.L. (2021) 'The impact of landuse and the relationship between NDVI on the bird species richness in Sukmajaya District, Depok', *IOP Conference Series: Earth and Environmental Science*, 846(1). Available at: https://doi.org/10.1088/1755-1315/846/1/012004.

Debus, S. (1995) 'Surveys of large forest owls in northern New South Wales: Methodology, calling behaviour and owl responses', *Corella*, 19(2), pp. 38–50.

Dechaume-Moncharmont, F.X., Monceau, K. and Cezilly, F. (2011) 'Sexing birds using Discriminant Function Analysis: A critical appraisal', *Auk*, 128(1), pp. 78–86.

Dema, T. *et al.* (2020) 'Acoustic detection and acoustic habitat characterisation of the critically endangered white-bellied heron (*Ardea insignis*) in Bhutan', *Freshwater Biology*, 65(1), pp. 153–164. Available at: https://doi.org/10.1111/fwb.13217.

Department of Agriculture, Water and the Environment (DAWE) (2023) 'Draft National Recovery Plan for the southern black-throated finch *Poephila cincta cincta*'. Available at: https://www.dcceew.gov.au/sites/default/files/documents/draft-national-recovery-plan-southern-black-throated-finch.pdf.

Department of Climate Change, Energy, the Environment and Water (2023) 'Draft national recovery plan for the southern black-throated finch *Poephila cincta cincta*'. Australian Government.

Department of Climate Change, Energy, the Environment and Water (2024) *EPBC Act List of Threatened Fauna*. Available at: https://www.environment.gov.au/cgi-bin/sprat/public/publicthreatenedlist.pl (Accessed: 16 January 2024).

Department of Environment and Science (2013a) *WetlandInfo: Alice Tableland biogeographic subregion*. Available at: https://wetlandinfo.des.qld.gov.au/wetlands/facts-maps/biogeographic-subregion-alice-tableland (Accessed: 16 January 2024).

Department of Environment and Science (2013b) *WetlandInfo: Townsville Plains biogeographic subregion*. Available at: https://wetlandinfo.des.qld.gov.au/wetlands/facts-maps/biogeographic-subregion-townsville-plains (Accessed: 16 January 2024).

Department of Environment and Science (2013c) *Wetlands hydro-climate tool*, *WetlandInfo*. Available at: https://wetlandinfo.des.qld.gov.au/wetlands/resources/tools/hydro-climate/.

Department of Environment and Science (2018a) *Regional Ecosystem Technical Descriptions for the Brigalow Belt Bioregion*. Brisbane: Queensland Government.

Department of Environment and Science (2018b) 'Remnant Vegetation Cover 2017'. Department of Environment and Science.

Department of Environment, Science and Innovation (2024) *Environmental Protection Act 1994 Public Register*. Available at: https://apps.des.qld.gov.au/public-register/.

Department of Natural Resources and Mines, Manufacturing and Regional and Rural Development (DNRMMRRD) (2025) 'Vegetation management Regional Ecosystem map Version 13.00'. Brisbane: Department of Natural Resources and Mines, Manufacturing and Regional and Rural Development (DNRMMRRD). Available at: https://qldspatial.information.qld.gov.au/catalogue/custom/detail.page?fid={BCFBDC29-85AF-46B5-85E3-41C5A4B2057A}.

Department of Resources (2025) 'Vegetation management regulated vegetation management map - version 7.03'. Brisbane: Department of Resources. Available at: https://www.data.qld.gov.au/dataset/vegetation-management-act-series.

Department of the Environment, Water, Heritage and the Arts (2009a) *Significant impact guidelines for the endangered black-throated finch (southern) (Poephila cincta cincta)*. Canberra: Australian Government.

Department of the Environment, Water, Heritage and the Arts (2009b) *Significant impact guidelines for the endangered black-throated finch (southern) (Poephila cincta cincta)*. Australian Government.

DeWalt, S.J., Maliakal, S.K. and Denslow, J.S. (2003) 'Changes in vegetation structure and composition along a tropical forest chronosequence: implications for wildlife', *Forest Ecology and Management*, 182(1–3), pp. 139–151.

Dias, R.I. (2019) 'Evolution of Phenotypic Sex Differences in Cooperative Species: is Competition an Opposing Force?', *Acta Ornithologica*, 53(2), pp. 125–142. Available at: https://doi.org/10.3161/00016454AO2018.53.2.003.

Díaz, M. (1990) 'Interspecific patterns of seed selection among granivorous passerines: effects of seed size, seed nutritive value and bird morphology', *Ibis*, 132(3), pp. 467–476. Available at: https://doi.org/10.1111/j.1474-919X.1990.tb01064.x.

Diaz, M. (1990) 'Interspecific patterns of seed selection among granivorous passerines: effects of seed size, seed nutritive value and bird morphology', *IBIS*, 132(3), pp. 349–513.

Díaz, M. (1996) 'Food choice by seed-eating birds in relation to seed chemistry', *Comparative Biochemistry and Physiology -- Part A: Physiology*, 113(3), pp. 239–246. Available at: https://doi.org/10.1016/0300-9629(95)02093-4.

Diaz, M. (1996) 'Food choice by seed-eating birds in relation to seed chemistry', *Comparative Biochemistry and Physiology Part A: Physiology*, 113(3), pp. 239–246.

Diemer, K.M., Wheeler, H.E. and Nocera, J.J. (2014) 'Retention rates of glue-attached radio-transmitters on two small bird species with contrasting life histories', *The Wilson Journal of Ornithology*, 126(1), p. 39. Available at: https://doi.org/10.1676/13-098.1.

Dierschke, J. (2002) 'Food preferences of Shorelarks *Eremophila alpestris*, Snow Buntings *Plectrophenax nivalis* and Twites *Carduelis flavirostris* wintering in the Wadden Sea', *Bird study*, 49(3), pp. 263–269. Available at: https://doi.org/10.1080/00063650209461274.

Doherty, P.F., White, G.C. and Burnham, K.P. (2010) 'Comparison of model building and selection strategies', *Journal of Ornithology*, 152(Suppl 2), pp. 317–323. Available at: https://doi.org/10.1007/s10336-010-0598-5.

Doligez, B. and Boulinier, T. (2008) 'Habitat selection and sabitat suitability preferences', in S.E. Jørgensen and B.D. Fath (eds) *Encyclopedia of ecology*. Oxford: Academic Press, pp. 1810–1830. Available at: https://doi.org/10.1016/B978-008045405-4.00015-X.

Donald, P. (2007) 'Adult sex ratios in wild bird populations', *International journal of avian science*, 149(4), pp. 671–692.

Dostine, P.L. and Franklin, D. (2002) 'A comparison of the diet of three finch species in the Yinberrie Hills area, Northern Territory', *Emu - Austral Ornithology*, 102(2), pp. 159–164. Available at: https://doi.org/10.1071/MU01034.

Douma, J.C. and Weedon, J.T. (2019) 'Analysing continuous proportions in ecology and evolution: A practical introduction to beta and Dirichlet regression', *Methods in Ecology and Evolution*, 10(9), pp. 1412–1430.

Dr Jon Coleman (2020) 'Bird Banding Measurement Suite of Morphology Measurements'.

Duc, P.N.H. *et al.* (2021) 'Assessing inter-annotator agreement from collaborative annotation campaign in marine bioacoustics', *Ecological Informatics*, 61, p. 101185. Available at: https://doi.org/10.1016/j.ecoinf.2020.101185.

Durell, S. (2000) 'Individual feeding specialisation in shorebirds: population consequences and conservation implications', *Biological Reviews*, 75(4), pp. 503–518. Available at: https://doi.org/10.1111/j.1469-185X.2000.tb00053.x.

Durell, S. *et al.* (2008) 'Sex, diet and feeding method-related differences in body condition in the Oystercatcher Haematopus ostralegus', *Ibis*, 143, pp. 107–119. Available at: https://doi.org/10.1111/j.1474-919X.2001.tb04175.x.

E2M (2020) *Black-throated Finch Research Program: 2020 Wet Season Survey*. Brisbane: Prepared for Adani Mining Pty Ltd.

E2M (2021) *Black-throated Finch Research Program: 2021 Early Wet Season Survey*.

E2M (2022) *Black-throated Finch Research Program: 2022 Early Wet Season Survey Report*.

Eaton, M.D. (2005) 'Human vision fails to distinguish widespread sexual dichromatism among sexually "monochromatic" birds', *Proceedings of the National Academy of Sciences*, 102(31), pp. 10942–10946. Available at: https://doi.org/10.1073/pnas.0501891102.

EcoLogical Australia (2020) *Black-throated Finch Research Program*.

EcoLogical Australia (2023) *Black-throated Finch Management Plan - Carmichael Coal Mine and Offsite Infrastructure*. Brisbane: Prepared for Bravus Mining and Resources.

Edwards, A., Russell-Smith, J. and Meyer, M. (2015) 'Contemporary fire regime risks to key ecological assets and processes in north Australian savannas', *International Journal of Wildland Fire*, 24(6), pp. 857–870. Available at: https://doi.org/10.1071/Wf14197.

Eldridge, D.J. *et al.* (2016) 'Ecosystem structure, function, and composition in rangelands are negatively affected by livestock grazing', *Ecological Applications*, 26(4), pp. 1273–1283.

Eldridge, D.J. and Soliveres, S. (2015) 'Are shrubs really a sign of declining ecosystem function? Disentangling the myths and truths of woody encroachment in Australia', *Australian Journal of Botany*, 62(7), pp. 594–608.

Elith, J., Phillips, S.J., Hastie, T., Dudík, M., *et al.* (2011) 'A statistical explanation of MaxEnt for ecologists', *Diversity & distributions*, 17(1), pp. 43–57. Available at: https://doi.org/10.1111/j.1472-4642.2010.00725.x.

Elith, J., Phillips, S.J., Hastie, T., Dudik, M., *et al.* (2011) 'A statistical explanation of MaxEnt for ecologists.(Report)', *Diversity and Distributions*, 17(1), p. 43. Available at: https://doi.org/10.1111/j.1472-4642.2010.00725.x.

Ellis, E.C. *et al.* (2010) 'Anthropogenic transformation of the biomes, 1700 to 2000', *Global ecology and biogeography*, 19(5), pp. 589–606. Available at: https://doi.org/10.1111/j.1466-8238.2010.00540.x.

Endo, S. (2012) 'Nest-Site Characteristics Affect Probability of Nest Predation of Bull-Headed Shrikes', *The Wilson Journal of Ornithology*, 124(3), pp. 513–517. Available at: https://doi.org/10.1676/11-164.1.

European Space Agency (2023) 'Copernicus Sentinel-2'. Copernicus Open Access Hub. Available at: https://scihub.copernicus.eu/.

Evans, S. *et al.* (1985) 'Patterns of drinking behaviour of some Australian estrildine finches', *Ibis*, 127(3), pp. 348–354. Available at: https://doi.org/10.1111/j.1474-919x.1985.tb05076.x.

Evans, S.M. *et al.* (1985) 'Patterns of drinking behaviour of some Australian estrildine finches', *Ibis*, 127(3), pp. 348–354. Available at: https://doi.org/10.1111/j.1474-919X.1985.tb05076.x.

Eyre, T.J. *et al.* (2009) 'Buffel grass in Queensland's semi-arid woodlands: Response to local and landscape scale variables, and relationship with grass, forb and reptile species', *The Rangeland Journal*, 31(3), pp. 293–305. Available at: https://doi.org/10.1071/RJ08035.

Eyre, T.J. *et al.* (2015) *BioCondition: A Condition Assessment Framework for Terrestrial Biodiversity in Queensland. Assessment Manual Version 2.2*. Brisbane: Queensland Herbarium, Department of Science, Information Technology, Innovation and Arts.

Eyre, T.J. *et al.* (2022) *Terrestrial Vertebrate Fauna Survey Assessment Guidelines for Queensland (V. 4.0)*. Queensland Government, Brisbane: Department of Environment and Science.

Fairbairn, D. (2013) *Fairbairn, Daphne J. 2013. Odd Couples. Extraordinary Differences between the Sexes in the Animal Kingdom. Princeton University Press*.

Fairbrass, A.J. *et al.* (2019) 'CityNet—Deep learning tools for urban ecoacoustic assessment', *Methods in Ecology and Evolution*, 10(2), pp. 186–197. Available at: https://doi.org/10.1111/2041-210x.13114.

Fan, X. *et al.* (2020) 'Compositing the minimum NDVI for daily water surface mapping', *Remote Sensing*, 12(4), p. 700.

Farrell, H.L. and Gornish, E.S. (2019) 'Pennisetum ciliare: a review of treatment efficacy, competitive traits, and restoration opportunities', *Invasive plant science and management*, 12(4), pp. 203–213. Available at: https://doi.org/10.1017/inp.2019.28.

Favretto, M.A. *et al.* (2024) 'Habitat selection in many-colored rush tyrant (Tachuris rubrigastra) and wren-like Rushbird (Phleocryptes melanops) in the subtropical salt marshes of Brazil', *Studies on Neotropical Fauna and Environment*, 59(2), pp. 276–288.

Fensham, R. (2012) 'Fire regimes in Australian tropical savanna: perspectives, paradigms and paradoxes', *Flammable Australia: Fire regimes, biodiversity and ecosystems in a changing world*, pp. 173–193.

Fensham, R., Fairfax, R. and Archer, S. (2005) 'Rainfall, land use and woody vegetation cover change in semi-arid Australian savanna', *Journal of Ecology*, 93(3), pp. 596–606.

Fensham, R.J. *et al.* (2013) 'Propagule pressure, not fire or cattle grazing, promotes invasion of buffel grass Cenchrus ciliaris', *The Journal of applied ecology*, 50(1), pp. 138–146. Available at: https://doi.org/10.1111/1365-2664.12009.

Fensham, R.J. and Fairfax, R.J. (2008) 'Water-remoteness for grazing relief in Australian arid-lands', *Biological conservation*, 141(6), pp. 1447–1460. Available at: https://doi.org/10.1016/j.biocon.2008.03.016.

Fensham, R.J., Fairfax, R.J. and Dwyer, J.M. (2010) 'Vegetation responses to the first 20 years of cattle grazing in an Australian desert', *Ecology*, 91(3), pp. 681–692. Available at: https://doi.org/10.1890/08-2356.1.

Fensham, R.J., Silcock, J.L. and Dwyer, J.M. (2011) 'Plant species richness responses to grazing protection and degradation history in a low productivity landscape', *Journal of Vegetation Science*, 22(6), pp. 997–1008. Available at: https://doi.org/10.1111/j.1654-1103.2011.01305.x.

Fensham, R.J. and Skull, S.D. (1999) 'Before Cattle: A Comparative Floristic Study of Eucalyptus Savanna Grazed by Macropods and Cattle in North Queensland, Australia', *Biotropica*, 31(1), pp. 37–47. Available at: https://doi.org/10.1111/j.1744-7429.1999.tb00115.x.

Fensham, R.J., Wang, J. and Kilgour, C. (2015) 'The relative impacts of grazing, fire and invasion by buffel grass (Cenchrus ciliaris) on the floristic composition of a rangeland savanna ecosystem', *The Rangeland Journal*, 37(3), pp. 227–237.

Ferger, S.W. *et al.* (2014) 'Food resources and vegetation structure mediate climatic effects on species richness of birds', *Global ecology and biogeography*, 23(5), pp. 541–549.

Fernández-Tizón, M. *et al.* (2020) 'Arthropod biomass increase in spring correlates with NDVI in grassland habitat', *The Science of Nature : Naturwissenschaften*, 107(5). Available at: https://doi.org/10.1007/s00114-020-01698-7.

Fisher, C.D., Lindgren, E. and Dawson, W.R. (1972) 'Drinking Patterns and Behavior of Australian Desert Birds in Relation to Their Ecology and Abundance on JSTOR', *The Condor*, 74(2), pp. 111–136. Available at: https://doi.org/10.2307/1366276.

Fisher, K.E. *et al.* (2021) 'Locating large insects using automated VHF radio telemetry with a multi-antennae array', *Methods in Ecology and Evolution*, 12(3), pp. 494–506. Available at: https://doi.org/10.1111/2041-210x.13529.

Fisher, R.A. (1930) *The genetical theory of natural selection.* Oxford University Press.

Fleming, C.H. *et al.* (2019) 'Overcoming the challenge of small effective sample sizes in home-range estimation', *Methods in Ecology and Evolution*, 10(10), pp. 1679–1689. Available at: https://doi.org/10.1111/2041-210x.13270.

Fleming, C.H. and Calabrese, J.M. (2017) 'A new kernel density estimator for accurate home-range and species-range area estimation', *Methods in Ecology and Evolution*, 8(5), pp. 571–579. Available at: https://doi.org/10.1111/2041-210x.12673.

Flowers, M. *et al.* (2008) 'Exploring remote sensing applications in perennial ryegrass seed production', *SEED PRODUCTION RESEARCH*, p. 1.

Ford, J. (1986a) 'Avian hybridization and allopatry in the region of the Einasleigh Uplands and Burdekin-Lynd Divide, north-eastern Queensland', *Emu-Austral Ornithology*, 86(2), pp. 87–110. Available at: https://doi.org/10.1071/mu9860087.

Ford, J. (1986b) 'Avian hybridization and allopatry in the region of the Einasleigh Uplands and Burdekin-Lynd Divide, north-eastern Queensland', *Emu-Austral Ornithology*, 86(2), pp. 87–110.

Fox *et al.* (2001) *Fox, I. D., Neldner, V. J., Wilson, G. W., and Bannink, P. J. (2001). The Vegetation of the Australian Tropical Savannas. Environmental Protection Agency, Brisbane.*

Frankham, R. (1995) 'Effective population size/adult population size ratios in wildlife: a review', *Genetics Research*, 66(2), pp. 95–107.

Frankham, R. (2005) 'Genetics and extinction.', *Biological Conservation*, 126(2), pp. 131–140.

Frankham, R. (2010a) 'Challenges and opportunities of genetic approaches to biological conservation', *Biological conservation*, 143(9), pp. 1919–1927.

Frankham, R. (2010b) 'Inbreeding in the wild really does matter', *Heredity*, 104(124). Available at: https://www.nature.com/articles/hdy2009155.

Frankham, R. *et al.* (2017) 'Population fragmentation causes inadequate gene flow and increases extinction risk.', *A Practical Guide for Genetic Management of Fragmented Animal and Plant Populations*, pp. 87–112. Available at: https://doi.org/10.1093/oso/9780198783398.003.0005.

Frankham, R., Bradshaw, C.J.A. and Brook, B.W. (2014) 'Genetics in conservation management: Revised recommendations for the 50/500 rules, Red List criteria and population viability analyses', *Biological Conservation*, 170, pp. 56–63. Available at: https://doi.org/10.1016/j.biocon.2013.12.036.

Frankham, R., Briscoe, A. and Ballou, J. (2002) *Introduction to Conservation Genetics*. Cambridge, UK: University of Cambridge Press.

Franklin, D. (1999) 'Evidence of disarray amongst granivorous bird assemblages in the savannas of northern Australia, a region of sparse human settlement', *Biological Conservation*, 90(1), pp. 53–68. Available at: https://doi.org/10.1016/S0006-3207(99)00010-5.

Franklin, D. *et al.* (2005) 'Geographic patterns and correlates of the decline of granivorous birds in northern Australia', *Wildlife Research*, 32(5), pp. 399–408. Available at: https://doi.org/10.1071/WR05052.

Franklin, D. *et al.* (2017) 'Wings of tropical finches: interspecific differences in shape are consistent with levels of mobility, but moult and feather fault patterns are more complex', *Emu-Austral Ornithology*, 117(4), pp. 370–381. Available at: https://doi.org/10.1080/01584197.2017.1361790.

Franklin, D.C. (1999) 'Evidence of disarray amongst granivorous bird assemblages in the savannas of northern Australia, a region of sparse human settlement', *Biological Conservation*, 90(1), pp. 53–68. Available at: https://doi.org/Doi%252010.1016/S0006-3207(99)00010-5.

Franklin, D.C. *et al.* (2005) 'Geographic patterns and correlates of the decline of granivorous birds in northern Australia', *Wildlife Research*, 32(5), pp. 399–408. Available at: https://doi.org/10.1071/Wr05052.

Franklin, D.C., Dostine, P.L. and Tidemann, S.A. (1998) 'Post-juvenile moult strategies of co-existing Gouldian, Long-tailed and Masked Finches', *Corella*, 22, pp. 73–79.

Franklin, D.C., Woinarski, J.C.Z. and Noske, R.A. (2000) 'Geographical patterning of species richness among granivorous birds in Australia', *Journal of Biogeography*, 27(4), pp. 829–842. Available at: https://doi.org/DOI%252010.1046/j.1365-2699.2000.00446.x.

Fraser, B.F. *et al.* (2003) 'Fire management experiment for the declining Partridge Pigeon, Kakadu National Park', *Ecological Management & Restoration*, 4(2), pp. 94–102. Available at: https://doi.org/10.1046/j.1442-8903.2003.00142.x.

Fraser, K.C. *et al.* (2018) 'Tracking the conservation promise of movement ecology', *Frontiers in Ecology and Evolution*, 6, p. 150. Available at: https://doi.org/10.3389/fevo.2018.00150.

Frens, K.M. (2010) 'Effects of food type and patch location on foraging: a field test of optimal foraging predictions'.

Frere, C. *et al.* (2023) *Koalas, friends, and foes – the application of airborne eDNA for the biomonitoring of threatened species*. The University of Queensland. Available at: https://www.authorea.com/users/466413/articles/632189-koalas-friends-and-foes-the-application-of-airborne-edna-for-the-biomonitoring-of-threatened-species.

Friedman, J. (2020) 'The Evolution of Annual and Perennial Plant Life Histories: Ecological Correlates and Genetic Mechanisms', *Annual Review of Ecology, Evolution, and Systematics*, 51(1), pp. 461–481. Available at: https://doi.org/10.1146/annurev-ecolsys-110218-024638.

Friendly, M. and Fox, J. (2024) *candisc: Visualizing Generalized Canonical Discriminant and Canonical Correlation Analysis*. Available at: https://CRAN.R-project.org/package=heplots.

Funghi, C. *et al.* (2020) 'Estimating food resource availability in arid environments with Sentinel 2 satellite imagery', *PeerJ*, 8, p. e9209.

Garabedian, J.E. *et al.* (2020) 'Effects of group size and group density on trade-offs in resource selection by a group-territorial central-place foraging woodpecker', *Ibis*, 162(2), pp. 477–491. Available at: https://doi.org/10.1111/ibi.12733.

García, D., Zamora, R. and Amico, G.C. (2011) 'The spatial scale of plant–animal interactions: effects of resource availability and habitat structure', *Ecological Monographs*, 81(1), pp. 103–121.

Gardener, C. (1993) 'The colonization of a tropical grassland by Stylosanthes from seed transported in cattle faeces', *Australian Journal of Agricultural Research*, 44(2), pp. 299–315.

Garnett, S., Baker, G.B. and ProQuest, E. (2021) *The Action Plan for Australian Birds 2020*. Victoria: CSIRO Publishing (Book, Whole).

Garnett, S. and Crowley, G. (1994) 'Wet-season feeding by four species of granivorous birds in the Northern Territory', *Australian Bird Watcher*, 15(7), pp. 306–309.

Garnett, S. and Crowley, G. (1995) 'Feeding ecology of hooded parrots *Psephotus dissimilis* during the early wet season', *Emu*, 95(1), pp. 54–61.

Garnett, S., Szabo, J. and Dutson, G. (2011) *The Action Plan for Australian Birds*. Melbourne: CSIRO Publishing.

Garnett, S.T. and Baker, G. (2020) *The action plan for Australian birds 2020*. Collingwood: CSIRO publishing. Available at: https://doi.org/10.1071/9781486311910.

Geiger, F. *et al.* (2014) 'Habitat use and diet of Skylarks (*Alauda arvensis*) wintering in an intensive agricultural landscape of the Netherlands', *Journal of Ornithology*, 155(2), pp. 507–518. Available at: https://doi.org/10.1007/s10336-013-1033-5.

Gelman, A., Hill, J. and Vehtari, A. (2020) *Regression and other stories*. Cambridge University Press. Available at: https://doi.org/10.1017/9781139161879.002.

GHD (2012) *Report on Carmiachael Coal Mine and Rail Project, mine technical report: Terrestrial ecology report 23244-D-RP-0024*. Brisbane: Prepared for Adani Mining Pty Ltd.

GHD (2013) *Carmichael Coal Mine and Rail Project SEIS: Report for updated mine ecology*. Brisbane: Prepared for Adani Mining Pty Ltd by GHD Pty Ltd.

Gibb, R. *et al.* (2019) 'Emerging opportunities and challenges for passive acoustics in ecological assessment and monitoring', *Methods in Ecology and Evolution*, 10(2), pp. 169–185. Available at: https://doi.org/10.1111/2041-210x.13101.

GLÜCK, E.E. (1985) 'Seed preference and energy intake of Goldfinches Carduelis carduelis in the breeding season', *Ibis*, 127(4), pp. 421–429.

Gluckman, T.-L. (2014) 'Pathways to elaboration of sexual dimorphism in bird plumage patterns', *Biological Journal of the Linnean Society*, 111(2), pp. 262–273. Available at: https://doi.org/10.1111/bij.12211.

Goodale, E. *et al.* (2015) 'The structure of mixed-species bird flocks, and their response to anthropogenic disturbance, with special reference to East Asia', *Avian Research*, 6(1), p. 14. Available at: https://doi.org/10.1186/s40657-015-0023-0.

Gottwald, J. *et al.* (2019) 'Introduction of an automatic and open-source radio-tracking system for small animals', *Methods in Ecology and Evolution*, 10(12), pp. 2163–2172. Available at: https://doi.org/10.1111/2041-210X.13294.

Gottwald, J. *et al.* (2023) 'Classifying the activity states of small vertebrates using automated VHF telemetry', *Methods in ecology and evolution*, 14(1), pp. 252–264. Available at: https://doi.org/10.1111/2041-210x.14037.

Goudet, J., Perrin, N. and Waser, P. (2002) 'Tests for sex-biased dispersal using bi-parentally inherited genetic markers.', *Molecular ecology*, 11(6), pp. 1103–1114.

Grant, P. (1986) *Ecology and evolution of Darwin's finches*. Princeton, NJ: Princeton University Press.

Green, S.E. *et al.* (2023) 'Camera trapping with photos and videos: implications for ecology and citizen science', *Remote Sensing in Ecology and Conservation*, 9(2), pp. 268–283.

Greenbaum, G. *et al.* (2014) 'Allelic richness following population founding events - a stochastic modeling framework incorporating gene flow and genetic drift.', *PLoS ONE*, 9(12), pp. 115–203.

Gregorič, M. *et al.* (2022) 'Spider webs as eDNA samplers: Biodiversity assessment across the tree of life', *Molecular Ecology*, 22(7), pp. 2534–2545.

Grey, M.J. (1996) 'An experimental investigation of the role of noisy miners (Manorina melanocephala) in rural woodlands', *Conservation Outside Nature Reserves* [Preprint].

Grice, A.C. *et al.* (2023) 'Surveys of a small population of the Endangered Black-throated Finch "Poephila cincta cincta" in North Queensland', *Australian Field Ornithology*, 40, pp. 152–164.

Griffin, A.S. *et al.* (2020) 'A large-scale automated radio telemetry network for monitoring movements of terrestrial wildlife in Australia', *Australian Zoologist*, 40(3), pp. 379–391. Available at: https://doi.org/10.7882/AZ.2019.026.

Griffiths, R. *et al.* (1998) 'A DNA test to sex most birds', *Molecular ecology*, 7(8), pp. 1071–1075.

Griffiths, R. (2000) 'Sex identification in birds', *Seminars in Avian and Exotic Pet Medicine - SEMIN AVIAN EXOTIC PET MED*, 9, pp. 14–26. Available at: https://doi.org/10.1016/S1055-937X(00)80012-2.

Groves, R. and Whalley, R. (2002) 'Grass and grassland ecology in Australia', *Flora of Australia*, p. 157.

Grubb, T.C. and Greenwald, L. (1982) 'Sparrows and a brushpile - foraging responses to different combinations of predation risk and energy-cost', *Animal Behaviour*, 30(Aug), pp. 637–640. Available at: https://doi.org/Doi%252010.1016/S0003-3472(82)80133-4.

Grüebler, M.U. *et al.* (2008) 'Female biased mortality caused by anthropogenic nest loss contributes to population decline and adult sex ratio of a meadow bird', *Biological Conservation*, 141(12), pp. 3040–3049. Available at: https://doi.org/10.1016/j.biocon.2008.09.008.

Grundel, R. (1992) 'How the mountain chickadee procures more food in less time for its nestlings', *Behavioral Ecology and Sociobiology*, 31, pp. 291–300.

Gutenkunst, R.N. *et al.* (2009) 'Inferring the joint demographic history of multiple populations from multidimensional SNP frequency data.', *PLoS genetics*, 5(10).

Gwet, K.L. (2014) *Handbook of inter-rater reliability: The definitive guide to measuring the extent of agreement among raters*. 4th edn. Advanced Analytics, LLC.

Haldane, J. (1937) 'The effect of variation of fitness.', *The American Naturalist*, 71(735), pp. 337–349.

Hale, R. *et al.* (2020) 'Harnessing knowledge of animal behavior to improve habitat restoration outcomes', *Ecosphere*, 11(4), p. e03104.

Hall, L.S. and Mannan, R.W. (1999) 'Multiscaled Habitat Selection by Elegant Trogons in Southeastern Arizona', *The Journal of Wildlife Management*, 63(2), pp. 451–461.

Hall, T.J. *et al.* (2017) 'Grazing pressure impacts on two Aristida/Bothriochloa native pasture communities of central Queensland', *The Rangeland Journal*, 39(3), pp. 227–243.

Hall, T.J. and Douglas, J.R. (2005) 'Cattle grazing management effects on pasture composition in semi-arid woodlands', in *XX International Grassland Conference: Offered papers*. Wageningen Academic, pp. 542–542.

Harbicht, A.B. *et al.* (2017) 'Novel, continuous monitoring of fine-scale movement using fixed-position radiotelemetry arrays and random forest location fingerprinting', *Methods in Ecology and Evolution*, 8(7), pp. 850–859. Available at: https://doi.org/10.1111/2041-210x.12745.

Harrington, R. (2002) *The effects of artificial watering points on the distribution and abundance of avifauna in an arid and semi-arid mallee environment*. PhD Thesis. The University of Melbourne. Available at: https://rest.neptune-prod.its.unimelb.edu.au/server/api/core/bitstreams/6e60afcc-1a8c-5260-961f-67ab1bba8b25/content (Accessed: 21 November 2023).

Hartig, F. (2022) *DHARMa: Residual diagnostics for hierarchical (multi-level / mixed) regression models (version 0.4.6)*. Available at: https://cran.r-project.org/web/packages/DHARMa/index.html.

Haskell, J.P., Ritchie, M.E. and Olff, H. (2002) 'Fractal geometry predicts varying body size scaling relationships for mammal and bird home ranges', *Nature*, 418(6897), pp. 527–530. Available at: https://doi.org/10.1038/nature00840.

Hayes, A.F. and Krippendorff, K. (2007) 'Answering the call for a standard reliability measure for coding data', *Communication Methods and Measures*, 1(1), pp. 77–89. Available at: https://doi.org/10.1080/19312450709336664.

Hayslette, S.E. and Mirarchi, R.E. (2002) 'Foraging-patch use and within-patch diet selectivity in mourning doves, *Zenaida macroura*', *Ecology*, 83(9), pp. 2637–2641.

He, K. *et al.* (2016) 'Deep residual learning for image recognition', in *Proceedings of the IEEE conference on computer vision and pattern recognition*, pp. 770–778. Available at: https://doi.org/10.1109/cvpr.2016.90.

Healy, A., Tulloch, A.I.T. and Fensham, R.J. (2020) 'Dryland communities find little refuge from grazing due to long-term changes in water availability', *Journal of Arid Environments*, 176. Available at: https://doi.org/10.1016/j.jaridenv.2019.104098.

Hedrick, A. and Temeles, E. (1989) 'The Evolution of Sexual Dimorphism in Animals: Hypotheses and Tests', *Trends in Ecology & Evolution*, 4(5), pp. 136–138.

Heinsohn, R. *et al.* (2019) 'Sex ratio bias and shared paternity reduce individual fitness and population viability in a critically endangered parrot', *Journal of Animal Ecology*, 88, pp. 502–510.

Heinze, G. and Schemper, M. (2002) 'A solution to the problem of separation in logistic regression', *Statistics in medicine*, 21(16), pp. 2409–2419. Available at: https://doi.org/10.1002/sim.1047.

Hervás, M., Alsina-Pagès, Rosa M., *et al.* (2017) 'An FPGA-Based WASN for Remote Real-Time Monitoring of Endangered Species: A Case Study on the Birdsong Recognition of Botaurus stellaris', *Sensors*, 17(6). Available at: https://doi.org/10.3390/s17061331.

Hervás, M., Alsina-Pagès, Rosa Ma, *et al.* (2017) 'An FPGA-based WASN for remote real-time monitoring of endangered species: A case study on the birdsong recognition of *Botaurus stellaris*', *Sensors*, 17(6), p. 1331. Available at: https://doi.org/10.3390/s17061331.

Higgins, P. (2006) *Handbook of Australian, New Zealand and Antarctic Birds: Volume 7 Boatbill to Starlings: Part B Dunnock to Starlings*. Oxford University Press.

Higgins, P.J. and Cowling, S. (2007) *Handbook of Australian, New Zealand & Antarctic Birds*. Oxford University Press.

Higgins, P.J., Peter, J.M. and Cowling, S.J. (2006) *Handbook of Australian, New Zealand and Antarctic Birds*. Melbourne: Oxford University Press (Boatbill to Starlings). Available at: https://doi.org/10.1676/1559-4491(2007)119[516:hoanza]2.0.co;2.

Hill, A.P. *et al.* (2019) 'AudioMoth: A low-cost acoustic device for monitoring biodiversity and the environment', *HardwareX*, 6. Available at: https://doi.org/10.1016/j.ohx.2019.e00073.

Hines, H.B. *et al.* (2021) 'Post-fire assessment report—Natural values: 2019 bushfire complex, Main Range National Park and adjacent QPWS estate, South West Queensland Region', *Brisbane: Queensland Government* [Preprint].

Hoban, S. *et al.* (2022) 'Global genetic diversity status and trends: towards a suite of Essential Biodiversity Variables (EBVs) for genetic composition.', *Biological Reviews*, 97(4), pp. 1511–1538.

Höchst, J. *et al.* (2021) 'tRackIT OS: Open-source software for reliable VHF wildlife tracking', in *Jahrestagung der Gesellschaft Informatik, Digitale Kulturen*. Berlin, Germany: Informatik.

Hodder, G.V. (2020) *The impact of habitat alteration on the population dynamics of a declining woodland bird in the Mount Lofty Ranges*. PhD Thesis. The University of Adelaide.

Hodges, J.A. *et al.* (2021) 'Smoke and heat accelerate and increase germination in fire-prone temperate grassy ecosystems', *Ecosphere*, 12(12), p. e03851. Available at: https://doi.org/10.1002/ecs2.3851.

Hodgkinson, K.C. *et al.* (1989) 'Comparative responses of the Savanna grasses Cenchrus ciliaris and Themeda triandra to defoliation', *Oecologia*, 79(1), pp. 45–52. Available at: https://doi.org/10.1007/BF00378238.

Holway, D.A. (1991) 'Nest-Site Selection and the Importance of Nest Concealment in the Black-Throated Blue Warbler', *The Condor*, 93(3), pp. 575–581. Available at: https://doi.org/10.2307/1368189.

Honarmand, M., Goymann, W. and Naguib, M. (2010) 'Stressful dieting: nutritional conditions but not compensatory growth elevate corticosterone levels in zebra finch nestlings and fledglings', *PLoS One*, 5(9), p. e12930.

Hooper, D.M. *et al.* (2025) 'Recent divergence and microgeographic genetic structure in an endangered Australian songbird: the southern black-throated finch', *bioRxiv*, pp. 2025–04.

Hosseini-Moosavi, S.M. *et al.* (2017) 'Nest-Site Selection and Breeding Success of the Semi-Desert Bird, Grey Hypocolius Hypocolius Ampelinus, in Relation to Plant Structure of Ziziphus Nummularia', *Avian Biology Research*, 10(3), pp. 181–189. Available at: https://doi.org/10.3184/175815617X14981258846760.

House, J.I. and Hall, D.O. (2001) 'Productivity of tropical savannas and grasslands', *Terrestrial global productivity*, 33, pp. 363–400.

Howard, S.D. and Bickford, D.P. (2014) 'Amphibians over the edge: silent extinction risk of Data Deficient species', *Diversity and distributions*, 20(7), pp. 837–846. Available at: https://doi.org/10.1111/ddi.12218.

Huang, X. *et al.* (2023) 'dadi-cli: Automated and distributed population genetic model inference from allele frequency spectra.', *bioRxiv* [Preprint]. Available at: https://doi.org/10.1101/2023.06.15.545182.

Hui, N.T. *et al.* (2021) 'A more precise way to localize animals using drones', *Journal of Field Robotics*, 38(6), pp. 917–928. Available at: https://doi.org/10.1002/rob.22017.

Hüllermeier, E. and Waegeman, W. (2021) 'Aleatoric and epistemic uncertainty in machine learning: An introduction to concepts and methods', *Machine Learning*, 110, pp. 457–506. Available at: https://doi.org/10.1007/s10994-021-05946-3.

Humbert, J.-Y. *et al.* (2009) 'A better way to estimate population trends', *Oikos*, 118(12), pp. 1940–1946. Available at: https://doi.org/10.1111/j.1600-0706.2009.17839.x.

Hunt, L.P. *et al.* (2014) 'Principles and guidelines for managing cattle grazing in the grazing lands of northern Australia: stocking rates, pasture resting, prescribed fire, paddock size and water points–a review', *The Rangeland Journal*, 36(2), pp. 105–119.

Hutschenreiter, A. *et al.* (2024) 'How to count bird calls? Vocal activity indices may provide different insights into bird abundance and behaviour depending on species traits', *Methods in Ecology and Evolution*, 15(6), pp. 1071–1083. Available at: https://doi.org/10.1111/2041-210X.14333.

Iacchei, M. *et al.* (2013) 'Combined analyses of kinship and FST suggest potential drivers of chaotic genetic patchiness in high gene-flow populations.', *Molecular ecology*, 22(13), pp. 3476–3494.

Isles, N. (2007) *Patterns of foraging behaviour of the black-throated finch during breeding season*. BSc (Honours) Thesis. James Cook University.

Ivosevic, B. *et al.* (2015) 'The use of conservation drones in ecology and wildlife research', *Journal of Ecology and Environment*, 38(1), pp. 113–118. Available at: https://doi.org/10.5141/ecoenv.2015.012.

Jackson, J. (2004) *Impacts and management of Cenchrus ciliaris (buffel grass) as an invasive species in northern Queensland*. PhD Thesis. James Cook University. Available at: http://researchonline.jcu.edu.au/1114/ (Accessed: 19 February 2025).

Jacobs, J. (1974) 'Quantitative measurement of food selection: a modification of the forage ratio and Ivlev's electivity index', *Oecologia*, 14, pp. 413–417.

James, C.D., Landsberg, J. and Morton, S.R. (1999) 'Provision of watering points in the Australian arid zone: a review of effects on biota', *Journal of arid environments*, 41(1), pp. 87–121. Available at: https://doi.org/10.1006/jare.1998.0467.

Janicke, T. and Fromonteil, S. (2021) 'Sexual selection and sexual size dimorphism in animals', *Biology Letters*, 17.

Jeltsch, F. *et al.* (2013) 'Integrating movement ecology with biodiversity research-exploring new avenues to address spatiotemporal biodiversity dynamics', *Movement Ecology*, 1(1), pp. 1–13. Available at: https://doi.org/10.1186/2051-3933-1-6.

Jenkins, S.H. (1981) 'Common patterns in home range-body size relationships of birds and mammals', *The American Naturalist*, 118(1), pp. 126–128. Available at: https://doi.org/10.1086/283807.

Jennings, W. and Edwards, S. (2005) 'Speciational history of Australian grass finches (Poephila) inferred from thirty gene trees', *EVOLUTION*, 59(9), pp. 2033–2047.

Jia, B. *et al.* (2022) 'TTSL: An indoor localization method based on Temporal Convolutional Network using time-series RSSI', *Computer Communications*, 193, pp. 293–301. Available at: https://doi.org/10.1016/j.comcom.2022.07.003.

Johnson, D.H. (1980) 'The Comparison of Usage and Availability Measurements for Evaluating Resource Preference', *Ecology*, 61(1), pp. 65–71. Available at: https://doi.org/10.2307/1937156.

Johnson, G.D., Pebworth, J.L. and Krueger, H.O. (1991) 'Retention of transmitters attached to passerines using a glue-on technique', *Journal of Field Ornithology*, 62(4), pp. 486–491. Available at: https://doi.org/10.1676/13-098.1.

Johnson, M.D. (2007) 'Measuring habitat quality: A review', *The Condor*, 109(3), pp. 489–504.

Jombart, T. (2008) ‘adegenet: a R package for the multivariate analysis of genetic markers’, *Bioinformatics*, 24(11), pp. 1403–1405.

Jombart, T. and Ahmed, I. (2011) ‘adegenet 1.3-1: new tools for the analysis of genome-wide SNP data.’, *Bioinformatics*, 27(21), pp. 3070–3071.

Jones, E.A.L., Contreras, D.J. and Everman, W.J. (2021) ‘Chapter 9 - Digitaria ciliaris, Digitaria ischaemum, and Digitaria sanguinalis’, in B.S. Chauhan (ed.) *Biology and Management of Problematic Crop Weed Species*. Academic Press, pp. 173–195. Available at: https://doi.org/10.1016/B978-0-12-822917-0.00014-8.

Jones, H.H., Walters, M.J. and Robinson, S.K. (2020) ‘Do similar foragers flock together? Nonbreeding foraging behavior and its impact on mixed-species flocking associations in a subtropical region’, *The Auk*, 137(2), p. ukz079. Available at: https://doi.org/10.1093/auk/ukz079.

Jones, K.A., Krebs, J.R. and Whittingham, M.J. (2006) ‘Interaction between seed crypsis and habitat structure influence patch choice in a granivorous bird, the chaffinch *Fringilla coelebs*’, *Journal of Avian Biology*, 37(5), pp. 413–418. Available at: https://doi.org/10.1111/j.2006.0908-8857.03883.x.

Jones, P., Aisthorpe, J. and Chilcott, C. (2006) ‘Developing long term carrying capacity models for the Desert Uplands’, in *Proceedings of the Australian Rangeland Society Biennial Conference*. *Australian Rangeland Society 14th Biennial Conference*, Renmark, South Australia: The Australian Rangeland Society.

Juhl, S. (2021) ‘spfilteR: An R package for semiparametric spatial filtering with eigenvectors in (generalized) linear models’. Available at: https://digitalcommons.unl.edu/r-journal/80/ (Accessed: 11 March 2025).

Kahl, S. *et al.* (2021) ‘BirdNET: A deep learning solution for avian diversity monitoring’, *Ecological Informatics*, 61, p. 101236. Available at: https://doi.org/10.1016/j.ecoinf.2021.101236.

Karimi, D. *et al.* (2020) ‘Deep learning with noisy labels: Exploring techniques and remedies in medical image analysis’, *Medical Image Analysis*, 65, p. 101759. Available at: https://doi.org/10.1016/j.media.2020.101759.

Katzner, T.E. and Arlettaz, R. (2020) ‘Evaluating contributions of recent tracking-based animal movement ecology to conservation management’, *Frontiers in Ecology and Evolution*, 7, p. 519. Available at: https://doi.org/10.3389/fevo.2019.00519.

Kays, R. *et al.* (2011) ‘Tracking animal location and activity with an automated radio telemetry system in a tropical rainforest’, *The Computer Journal*, 54(12), pp. 1931–1948. Available at: https://doi.org/10.1093/comjnl/bxr072.

Kearns, A.M., Joseph, L. and Cook, L.G. (2010) ‘The impact of Pleistocene changes of climate and landscape on Australian birds: a test using the Pied Butcherbird (Cracticus nigrogularis).’, *Emu-Austral Ornithology*, 110(4), pp. 285–295.

Keating, J.F. *et al.* (1992) ‘Role of handling time in selection of extruded food morsels by two granivorous bird species’, *The Auk*, 109(4), pp. 863–868.

Keenan, K. *et al.* (2013) ‘diveRsity: an R package for the estimation and exploration of population genetics parameters and their associated errors.’, *Methods in Ecology and Evolution*, 4(8), pp. 782–788.

Kemp, J. and Kutt, A. (2020) ‘Vegetation change 10 years after cattle removal in a savanna landscape’, *The Rangeland Journal*, 42. Available at: https://doi.org/10.1071/RJ19092.

Kenny, S. *et al*. (2018) 'Time-since-fire and climate interact to affect the structural recovery of an Australian semi-arid plant community', *Austral Ecology*, 43. Available at: https://doi.org/10.1111/aec.12582.

Kent, C.M. and Sherry, T.W. (2020) 'Behavioral niche partitioning reexamined: Do behavioral differences predict dietary differences in warblers?', *Ecology*, 101(8), p. e03077.

Kenward, R. (2004) 'Radio-tagging', in *Bird ecology and conservation: a handbook of techniques*. Oxford University Press. Available at: https://doi.org/10.1093/acprof:oso/9780198520863.001.0001.

Kenward, R.E. (2000) *A manual for wildlife radio tagging*. 2nd Edition. San Diego, California: Academic Press.

King, A.J. and Marshall, H.H. (2022) 'Optimal foraging', *Current Biology*, 32(12), pp. R680–R683.

Kingma, D.P. and Ba, J. (2014) 'Adam: A method for stochastic optimization', in *3rd International Conference for Learning Representations*.

Kirkpatrick, J.B. and Jenkinson, I. (2022) 'Effects of increasing fire frequency on conservation values in Eucalyptus grassy woodland in the process of invasion by Allocasuarina verticillata', *Fire*, 5(2), p. 31.

Klasing, K.C. (1999) 'Comparative Avian Nutrition.', *The Quarterly review of biology*, 74(3), pp. 358–358. Available at: https://doi.org/10.1086/393220.

Koehler, J. *et al*. (2017) 'The use of bioacoustics in anuran taxonomy: theory, terminology, methods and recommendations for best practice', *Zootaxa*, 4251(1), pp. 1–124. Available at: https://doi.org/10.11646/zootaxa.4251.1.1.

Koenecke, A. *et al*. (2020) 'Racial disparities in automated speech recognition', *Proceedings of the National Academy of Sciences*, 117(14), pp. 7684–7689. Available at: https://doi.org/10.1073/pnas.1915768117.

Kohlmann, S.G. and Risenhoover, K.L. (1996) 'Using artificial food patches to evaluate habitat quality for granivorous birds: An application of foraging theory', *The Condor*, 98(4), pp. 854–857. Available at: https://doi.org/10.2307/1369870.

Kolts, J. and McRae, S. (2017) 'Seasonal home range dynamics and sex differences in habitat use in a threatened, coastal marsh bird', *Ecology and Evolution*, 7(4), pp. 1101–1111.

Korniluk, M. *et al*. (2021) 'Habitat selection of foraging male Great Snipes on floodplain meadows: importance of proximity to the lek, vegetation cover and bare ground', *Ibis*, 163(2), pp. 486–506.

Kramer, D.L. (2001) 'Foraging behavior', in *Evolutionary Ecology*. Oxford University Press.

Krippendorff, K. (2011) *Computing Krippendorff's alpha-reliability*. Working Paper. University of Pennsylvania. Available at: https://repository.upenn.edu/asc_papers/43.

Kułaga, K. and Budka, M. (2019) 'Bird species detection by an observer and an autonomous sound recorder in two different environments: Forest and farmland', *PLoS ONE*, 14(2).

Kutt, A.S. and Woinarski, J. (2007) 'The effects of grazing and fire on vegetation and the vertebrate assemblage in a tropical savanna woodland in north-eastern Australia', *Journal of Tropical Ecology*, 23(1), pp. 95–106. Available at: https://doi.org/10.1017/s0266467406003579.

Laguna, J.M. *et al.* (2019) 'Conserving the endangered Black-throated Finch southern subspecies: what do we need to know?', *Emu-Austral Ornithology*, 119(4), pp. 331–345. Available at: https://doi.org/10.1080/01584197.2019.1605830.

Laguna, J.M. (2020) *Understanding uncertainty to inform conservation: tools to protect the endangered black-throated finch southern subspecies*. James Cook University.

Lambert, K.T.A. and McDonald, P.G. (2014) 'A low-cost, yet simple and highly repeatable system for acoustically surveying cryptic species', *Austral Ecology*, 39(7), pp. 779–785. Available at: https://doi.org/10.1111/aec.12143.

Landsberg, J. *et al.* (2003) 'Abundance and composition of plant species along grazing gradients in Australian rangelands', *Journal of Applied Ecology*, 40(6), pp. 1008–1024. Available at: https://doi.org/10.1111/j.1365-2664.2003.00862.x.

Lange, .T (1969) 'The Piosphere: Sheep Track and Dung Pattern', *Journal of Range Management* [Preprint]. Available at: https://doi.org/10.2307/3895849.

Langmore, N. and Bennett, A. (1999) 'Strategic concealment of sexual identity in an estrilid finch', *Proceedings of the Royal Society of London. Series B: Biological Sciences*, 266(1419), pp. 543–550.

Latif, Q.S. *et al.* (2020) 'Development and evaluation of habitat suitability models for nesting white-headed woodpecker (Dryobates albolarvatus) in burned forest', *PloS One*, 15(5), p. e0233043.

Laver, P.N. and Kelly, M.J. (2008) 'A critical review of home range studies', *The Journal of Wildlife Management*, 72(1), pp. 290–298. Available at: https://doi.org/10.2193/2005-589.

Le Galliard, J.F., Massot, M. and Clobert, J. (2012) 'Dispersal and range dynamics in changing climates: a review', *Dispersal ecology and evolution*, pp. 317–336.

Lebbink, G., Dwyer, J.M. and Fensham, R.J. (2021) 'Managed livestock grazing for conservation outcomes in a Queensland fragmented landscape', *Ecological management & restoration*, 22(1), pp. 5–9. Available at: https://doi.org/10.1111/emr.12460.

Lebbink, G., Fensham, R. and Cowley, R. (2018) 'Vegetation responses to fire history and soil properties in grazed semi-arid tropical savanna', *Rangeland Journal*, 40, pp. 271–285. Available at: https://doi.org/10.1071/RJ17075.

LeBien, J. *et al.* (2020) 'A pipeline for identification of bird and frog species in tropical soundscape recordings using a convolutional neural network', *Ecological Informatics*, 59, p. 101113. Available at: https://doi.org/10.1016/j.ecoinf.2020.101113.

LeDell, E. and Poirier, S. (2020) 'H2O AutoML: Scalable automatic machine learning', in *Proceedings of the 7th ICML Workshop on Machine Learning*.

Legge, S. *et al.* (2018) *Monitoring threatened species and ecological communities* (1 online resource (xxvii, 451 pages .) vol). Clayton, Vic.: CSIRO Publishing. Available at: https://search.ebscohost.com/login.aspx?direct=true&scope=site&db=nlebk&db=nlabk&AN=1688598.

Legge, S. *et al.* (2019) 'Interactions among threats affect conservation management outcomes: Livestock grazing removes the benefits of fire management for small mammals in Australian tropical savannas', *Conservation Science and Practice*, 1(7), p. e52. Available at: https://doi.org/10.1111/csp2.52.

Legge, S., Garnett, S. and Maute, K. (2015) 'A landscape-scale, applied fire management experiment promotes recovery of a population of the threatened Gouldian finch, Erythrura gouldiae, in Australia's

tropical savannas'. Available at: https://ro.uow.edu.au/cgi/viewcontent.cgi?article=4330&context=smhpapers.

Lenske, A.K. and Nocera, J.J. (2018) 'Field test of an automated radio-telemetry system: tracking local space use of aerial insectivores', *Journal of Field Ornithology*, 89(2), pp. 173–187. Available at: https://doi.org/10.1111/jofo.12254.

Lenz, T.I. and Facelli, J.M. (2005) 'The role of seed limitation and resource availability in the recruitment of native perennial grasses and exotics in a South Australian grassland', *Austral Ecology*, 30(6), pp. 684–694.

Lequitte-Charransol, P., Robert, A. and Jiguet, F. (2024) 'Increased adult movements and decreased juvenile apparent survival of urban crows during COVID-19 lockdowns', *Scientific Reports*, 14(1), p. 24135. Available at: https://doi.org/10.1038/s41598-024-74828-0.

Leveau, L.M. and Isla, F.I. (2021) 'Predicting bird species presence in urban areas with NDVI: An assessment within and between cities', *Urban Forestry & Urban Greening*, 63. Available at: https://doi.org/10.1016/j.ufug.2021.127199.

Lewandrowski, W. *et al.* (2018) 'Ecological niche and bet-hedging strategies for Triodia (R.Br.) seed germination', *Annals of Botany*, 121(2), pp. 367–375. Available at: https://doi.org/10.1093/aob/mcx158.

Ley, A. and Cook, S. (2001) 'The black-throated finch *Poephila cincta* in New South Wales', *Australian Field Ornithology*, 19(4), pp. 115–120.

Ley, A.J. and Cook, S.M. (2001) 'The Black-throated Finch "*Poephila cincta*" in New South Wales', *Australian Field Ornithology*, 19(4), pp. 115–120.

Li, X. and Kokko, H. (2019) 'Sex-biased dispersal: A review of the theory', *Biological Reviews*, 94(2), pp. 721–736.

Liedloff, A.C. *et al.* (2001) 'Modelling the trade-off between fire and grazing in a tropical savanna landscape, northern Australia', *Modelling & Sustainability*, 27(2), pp. 173–180. Available at: https://doi.org/10.1016/S0160-4120(01)00079-4.

Lieth, H. (1974) 'Purposes of a Phenology Book', in *Phenology and Seasonality Modeling*. New York: Springer Science & Business Media.

Liker, A. and Székely, T. (2005) 'MORTALITY COSTS OF SEXUAL SELECTION AND PARENTAL CARE IN NATURAL POPULATIONS OF BIRDS', *Evolution*, 59(4), pp. 890–897. Available at: https://doi.org/10.1111/j.0014-3820.2005.tb01762.x.

Lind, O. *et al.* (2014) 'Ultraviolet vision in birds: the importance of transparent eye media', *Proceedings of the Royal Society B: Biological Sciences*, 281(1774), p. 20132209. Available at: https://doi.org/10.1098/rspb.2013.2209.

Lindberg, M.S. and Walker, J. (2007) 'Satellite telemetry in avian research and management: sample size considerations', *The Journal of Wildlife Management*, 71(3), pp. 1002–1009. Available at: https://doi.org/10.2193/2005-696.

Liu, F. and Kong, Y. (2015) 'Zoib: an R package for Bayesian inference for beta regression and zero/one inflated beta regression.', *R J.*, 7(2), p. 34.

Liu, X. and Fu, Y.X. (2015) 'Exploring population size changes using SNP frequency spectra.', *Nature genetics*, 47(5), pp. 555–559.

Lloyd, M.V., Dixon, K.W. and Sivasithamparam, K. (2000) 'Comparative effects of different smoke treatments on germination of Australian native plants', *Austral Ecology*, 25(6), pp. 610–615. Available at: https://doi.org/10.1111/j.1442-9993.2000.tb00066.x.

Loiseau, N. *et al.* (2020) 'Global distribution and conservation status of ecologically rare mammal and bird species', *Nature Communications*, 11(1), p. 5071. Available at: https://doi.org/10.1038/s41467-020-18779-w.

Londe, D.W. *et al.* (2021) 'Fine-scale habitat selection limits trade-offs between foraging and temperature in a grassland bird', *Behavioral Ecology*, 32(4), pp. 625–637.

Lopez, K. *et al.* (2021) 'Evaluating evidence of mitonuclear incompatibilities with the sex chromosomes in an avian hybrid zone', *Evolution*, 75(6), pp. 1395–1414.

Lowe, K. (1989) *The Australian Bird Bander's Manual*. Canberra: Australian National Parks and Wildlife Service.

Lowe, W.H. and Allendorf, F.W. (2010) 'What can genetics tell us about population connectivity?', *Molecular ecology*, 19(15), pp. 3038–3051.

Lüdecke, D. (2023) *sjPlot: Data visualization for statistics in social science*. Available at: https://cran.r-project.org/web/packages/sjPlot/index.html.

Luikart, G. *et al.* (2010) 'Estimation of census and effective population sizes: the increasing usefulness of DNA-based approaches', *Conservation Genetics*, 11(2), pp. 355–373. Available at: https://doi.org/10.1007/s10592-010-0050-7.

Luis Marone *et al.* (2022) 'Plausible causes of seed preferences and diet composition in seed-eating passerines', *Journal of Avian Biology*, 2022(3), p. n/a-n/a. Available at: https://doi.org/10.1111/jav.02875.

Lunt, I.D. *et al.* (2007) 'A framework to predict the effects of livestock grazing and grazing exclusion on conservation values in natural ecosystems in Australia', *Australian Journal of Botany*, 55(4), pp. 401–415.

Mac Aodha, O. *et al.* (2018) 'Bat detective—Deep learning tools for bat acoustic signal detection', *PLoS Computational Biology*, 14(3), p. e1005995. Available at: https://doi.org/10.1371/journal.pcbi.1005995.

MacArthur, R.H. and Pianka, E.R. (1966) 'On optimal use of a patchy environment', *American Naturalist*, 100(916). Available at: https://doi.org/Doi%252010.1086/282454.

Magnusson, A. *et al.* (2023) *glmmTMB: Generalized linear mixed models using template model builder*. Available at: https://cran.r-project.org/web/packages/glmmTMB/index.html.

Magome, H. *et al.* (2008) 'Forage selection of sable antelope in Pilanesberg Game Reserve, South Africa', *South African Journal of Wildlife Research*, 38(1), pp. 35–41.

Maness, T.J. and Anderson, D.J. (2013) 'Predictors of Juvenile Survival in Birds - Predictores de la Supervivencia de Aves Juveniles', *Ornithological Monographs*, 78(1), pp. 1–55. Available at: https://doi.org/10.1525/om.2013.78.1.1.

Manly, B. *et al.* (2007) *Resource selection by animals: statistical design and analysis for field studies*. Springer Science & Business Media.

Mansouri, I. *et al.* (2022) 'Segregation of breeding habitats and feeding resources among five north African game species in Midelt province, Morocco', *Biologia*, 77(1), pp. 137–148.

Manu, S. and Cresswell, W. (2013) 'Diurnal patterns of mass gain in tropical granivores suggest avoidance of high midday temperatures during foraging, rather than the starvation–predation risk trade-off', *Ostrich*, 84(2), pp. 95–100. Available at: https://doi.org/10.2989/00306525.2013.821683.

Marone, L. *et al.* (2017) 'Diet switching of seed-eating birds wintering in grazed habitats of the central Monte Desert, Argentina', *Condor*, 119(4), pp. 673–682. Available at: https://doi.org/10.1650/Condor-17-61.1.

Marone, Luis *et al.* (2017) 'Diet switching of seed-eating birds wintering in grazed habitats of the central Monte Desert, Argentina/Modificacion de la dieta invernal de las aves granivoras en habitats sujetos a pastoreo de la porcion central del Desierto del Monte, Argentina.(RESEARCH ARTICLE)(Report)(Author abstract)', *The Condor*, 119(4), p. 673. Available at: https://doi.org/10.1650/CONDOR-17-61.1.

Marshall, N.A. *et al.* (2011) 'Considering the social dimension of invasive species: the case of buffel grass', *Environmental Science and Policy*, 14(3), pp. 327–338.

Marshall, V.M., Lewis, M.M. and Ostendorf, B. (2012) 'Buffel grass (Cenchrus ciliaris) as an invader and threat to biodiversity in arid environments: A review', *Journal of Arid Environments*, 78, pp. 1–12. Available at: https://doi.org/10.1016/j.jaridenv.2011.11.005.

Martin, T.E. (1987) 'Food as a limit on breeding birds: A life-history perspective', *Annual Review of Ecology and Systematics*, 18, pp. 453–487.

Martin, T.E. (1992) 'Breeding productivity considerations: What are the appropriate habitat features for management?', *Ecology and conservation of neotropical migrant landbirds*, pp. 455–473.

Martin, T.E. (1993) 'Nest predation and nest sites.', *BioScience*, 43(8), pp. 523–532.

Martin, T.E. (1995) 'Avian Life History Evolution in Relation to Nest Sites, Nest Predation, and Food', *Ecological Monographs*, 65(1), pp. 101–127.

Martin, T.E. (1998) 'ARE MICROHABITAT PREFERENCES OF COEXISTING SPECIES UNDER SELECTION AND ADAPTIVE?', *Ecology*, 79(2), pp. 656–670. Available at: https://doi.org/10.1890/0012-9658(1998)079%255B0656:AMPOCS%255D2.0.CO;2.

Martin, T.E. and Roper, J.J. (1988) 'Nest Predation and Nest-Site Selection of a Western Population of the Hermit Thrush', *The Condor*, 90(1), pp. 51–57.

Martin, T.G. *et al.* (2017) 'Timing of Protection of Critical Habitat Matters', *Conservation Letters*, 10(3), pp. 308–316. Available at: https://doi.org/10.1111/conl.12266.

Martín-Forés, I., Guerin, G.R. and Lowe, A.J. (2017) 'Weed abundance is positively correlated with native plant diversity in grasslands of southern Australia', *PLoS One*, 12(6), p. e0178681.

Marvin, D.C. *et al.* (2016) 'Integrating technologies for scalable ecology and conservation', *Global Ecology and Conservation*, 7, pp. 262–275. Available at: https://doi.org/10.1016/j.gecco.2016.07.002.

Maslo, B. *et al.* (2016) 'Regional drivers of clutch loss reveal important trade-offs for beach-nesting birds', *PeerJ*, 13(4), p. e2460.

Mathur, S. *et al.* (2023) 'An evolutionary perspective on genetic load in small, isolated populations as informed by whole genome resequencing and forward-time simulations.', *Evolution*, 77(3), pp. 690–704.

Mathur, S. and DeWoody, J.A. (2021) 'Genetic load has potential in large populations but is realized in small inbred populations.', *Evolutionary Applications*, 14(6), pp. 1540–1557.

Matthiopoulos, J. *et al.* (2015) 'Establishing the link between habitat selection and animal population dynamics', *Ecological Monographs*, 85(3), pp. 413–436. Available at: https://doi.org/10.1890/14-2244.1.

Maute, K. (2011) *Variation in the health of tropical finches in relation to conservation status, season and land tenure*. University of Wollongong.

Mayor, S.J. *et al.* (2009a) 'Habitat selection at multiple scales', *Ecoscience*, 16(2), pp. 238–247. Available at: https://doi.org/10.2980/16-2-3238.

Mayor, S.J. *et al.* (2009b) 'Habitat Selection at Multiple Scales', *Ecoscience*, 16(2), pp. 238–247. Available at: https://doi.org/10.2980/16-2-3238.

McCaffrey, N.B. *et al.* (2014) 'Novel "stratified-meander" technique improves survey effort of the rare Pagoda Rock Daisy growing remotely on rocky cliff edges', *Ecological Management & Restoration*, 15(1), pp. 94–97. Available at: https://doi.org/10.1111/emr.12087.

McClintock, B.T., White, G.C. and Pryde, M.A. (2019) 'Improved methods for estimating abundance and related demographic parameters from mark-resight data', *Biometrics*, 75(3), pp. 799–809. Available at: https://doi.org/10.1111/biom.13058.

McCosker, J., Rolfe, J. and Fensham, R. (2009) 'Can bare ground cover server as a surrogate for plant biodiversity in grazed tropical woodlands?', *The Rangeland Journal*, 31(1), pp. 103–109.

McGregor, H. *et al.* (2014) 'Landscape Management of Fire and Grazing Regimes Alters the Fine-Scale Habitat Utilisation by Feral Cats', *PLOS ONE* [Preprint]. Available at: https://doi.org/10.1371/journal.pone.0109097.

Mcintyre, S., Heard, K.M. and Martin, T.G. (2003) 'The relative importance of cattle grazing in subtropical grasslands: does it reduce or enhance plant biodiversity?', *Journal of Applied Ecology*, 40(3), pp. 445–457. Available at: https://doi.org/10.1046/j.1365-2664.2003.00823.x.

McIntyre, S. and Lavorel, S. (2001) 'Livestock grazing in subtropical pastures: steps in the analysis of attribute response and plant functional types', *Journal of Ecology*, 89(2), pp. 209–226. Available at: https://doi.org/10.1046/j.1365-2745.2001.00535.x.

McIvor, J. (2012) *Sustainable management of the Burdekin grazing lands – A technical guide of options for stocking rate management, pasture spelling, infrastructure development and prescribed burning to optimise animal production, profitability, land condition and water quality outcomes*. State of Queensland.

McIvor, J.G. *et al.* (1996) 'Seed production by native and naturalised grasses in north-east Queensland: effects of stocking rate and season', *Tropical Grasslands*, 30, pp. 262–269.

McWilliams, S.R. (2001) 'Ecology of vertebrate nutrition', *e LS* [Preprint].

Meij, van der, Griekspoor, and Bout (2004) 'The effect of seed hardness on husking time in finches', *Animal Biology*, 54(2), pp. 195–205. Available at: https://doi.org/10.1163/1570756041445164.

Melzer, R. (2015) 'When is stock grazing an appropriate "tool" for reducing "Cenchrus ciliaris" (Buffel grass) on conservation reserves?', *Proceedings of the Royal Society of Queensland*, 120(Journal Article), pp. 53–68.

Melzer, R. and Hines, H.B. (2022) 'Queensland Parks and Wildlife Service Planned Burn Guidelines: Introductory Volume', *Ecological Assessment Unit, Queensland Parks and Wildlife Service and Partnerships, Department of Environment and Science, Queensland Government* [Preprint].

Mendoza-Silva, G.M., Torres-Sospedra, J. and Huerta, J. (2019) 'A meta-review of indoor positioning systems', *Sensors*, 19(20), p. 4507. Available at: https://doi.org/10.3390/s19204507.

Merow, C., Smith, M.J. and Silander Jr, J.A. (2013) 'A practical guide to MaxEnt for modeling species' distributions: what it does, and why inputs and settings matter', *Ecography*, 36(10), pp. 1058–1069. Available at: https://doi.org/10.1111/j.1600-0587.2013.07872.x.

Metcalfe, N.B. and Monaghan, P. (2001) 'Compensation for a bad start: grow now, pay later?', *Trends in ecology & evolution*, 16(5), pp. 254–260.

Meyer, E. and Agnew, L.R. (2012) 'Black-throated finch "poephila cincta cincta" observed feeding on eucalypt seed', *Queensland Naturalist*, 50(4/5/6), pp. 66–68.

Milesi, F.A., De Casenave, J.L. and Cueto, V.R. (2008) 'Selection of foraging sites by desert granivorous birds: vegetation structure, seed availability, species-specific foraging tactics, and spatial scale', *The Auk*, 125(2), pp. 473–484. Available at: https://doi.org/10.1525/auk.2008.07085.

Miller, G. *et al.* (2010) 'Ecological impacts of buffel grass (Cenchrus ciliaris L.) invasion in central Australia – does field evidence support a fire-invasion feedback?', *The Rangeland Journal*, 32(4). Available at: https://www-publish-csiro-au.ezp01.library.qut.edu.au/rj/Fulltext/RJ09076.

Misenhelter, M.D. and Rotenberry, J.T. (2000) 'CHOICES AND CONSEQUENCES OF HABITAT OCCUPANCY AND NEST SITE SELECTION IN SAGE SPARROWS', *Ecology*, 81(10), pp. 2892–2901. Available at: https://doi.org/10.1890/0012-9658(2000)081%255B2892:CACOHO%255D2.0.CO;2.

Mishra, I. and Kumar, V. (2019) 'The quantity-quality trade-off: differential effects of daily food times on reproductive performance and offspring quality in diurnal zebra finches', *Journal of Experimental Biology*, p. jeb.196667. Available at: https://doi.org/10.1242/jeb.196667.

Mitchell, D.F. (1996) *Foraging ecology of the black-throated finch Poephila cincta cincta*. M. Sc. Thesis. James Cook University.

Mitchell, D. F. (1996) *Foraging Ecology of the Black-throated Finch Poephila cincta cincta*. M. Sc. Thesis. James Cook University.

Mitchell, M.S. and Powell, R.A. (2004) 'A mechanistic home range model for optimal use of spatially distributed resources', *Ecological Modelling*, 177(1–2), pp. 209–232. Available at: https://doi.org/10.1016/j.ecolmodel.2004.01.015.

Moen, C.A. and Gutiérrez, R.J. (1997) 'California Spotted Owl Habitat Selection in the Central Sierra Nevada', *The Journal of Wildlife Management*, 61(4), pp. 1281–1287.

Mohri, M., Rostamizadeh, A. and Talwalkar, A. (2018) *Foundations of machine learning*. MIT Press.

Molokwu, M.N., Nilsson, J.-Å. and Olsson, O. (2011) 'Diet selection in birds: trade-off between energetic content and digestibility of seeds', *Behavioral Ecology*, 22(3), pp. 639–647. Available at: https://doi.org/10.1093/beheco/arr025.

Molokwu, M.N., Olsson, O. and Ottosson, U. (2007) 'Feeding behaviour of birds foraging on predictable resources in habitats of different quality', *Ostrich*, 78(2), pp. 295–298. Available at: https://doi.org/Doi%252010.2989/Ostrich.2007.78.2.28.108.

Monarch, R.M. (2021) *Human-in-the-loop machine learning: Active learning and annotation for human-centered AI*. Shelter Island, New York: Manning Publications.

Monroe, A.P. *et al.* (2022) 'Spatial scale selection for informing species conservation in a changing landscape', *Ecosphere*, 13(12), p. e4320. Available at: https://doi.org/10.1002/ecs2.4320.

Moorcroft, D. *et al.* (2002) 'The selection of stubble fields by wintering granivorous birds reflects vegetation cover and food abundance', *Journal of applied Ecology*, pp. 535–547.

Moore, C.E. *et al.* (2018) 'Seasonal, interannual and decadal drivers of tree and grass productivity in an Australian tropical savanna', *Global Change Biology*, 24(6), pp. 2530–2544. Available at: https://doi.org/10.1111/gcb.14072.

Moore, N.A., Camac, J.S. and Morgan, J.W. (2019) 'Effects of drought and fire on resprouting capacity of 52 temperate Australian perennial native grasses', *New Phytologist*, 221(3), pp. 1424–1433. Available at: https://doi.org/10.1111/nph.15480.

Morgan, G. *et al.* (2002) *The Conservation of Biodiversity in the Desert Uplands*.

Morgan, J.W. and Lunt, I.D. (1999) 'Effects of time-since-fire on the tussock dynamics of a dominant grass (Themeda triandra) in a temperate Australian grassland', *Biological Conservation*, 88(3), pp. 379–386. Available at: https://doi.org/10.1016/S0006-3207(98)00112-8.

Morris, D.W. (1987) 'Ecological Scale and Habitat Use', *Ecology*, 68(2), pp. 362–369.

Morrison, C. *et al.* (2016) 'Causes and consequences of spatial variation in sexc ratios in a declining bird species', *Journal of Animal Ecology*, 85, pp. 1298–1306.

Mortimer, J.A. and Greene, T.C. (2017) 'Investigating bird call identification uncertainty using data from processed audio recordings', *New Zealand Journal of Ecology*, 41(1), pp. 126–133. Available at: https://doi.org/10.20417/nzjecol.41.10.

Mott, J. (1978) 'Dormancy and Germination in Five Native Grass Species From Savannah Woodland Communities of the Northern Territory', *Australian Journal of Botany*, 26(5), p. 621. Available at: https://doi.org/10.1071/BT9780621.

Mula Laguna, J. *et al.* (2019) 'Conserving the endangered black-throated finch southern subspecies: what do we need to know?', *Emu - Austral Ornithology*, 119(4), pp. 331–345. Available at: https://doi.org/10.1080/01584197.2019.1605830.

Muller, H.J. (1950) 'Our load of mutations.', *American journal of human genetics*, 2(2), p. 111.

Murphy, B., Andersen, A. and Parr, C. (2016) 'The underestimated biodiversity of tropical grassy biomes', *Philosophical Transactions of the Royal Society B: Biological Sciences*, 371(1703), p. 20150319. Available at: https://doi.org/10.1098/rstb.2015.0319.

Murphy, B. and Russell-Smith, J. (2010) 'Fire severity in a northern Australian savanna landscape: The importance of time since previous fire', *International Journal of Wildland Fire*, 19, pp. 46–51. Available at: https://doi.org/10.1071/WF08202.

Murphy, B.P. *et al.* (2014) 'Fire regimes and woody biomass dynamics in Australian savannas', *Journal of Biogeography*, 41(1), pp. 133–144.

Nadachowska-Brzyska, K. *et al.* (2015) 'Temporal dynamics of avian populations during Pleistocene revealed by whole-genome sequences.', *Current Biology*, 25(10), pp. 1375–1380.

Naef-Daenzer, B. *et al.* (2005) 'Miniaturization (0.2 g) and evaluation of attachment techniques of telemetry transmitters', *Journal of Experimental Biology*, 208(21), pp. 4063–4068. Available at: https://doi.org/10.1242/jeb.01870.

Nagy Koves Hrabar, H.D. and Perrin, M. (2002) 'The effect of bill structure on seed selection by granivorous birds', *African Zoology*, 37(1), pp. 67–80.

Nathan, R. *et al.* (2008) 'A movement ecology paradigm for unifying organismal movement research', *Proceedings of the National Academy of Sciences*, 105(49), pp. 19052–19059. Available at: https://doi.org/10.1073/pnas.0800375105.

Navarro, J., Kaliontzopoulou, A. and González-Solís, J. (2009) 'Sexual dimorphism in bill morphology and feeding ecology in Cory's shearwater (Calonectris diomedea)', *Zoology (Jena)*, 112(2), pp. 128–138. Available at: https://doi.org/10.1016/j.zool.2008.05.001.

Neldner, V.J. *et al.* (2022) *Methodology for survey and mapping of regional ecosystems and vegetation communities in Queensland v.6.0*. Brisbane: Queensland Herbarium, Queensland Department of Environment and Science.

Neldner, V.J. *et al.* (2023) 'Methodology for surveying and mapping of regional ecosystems and vegetation communities in Queensland. Version 7.0'. Queensland Herbarium, Queensland Department of Environment, Science and Innovation, Brisbane.

Nguyen, Q., Valizadegan, H. and Hauskrecht, M. (2014) 'Learning classification models with soft-label information', *Journal of the American Medical Informatics Association*, 21(3), pp. 501–508. Available at: https://doi.org/10.1136/amiajnl-2013-001964.

Nicole Isles (2007) *Patterns of Foraging Behaviour of the Black-throated Finch during Breeding Season*. James Cook University.

Nieto, S., Flombaum, P. and Garbulsky, M.F. (2015) 'Can temporal and spatial NDVI predict regional bird-species richness?', *Global Ecology and Conservation*, 3, pp. 729–735.

NRA (2007) *Review of the Ecology, Threats and Management Requirements of the Black-Throated Finch (Poephila Cincta Cincta) to Support Assessment Processes under the Environment Protection and Biodiversity Conservation Act 1999*.

NRA Environmental Consultants (2005) *Enertrade North Queensland Gas Pipeline Black-throated Finch Studies (Post-Construction)*. Brisbane: Unpublished report prepared by NRA Environmental Consultants for Enertrade.

NRA Environmental Consultants (2009) *Distribution of potential nesting habitat for the Black-throated Finch (Poephila cincta cincta) in the Townsville Local Government Area - A predictive model*. Unpublished report. Black-throated Finch Trust.

Nugent, D.T. *et al.* (2022) 'Multi-scale habitat selection by a cryptic, critically endangered grassland bird—The Plains-wanderer (Pedionomus torquatus): Implications for habitat management and conservation', *Austral Ecology*, 47(3), pp. 698–712.

O'Connor, T. and Pickett, G. (1992) 'The influence of grazing on seed production and seed banks of some African savanna grasslands', *Journal of Applied ecology*, pp. 247–260. Available at: https://doi.org/10.2307/2404367.

Oksanen, J. *et al.* (2024) 'vegan: Community Ecology Package'. (R package version 2.6-8). Available at: https://CRAN.R-project.org/package=vegan.

Olkola Aboriginal Corporation (2022) *Draft Recovery Plan for Golden-shouldered Parrot (Psephotus chrysopterygius)*. Canberra: Department of Agriculture, Water and the Environment. Available at: https://www.agriculture.gov.au/sites/default/files/documents/draft-recovery-plan-golden-shouldered-parrot.pdf (Accessed: 12 March 2025).

Olsen, B.J. *et al.* (2013) 'Sexual dimorphism in a feeding apparatus is driven by mate choice and not niche partitioning', *Behavioral Ecology*, 24(6), pp. 1327–1338. Available at: https://doi.org/10.1093/beheco/art071.

Open Acoustic Devices (n.d.) *Audio Quality*. Available at: https://www.openacousticdevices.info/audio (Accessed: 19 March 2023).

O'Reagain, P.J. *et al.* (2018) 'Wambiana grazing trial Phase 3: stocking strategies for improving carrying capacity, land condition and biodiversity outcomes.' Meat and Livestock Australia. Available at: http://era.daf.qld.gov.au/id/eprint/6338/ (Accessed: 17 February 2025).

Orr, D. (2010) 'Managing the grass-legume balance in Stylosanthes scabra cv. Seca pastures in central Queensland', *Tropical Grasslands*, 44, pp. 174–183.

Orr, D. and O'Reagain, P. (2011) 'Managing for rainfall variability: Impacts of grazing strategies on perennial grass dynamics in a dry tropical savanna', *The Rangeland Journal*, 33, pp. 209–220. Available at: https://doi.org/10.1071/RJ11032.

Orr, D.M., McKeon, G.M. and Day, K.A. (1991) 'Burning and exclosure can rehabilitate degraded black speargrass (Heteropogon contortus) pastures', *Tropical Grasslands*, 25, pp. 333–336.

van Osta, J. *et al.* (2023) 'An active learning framework and assessment of inter-annotator agreement facilitate automated recogniser development for vocalisations of a rare species, the southern black-throated finch (*Poephila cincta cincta*)', *Ecological Informatics*, 77, p. 102233. Available at: https://doi.org/10.1016/j.ecoinf.2023.102233.

van Osta, J. *et al.* (2024a) 'A location fingerprinting approach for the automated radio telemetry of wildlife and comparison to alternative methods', *Animal Biotelemetry*, 12(1), p. 24. Available at: https://doi.org/10.1186/s40317-024-00379-w.

van Osta, J. *et al.* (2024b) 'Local resource availability drives habitat use by a threatened avian granivore in savanna woodlands', *PLoS ONE*, 19(8), p. e0306842. Available at: https://doi.org/10.1371/journal.pone.0306842.

van Osta, J. (2025) *Machine learning methods to investigate the movement ecology of the endangered southern black-throated finch*. PhD Thesis. Griffith University. Available at: https://research-repository.griffith.edu.au/items/24350705-025f-4c82-b5b9-1254b97da1a6 (Accessed: 11 March 2025).

Otani, N. *et al.* (2020) 'Binary classification with ambiguous training data', *Machine Learning*, 109, pp. 2369–2388. Available at: https://doi.org/10.1007/s10994-020-05915-2.

Otis, D.L. (1997) 'Analysis of Habitat Selection Studies with Multiple Patches within Cover Types', *The Journal of Wildlife Management*, 61(4), pp. 1016–1022.

Owens, I. and Hartley, I. (1998) 'Sexual dimorphism in birds: Why are there so many different forms of dimorphism?', *Proceedings of the Royal Society B: Biological Sciences*, 265, pp. 397–407.

Owens, I.P.F. and Bennett, P.M. (1994) 'Mortality Costs of Parental Care and Sexual Dimorphism in Birds', *Proceedings: Biological Sciences*, 257(1348), pp. 1–8.

Palm, E.C. *et al.* (2020) 'The long road to protecting critical habitat for species at risk: The case of southern mountain woodland caribou', *Conservation Science and Practice*, 2(7), p. e219.

Palsbøll, P.J., Berube, M. and Allendorf, F.W. (2007) 'Identification of management units using population genetic data.', *Trends in Ecology & Evolution*, 22(1), pp. 11–16.

Palstra, F.P. and Fraser, D.J. (2012) 'Effective/census population size ratio estimation: a compendium and appraisal', *Ecology and Evolution*, 2(9), pp. 2357–2365. Available at: https://doi.org/10.1002/ece3.329.

Paltridge, R., Gibson, D. and Edwards, G. (1997) 'Diet of the feral cat, Felis catus, in central Australian grassland habitats during population cycles of its principal prey', *Wildlife Research*, 24(1), pp. 67–76.

Papageorgiou, D. *et al.* (2021) 'Seasonality impacts collective movements in a wild group-living bird', *Movement Ecology*, 9(1), p. 38. Available at: https://doi.org/10.1186/s40462-021-00271-9.

Paradis, E. *et al.* (1998) 'Patterns of natal and breeding dispersal in birds', *Journal of Animal ecology*, 67(4), pp. 518–536.

Paradis, E. and Schliep, K. (2019) 'ape 5.0: an environment for modern phylogenetics and evolutionary analyses in R', 35, pp. 526–528. Available at: https://doi.org/10.1093/bioinformatics/bty633.

Paruk, J.D. (2018) 'The Cornell Lab of Ornithology Handbook of Bird Biology'.

Pascoe, B.A. *et al.* (2021) 'Dynamics of bird assemblages in response to temporally and spatially variable resources in arid Australia', *Ecology and evolution*, 11(9), pp. 3977–3990.

Paszke, A. *et al.* (2019) 'Pytorch: An imperative style, high-performance deep learning library', *Advances in Neural Information Processing Systems*, 32, pp. 8026–8037.

Paterson, A. (2005) 'Historical Interactions between Aborigines and European Pastoralists in Australia's Drylands', in, pp. 267–280.

Paton, C., Hamilton, J. and Emery, T. (2011) 'Best-bet practices for managing the grazing lands of the Maranoa Balonne: a technical guide of options for optimising animal production, profitability and land condition', *Queensland Department of Employment, Economic Development and Innovation: Brisbane, Qld, Australia* [Preprint].

Paton, C. and Tyler, R. (2017) *Managing famine, fire and flood in the Desert Uplands: Graziers' experiences*. Desert Uplands Build-Up and Development Strategy Committee.

Paton, R.S. and Matthiopoulos, J. (2016) 'Defining the scale of habitat availability for models of habitat selection', *Ecology*, 97(5), pp. 1113–1122. Available at: https://doi.org/10.1890/14-2241.1.

Pausas, J.G. and Lamont, B.B. (2022) 'Fire-released seed dormancy - a global synthesis', *Biological Reviews*, 97(4), pp. 1612–1639. Available at: https://doi.org/10.1111/brv.12855.

Paxton, K.L. *et al.* (2022) 'Optimizing trilateration estimates for tracking fine-scale movement of wildlife using automated radio telemetry networks', *Ecology and Evolution*, 12(2), p. e8561. Available at: https://doi.org/10.1002/ece3.8561.

Peacock, J., Macleod, R., Davies, G Matt, *et al.* (2024) 'Habitat Selection of Three Neotropical Grassland Birds Is Dependent on Vegetation Structure and Resources', *Diversity*, 16(4), p. 229.

Peacock, J., Macleod, R., Davies, G. Matt, *et al.* (2024) 'Habitat Selection of three neotropical grassland birds is dependent on vegetation structure and resources', *Diversity*, 16(4), p. 229. Available at: https://doi.org/10.3390/d16040229.

Pedregosa, F. *et al.* (2011) 'Scikit-learn: Machine learning in python', *Journal of Machine Learning Research*, 12, pp. 2825–2830.

Peel, M.C., Finlayson, B.L. and McMahon, T.A. (2007) 'Updated world map of the Köppen-Geiger climate classification', *Hydrology and Earth System Sciences*, 11(5), pp. 1633–1644. Available at: https://doi.org/10.5194/hess-11-1633-2007.

Perea, R., Venturas, M. and Gil, L. (2014) 'Seed predation on the ground or in the tree? Size-related differences in behavior and ecology of granivorous birds', *Acta Ornithologica*, 49(1), pp. 119–130.

Pérez-Granados, C. and Traba, J. (2021) 'Estimating bird density using passive acoustic monitoring: a review of methods and suggestions for further research', *Ibis*, 163(3), pp. 765–783. Available at: https://doi.org/10.1111/ibi.12944.

Petkova, D. (2024) *reemsplots2: Generate plots to inspect and visualize the results of EEMS. R package version 0.1.0, commit fbc2de644d9f591c91b795489b5026f9851f7fd8*. Available at: https://github.com/dipetkov/reemsplots2.

Petkova, D., Novembre, J. and Stephens, M. (2016) 'Visualizing spatial population structure with estimated effective migration surfaces', *Nature genetics*, 48(1), pp. 94–100.

Pfeiffer, M. *et al.* (2019) 'Grazing and aridity reduce perennial grass abundance in semi-arid rangelands- Insights from a trait-based dynamic vegetation model', *Ecological Modelling*, 395, pp. 11–22.

Phillips, S.J. and Dudík, M. (2008) 'Modeling of species distributions with Maxent: new extensions and a comprehensive evaluation', *Ecography*, 31(2), pp. 161–175. Available at: https://doi.org/10.1111/j.0906-7590.2008.5203.x.

Pickett, S.R. and Siriwardena, G.M. (2011) 'The relationship between multi-scale habitat heterogeneity and farmland bird abundance', *Ecography*, 34(6), pp. 955–969.

Pilar, C. and Gabriel, M. (1998) 'Phenological pattern of fifteen Mediterranean phanaerophytes from Quercus ilex communities of NE-Spain', *Plant Ecology*, 139, pp. 103–112.

Pinowska, B. (1979) 'The effect of energy and building resources of females on the production of house sparrow (Passer domesticus (L.)) populations', *Ekol. pol*, 27(36), p. 396.

Polak, M. (2007) 'Nest-Site Selection and Nest Predation in the Great Bittern Botaurus stellaris Population in Eastern Poland', *Ardea*, 95(1), pp. 31–38. Available at: https://doi.org/10.5253/078.095.0104.

Preston, J.C. and Fjellheim, S. (2020) 'Understanding past, and predicting future, niche transitions based on grass flowering time variation', *Plant Physiology*, 183(3), pp. 822–839.

Price, B. *et al.* (2013) 'Disentangling how landscape spatial and temporal heterogeneity affects Savanna birds', *PLoS One*, 8(9), p. e74333.

Priyadarshani, N., Marsland, S. and Castro, I. (2018) 'Automated birdsong recognition in complex acoustic environments: a review', *Journal of Avian Biology*, 49(5). Available at: https://doi.org/10.1111/jav.01447.

Promislow, D.E.L., Robert, M. and Thomas, M. (1992) 'Mortality Costs of Sexual Dimorphism in Birds', *Proceedings: Biological Sciences*, 250(1328), pp. 143–150.

Pryke, S.R. *et al.* (2012) 'Covariation in Life-History Traits: Differential Effects of Diet on Condition, Hormones, Behavior, and Reproduction in Genetic Finch Morphs', *The American Naturalist*, 179(3), pp. 375–390. Available at: https://doi.org/10.1086/664078.

Puhr, R. *et al.* (2017) 'Firth's logistic regression with rare events: accurate effect estimates and predictions?', *Statistics in Medicine*, 36(14), pp. 2302–2317. Available at: https://doi.org/10.1002/sim.7273.

Purvis, A. *et al.* (2019) *Status and trends-nature*. Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. Available at: https://doi.org/10.5281/zenodo.3831674.

Pustejovsky, J. and Stubbs, A. (2012) *Natural Language Annotation for Machine Learning: A guide to corpus-building for applications*. O'Reilly Media, Inc.

Pyke, G.H. (1984) 'Optimal foraging theory: a critical review', *Annual review of ecology and systematics*, 15, pp. 523–575.

Queensland Department of Environment, Tourism, and Science and Innovation (2021) 'Biodiversity status of 2019 remnant regional ecosystems - Queensland - Version 12.2'. Queensland Government.

Queensland Herbarium (2024) *Regional Ecosystem Description Database (REDD). Version 13.1*. Brisbane: Department of Environment, Science and Innovation.

Qvarnström, A. *et al.* (2003) 'Male ornamentation, timing of breeding, and cost of polygyny in the collared flycatcher', *Behavioral Ecology*, 14(1), pp. 68–73. Available at: https://doi.org/10.1093/beheco/14.1.68.

Qvarnström, A., Brommer, J.E. and Gustafsson, L. (2006) 'Testing the genetics underlying the co-evolution of mate choice and ornament in the wild', *Nature*, 441(7089), pp. 84–86. Available at: https://doi.org/10.1038/nature04564.

R Core Team (2023) 'R: A language and environment for statistical computing'. Vienna, Austria: R Foundation for Statistical Computing. Available at: https://www.R-project.org.

Radford, I.J. *et al.* (2024) 'Prescribed burning beneficial or neutral for native herbaceous vegetation in an invader dominated commercially grazed savanna', *Austral Ecology*, 49(8), p. e13581. Available at: https://doi.org/10.1111/aec.13581.

Raeder, T., Forman, G. and Chawla, N.V. (2012) 'Learning from imbalanced data: Evaluation matters', in *Data Mining: Foundations and Intelligent Paradigms: Volume 1: Clustering, Association and Classification*. New York: Springer, pp. 315–331. Available at: https://doi.org/10.1007/978-3-642-23166-7_12.

Raffard, A. *et al.* (2019) 'The community and ecosystem consequences of intraspecific diversity: A meta-analysis.', *Biological Reviews*, 94(2), pp. 648–661.

Raphael, M.G. and Molina, R. (2007) *Conservation of rare or little-known species: biological, social, and economic considerations*. Washington: Island Press (Book, Whole).

Rathcke, B. and Lacey, E.P. (1985) 'Phenological Patterns of Terrestrial Plants', *Annual Review of Ecology and Systematics*, 16, pp. 179–214.

Read, J.L. *et al.* (2020) 'Ranking buffel: Comparative risk and mitigation costs of key environmental and socio-cultural threats in central Australia', *Ecology and Evolution*, 10(23), pp. 12745–12763.

Read, T. and Bellairs, S. (1999) 'Smoke affects the Germination of Native Grasses of New South Wales', *Australian Journal of Botany*, 47, pp. 563–576. Available at: https://doi.org/10.1071/BT97124.

Rechetelo, J. (2015) *Movement, habitat requirements, nesting and foraging site selection: a case study of an endangered granivorous bird, the Black-throated finch* Poephila cincta cincta *in north-eastern Australia*. James Cook University.

Rechetelo, Juliana (2015) *Movement, habitat requirements, nesting and foraging site selection: a case study of an endangered granivorous bird, the Black-throated finch* Poephila cincta cincta *in north-eastern Australia*. James Cook University.

Rechetelo, J. *et al.* (2016a) 'Movement patterns, home range size and habitat selection of an endangered resource tracking species, the black-throated finch (Poephila cincta cincta)', *PLoS One*, 11(11). Available at: https://doi.org/10.1371/journal.pone.0167254.

Rechetelo, J. *et al.* (2016b) 'Movement patterns, home range size and habitat selection of an endangered resource tracking species, the black-throated finch (Poephila cincta cincta)', *PloS one*. Edited by D.A. Lightfoot, 11(11), pp. e0167254–e0167254. Available at: https://doi.org/10.1371/journal.pone.0167254.

Reed, D.H. and Frankham, R. (2003) 'Correlation between fitness and genetic diversity.', *Conservation biology*, 17(1), pp. 230–237.

Reid, J. and Fleming, M. (1992) 'The conservation status of birds in arid Australia.', *The Rangeland Journal*, 14(2), pp. 65–91. Available at: https://doi.org/10.1071/RJ9920065.

Reidsma, D. and Carletta, J. (2008) 'Reliability measurement without limits', *Computational Linguistics*, 34(3), pp. 319–326. Available at: https://doi.org/10.1162/coli.2008.34.3.319.

Remešová, E. *et al.* (2020) 'Foraging behaviour of songbirds in woodlands and forests in eastern Australia: resource partitioning and guild structure', *Emu-Austral Ornithology*, 120(1), pp. 22–32.

Renton, K. *et al.* (2015) 'Resource requirements of parrots: nest site selectivity and dietary plasticity of Psittaciformes', *Journal of Ornithology*, 156, pp. 73–90.

Reside, A *et al.* (2019) 'How to send a finch extinct', *Environmental Science & Policy*, 94, pp. 163–173.

Reside, April *et al.* (2019) 'How to send a finch extinct', *Environmental science & policy*, 94, pp. 163–173. Available at: https://doi.org/10.1016/j.envsci.2019.01.005.

Ricci, F., Rokach, L. and Shapira, B. (2022) *Recommender systems handbook*. Third edition. New York, NY: Springer. Available at: https://doi.org/10.1007/978-1-0716-2197-4.

Richardson, K. *et al.* (2022) 'Social dynamics of core members in mixed-species bird flocks change across a gradient of foraging habitat quality', *PLOS ONE*, 17, p. e0262385. Available at: https://doi.org/10.1371/journal.pone.0262385.

Richie, R., Grover, S. and Tsui, F.R. (2022) 'Inter-annotator agreement is not the ceiling of machine learning performance: Evidence from a comprehensive set of simulations', in *Proceedings of the 21st Workshop on Biomedical Language Processing*, pp. 275–284. Available at: https://doi.org/10.18653/v1/2022.bionlp-1.26.

Ríos, J.M., Mangione, A. and Marone, L. (2012) 'Effects of Nutritional and Anti-Nutritional Properties of Seeds on the Feeding Ecology of Seed-Eating Birds of the Monte Desert, Argentina', *The Condor*, 114(1), pp. 44–55. Available at: https://doi.org/10.1525/cond.2012.110043.

Ripperger, S.P. *et al.* (2020) 'Thinking small: next-generation sensor networks close the size gap in vertebrate biologging', *PLoS biology*, 18(4), p. e3000655. Available at: https://doi.org/10.1371/journal.pbio.3000655.

Robb, G. *et al.* (2008) 'Food for thought: supplementary feeding as a driver of ecological change in avian populations', *Frontiers in Ecology and the Environment*, 6(9), pp. 476–484.

Roberts, N.J. (2011) 'Investigation into survey techniques of large mammals: surveyor competence and camera-trapping vs. transect-sampling', *Bioscience Horizons*, 4(1).

Robinson, N.M. *et al.* (2018) 'Essential principles to guide monitoring of threatened biodiversity', in *Monitoring threatened species and ecological communities*. CSIRO Publishing, pp. 427–438. Available at: https://researchportalplus.anu.edu.au/en/publications/essential-principles-to-guide-monitoring-of-threatened-biodiversi (Accessed: 13 November 2024).

Roe, P. *et al.* (2021) 'The Australian acoustic observatory', *Methods in Ecology and Evolution*, 12(10), pp. 1802–1808. Available at: https://doi.org/10.1111/2041-210x.13660.

Rogers, A.M. *et al.* (2020) 'Noisy neighbours and myna problems: Interaction webs and aggression around tree hollows in urban habitats', *Jounral of Applied Ecology*, 57(10), pp. 1891–1901.

Roh, Y., Heo, G. and Whang, S.E. (2019) 'A survey on data collection for machine learning: a big data-ai integration perspective', *IEEE Transactions on Knowledge and Data Engineering*, 33(4), pp. 1328–1347. Available at: https://doi.org/10.1109/tkde.2019.2946162.

Rollins, L.A., Svedin, N. and Griffith, S. (2012) 'The role of the Orid Arid Intrusion in the historical and contemporary genetic division of long-tailed finch subspecies in northern Australia', *Ecology and Evolution*, 2(6), pp. 1208–1219. Available at: https://doi.org/10.1002/ece3.259.

Ross, J. (2009) *Sustainable development of northern Australia : a report to government from the Northern Australia Land and Water Taskforce*. Canberra: Department of Infrastructure, Transport, Regional Development and Local Government.

Ross, S. *et al.* (2023) 'Passive acoustic monitoring provides a fresh perspective on fundamental ecological questions', *Functional Ecology*, 37. Available at: https://doi.org/10.1111/1365-2435.14275.

Ross, S.R. *et al.* (2023) 'Passive acoustic monitoring provides a fresh perspective on fundamental ecological questions', *Functional Ecology*, 37(4), pp. 959–975. Available at: https://doi.org/10.1111/1365-2435.14275.

Rotenberry, J.T. and Wiens, J.A. (1998) 'FORAGING PATCH SELECTION BY SHRUBSTEPPE SPARROWS', *Ecology*, 79(4), pp. 1160–1173. Available at: https://doi.org/10.1890/0012-9658(1998)079%255B1160:FPSBSS%255D2.0.CO;2.

Rowcliffe, M. (2023) *activity: Animal Activity Statistics*. Available at: https://cran.r-project.org/web/packages/activity/index.html.

Rowe, C. and Skelhorn, J. (2004) 'Avian psychology and communication', *Proceedings of the Royal Society of London. Series B: Biological Sciences*, 271(1547), pp. 1435–1442. Available at: https://doi.org/10.1098/rspb.2004.2753.

Ruckstuhl, K. and Neuhaus, P. (2006) 'Sexual segregation in vertebrates: Ecology of the two sexes', *Sexual Segregation in Vertebrates: Ecology of the Two Sexes*, pp. 1–488. Available at: https://doi.org/10.1017/CBO9780511525629.

Ruff, Z.J. *et al.* (2020) 'Automated identification of avian vocalizations with deep convolutional neural networks', *Remote Sensing in Ecology and Conservation*, 6(1), pp. 79–92. Available at: https://doi.org/10.1002/rse2.125.

Ruff, Z.J. *et al.* (2021) 'Workflow and convolutional neural network for automated identification of animal sounds', *Ecological Indicators*, 124, p. 107419. Available at: https://doi.org/10.1016/j.ecolind.2021.107419.

Russell-Smith, J. and Edwards, A.C. (2006) 'Seasonality and fire severity in savanna landscapes of monsoonal northern Australia', *International Journal of Wildland Fire*, 15(4), pp. 541–550.

Russell-Smith, J., Start, T. and Woinarski, J. (2001) 'Effects of Fire in The Landscape', in *Savanna Burning - Understanding and using fire in northern Australia*. Darwin: Australia: Tropical Savannas Cooperative Research Centre.

Russell-Smith, J. and Yates, C. (2007) 'Australian Savanna Fire Regimes: Context, Scales, Patchiness', *Fire Ecology*, 3, pp. 48–63. Available at: https://doi.org/10.4996/fireecology.0301048.

Ryan-Colton, E. *et al.* (2024) 'Long-term and landscape impacts of buffel grass on arid plant communities: Ecosystem shifts and acceleration by fire', *Ecosphere*, 15(10), p. e70033. Available at: https://doi.org/10.1002/ecs2.70033.

Salema, C.A., Gale, G.A. and Bumrungsri, S. (2018) 'Nest-site selection by Common Green Magpie (Cissa chinensis) in a tropical dry evergreen forest, northeast Thailand', *The Wilson Journal of Ornithology*, 130(1), pp. 256–261. Available at: https://doi.org/10.1676/17-069.1.

Sankupellay, M. and Konovalov, D. (2018) 'Bird call recognition using deep convolutional neural network, ResNet-50', in *Proceedings of Acoustics*. Available at: https://doi.org/10.13140/RG.2.2.31865.31847.

Santos, S. and Lumeij, J. (2007) 'Comparison of multiple-angle spectrometry of plumage versus individual feathers for the assessment of sexual dichromatism in the long-tailed finch (Poephila acuticauda)', *Journal of Ornithology*, 148, pp. 281–291. Available at: https://doi.org/10.1007/s10336-007-0131-7.

Sarmiento, G. (1992) 'Adaptive strategies of perennial grasses in South American savannas', *Journal of Vegetation Science*, 3(3), pp. 325–336. Available at: https://doi.org/10.2307/3235757.

Sattler, P. and Williams, R. (1999) *The conservation status of Queensland's bioregional ecosystems*. Brisbane: Environmental Protection Agency.

Sattler, P.S. (1999) *The conservation status of Queensland's bioregional ecosystems*. Environmental Protection Agency, Queensland Government.

Saunders, D. *et al.* (2022) 'Radio-tracking wildlife with drones: a viewshed analysis quantifying survey coverage across diverse landscapes', *Wildlife Research*, 49(1), pp. 1–10. Available at: https://doi.org/10.1071/wr21033.

Sauter, A. *et al.* (2006) 'Does optimal foraging theory explain why suburban Florida scrub-jays (Aphelocoma coerulescens) feed their young human-provided food?', *Behavioral Ecology and Sociobiology*, 60, pp. 465–474.

Scardamaglia, R.C. *et al.* (2022) 'Automated radio tracking provides evidence for social pair bonds in an obligate brood parasite', *Ibis*, 164(4), pp. 1180–1191. Available at: https://doi.org/10.1111/ibi.13086.

Schaefer, H.M., Schmidt, V. and Bairlein, F. (2003) 'Discrimination abilities for nutrients: which difference matters for choosy birds and why?', *Animal behaviour*, 65(3), pp. 531–541.

Scheele, B.C. *et al.* (2018) 'How to improve threatened species management: An Australian perspective', *Journal of Environmental Management*, 223, pp. 668–675. Available at: https://doi.org/10.1016/j.jenvman.2018.06.084.

Schidelko, K. *et al.* (2013) 'Continental shelf as potential retreat areas for Austral-Asian estrildid finches (Passeriformes: Estrildidae) during the Pleistocene', *Journal of Avian Biology*, 44(2), pp. 121–132. Available at: https://doi.org/10.1111/j.1600-048x.2012.05743.x.

Schidelko, K., Stiels, D. and Rödder, D. (2011) 'Historical stability of diversity patterns in African estrildid finches (Aves: Estrildidae)?', *Biological Journal of the Linnean Society*, 102(2), pp. 455–470.

Schlesinger, C., White, S. and Muldoon, S. (2013) 'Spatial pattern and severity of fire in areas with and without buffel grass (enchrus ciliaris) and effects on native vegetation in central Australia', *Austral Ecology*, 38(7), pp. 831–840. Available at: https://doi.org/10.1111/aec.12039.

Schluter, D. (1982) 'Seed and Patch Selection by Galapagos Ground Finches: Relation to Foraging Efficiency and Food Supply', *Ecology*, 63(4), pp. 1106–1120. Available at: https://doi.org/10.2307/1937248.

Schodde, R. and Mason, I.J. (1999) *Directory of Australian Birds: Passerines: Passerines*. CSIRO PUBLISHING.

Schodde, R. and Mason, I. J. (1999) *Directory of Australian Birds: Passerines: Passerines*. CSIRO PUBLISHING (Book, Whole).

Schofield, L.N. *et al.* (2018) 'Using automated radio telemetry to quantify activity patterns of songbirds during stopover', *The Auk: Ornithological Advances*, 135(4), pp. 949–963. Available at: https://doi.org/10.1642/auk-17-229.1.

van de Schoot, R. *et al.* (2021) 'Bayesian statistics and modelling', *Nature Reviews Methods Primers*, 1(1), p. 1.

Segelbacher, G. *et al.* (2010) 'Review: Applications of landscape genetics in conservation biology: concepts and challenges', *Conservation Genetics*, 11, pp. 375–385. Available at: https://doi.org/10.1007/s10592-009-0044-5.

Segura, A. and Acevedo, P. (2021) 'Influence of habitat and food resource availability on Common raven nest site selection and reproductive success in Mediterranean forests', *Birds*, 2(3), pp. 302–313.

Sekercioglu, C.H. *et al.* (2008) 'Climate change, elevational range shifts, and bird extinctions', *Conservation biology*, 22(1), pp. 140–150. Available at: https://doi.org/10.1111/j.1523-1739.2007.00852.x.

Serrano-Davies, E. *et al.* (2022) 'Biased adult sex ratios in Western Europe populations of Little Bustard Tetrax tetrax as a potential warning signal of unbalanced mortalities', *Bird Conservation International*, 33(e40), pp. 1–8.

Settles, B. and Craven, M. (2008) 'An analysis of active learning strategies for sequence labeling tasks', in *Proceedings of the 2008 conference on empirical methods in natural language processing*, pp. 1070–1079. Available at: https://doi.org/10.3115/1613715.1613855.

Seymour, C.L. *et al.* (2010) 'Twenty years of rest returns grazing potential, but not palatable plant diversity, to Karoo rangeland, South Africa', *Journal of Applied Ecology*, 47(4), pp. 859–867. Available at: https://doi.org/10.1111/j.1365-2664.2010.01833.x.

Shackelford, C.E. and Conner, R.N. (1997) 'Woodpecker Abundance and Habitat Use in Three Forest Types in Eastern Texas', *The Wilson Bulletin*, 109(4), pp. 614–629.

Shaffer, S., Weimerskirch, H. and Costa, D. (2001) 'Functional significance of sexual dimorphism in Wandering Albatrosses, Diomedea exulans', *Functional Ecology*, 15, pp. 203–210. Available at: https://doi.org/10.1046/j.1365-2435.2001.00514.x.

Sheard, C. *et al.* (2020) 'Ecological drivers of global gradients in avian dispersal inferred from wing morphology', *Nature communications*, 11(1), p. 2463. Available at: https://doi.org/10.1038/s41467-020-16313-6.

Shephard, M. (1989) 'Aviculture in Australia: Keeping and Breeding Aviary Birds.', *Black Cockatoo Press* [Preprint].

Shephard, M., Pridham, A. and Forshaw, J.M. (2012) *Grassfinches in Australia*. Melbourne: CSIRO Publishing. Available at: https://doi.org/10.1071/9780643107878.

Shi, B. *et al.* (2019) 'Semi-supervised acoustic event detection based on tri-training', in *ICASSP 2019-2019 IEEE International Conference on Acoustics, Speech and Signal Processing*. IEEE, pp. 750–754. Available at: https://doi.org/10.1109/icassp.2019.8683710.

Shine, R. (1989) 'Ecological Causes for the Evolution of Sexual Dimorphism: A Review of the Evidence', *The Quarterly Review of Biology*, 64(4), pp. 419–61.

Sica, Y.V. *et al.* (2020) 'Wetland bird response to habitat composition and configuration at multiple spatial scales', *Wetlands*, 40(6), pp. 2513–2525.

Silcock, J. and Fensham, R. (2018) 'Using evidence of decline and extinction risk to identify priority regions, habitats and threats for plant conservation in Australia', *Australian Journal of Botany*, 66, p. 541. Available at: https://doi.org/10.1071/BT18056.

Silcock, R.G. *et al.* (2018) 'Spring fire effects on two Aristida/Bothriochloa native pastures in central Queensland, Australia', *The Rangeland Journal*, 40(5), pp. 485–500.

Silva, I. *et al.* (2022) 'Autocorrelation-informed home range estimation: A review and practical guide', *Methods in Ecology and Evolution*, 13(3), pp. 534–544. Available at: https://doi.org/10.1111/2041-210X.13786.

Simon, B.K. and Alfonso, Y. (2011) *AusGrass2 Grasses of Australia*.

Simons, T.R. *et al.* (2007) 'Experimental analysis of the auditory detection process on avian point counts', *The Auk*, 124(3), pp. 986–999. Available at: https://doi.org/10.1642/0004-8038(2007)124%5B986:eaotad%5D2.0.co;2.

Siriwardena, G.M., Calbrade, N.A. and Vickery, J.A. (2008) 'Farmland birds and late winter food: does seed supply fail to meet demand?', *Ibis*, 150(3), pp. 585–595. Available at: https://doi.org/10.1111/j.1474-919X.2008.00828.x.

Smetzer, J. (2018) *Tracking migratory bird movements in the Gulf of Maine with automated radio telemetry and stable hydrogen isotope markers*. University of Massachusetts Amherst.

Smetzer, J.R. *et al.* (2021) 'Automated telemetry reveals post-reintroduction exploratory behavior and movement patterns of an endangered corvid, ʻAlalā (*Corvus hawaiiensis*) in Hawaiʻi, USA', *Global Ecology and Conservation*, 26, p. e01522. Available at: https://doi.org/10.1016/j.gecco.2021.e01522.

Smith, K.T. *et al.* (2019) 'Reconstructing Greater Sage-Grouse chick diets: Diet selection, body condition, and food availability at brood-rearing sites', *The Condor*, 121(1), pp. 1–12. Available at: https://doi.org/10.1093/condor/duy012.

Socher, S.A. *et al.* (2013) 'Interacting effects of fertilization, mowing and grazing on plant species diversity of 1500 grasslands in Germany differ between regions', *Basic and Applied Ecology*, 14(2), pp. 126–136.

Soobramoney, S. and Perrin, M.R. (2007) 'The effect of bill structure on seed selection and handling ability of five species of granivorous birds', *Emu-Austral Ornithology*, 107(3), pp. 169–176.

Southwood, T.R. (1977) 'Habitat, the templet for ecological strategies?', *Journal of animal ecology*, 46(2), pp. 337–365.

Southwood, T.R.E. (1988) 'Habitat, the templet for ecological strategies?', *Current Contents/Agriculture Biology & Environmental Sciences*, (16), pp. 14–14.

Sripathy, K.V. and Groot, S.P. (2023) 'Seed development and maturation', in *Seed science and technology: Biology, production, quality*. Springer Nature Singapore Singapore, pp. 17–38. Available at: https://library.oapen.org/bitstream/handle/20.500.12657/61883/1/978-981-19-5888-5.pdf#page=32 (Accessed: 11 March 2025).

Stevens, M. and Cuthill, I.C. (2007) 'Hidden Messages: Are Ultraviolet Signals a Special Channel in Avian Communication?', *BioScience*, 57(6), pp. 501–507. Available at: https://doi.org/10.1641/B570607.

Stewart, K.A. (2019) 'Understanding the effects of biotic and abiotic factors on sources of aquatic environmental DNA', *Biodiversity and Conservation*, 28(5), pp. 983–1001.

Stock, W.D. *et al.* (2013) 'Pine as fast food: foraging ecology of an endangered cockatoo in a forestry landscape', *Plos One*. Edited by C. Mettke-Hofmann, 8(4), pp. e61145–e61145. Available at: https://doi.org/10.1371/journal.pone.0061145.

Stone, Z.L., Tasker, E. and Maron, M. (2019) 'Patterns of invertebrate food availability and the persistence of an avian insectivore on the brink', *Austral Ecology*, 44(4), pp. 680–690. Available at: https://doi.org/10.1111/aec.12713.

Stowell, D. *et al.* (2019) 'Automatic acoustic detection of birds through deep learning: the first Bird Audio Detection challenge', *Methods in Ecology and Evolution*, 10(3), pp. 368–380. Available at: https://doi.org/10.1111/2041-210x.13103.

Stowell, D. (2022) 'Computational bioacoustics with deep learning: a review and roadmap', *PeerJ*, 10, pp. e13152–e13152. Available at: https://doi.org/10.7717/peerj.13152.

Sugai, L.S.M., Silva, T.S.F., Ribeiro, J.W., Jr, *et al.* (2019) 'Terrestrial Passive Acoustic Monitoring: Review and Perspectives', *BioScience*, 69(1), pp. 15–25. Available at: https://doi.org/10.1093/biosci/biy147.

Sugai, L.S.M., Silva, T.S.F., Ribeiro Jr, J.W., *et al.* (2019) 'Terrestrial passive acoustic monitoring: review and perspectives', *BioScience*, 69(1), pp. 15–25. Available at: https://doi.org/10.1093/biosci/biy147.

Sulaiman, B. *et al.* (2023) 'Radio map generation approaches for an RSSI-based indoor positioning system', *Systems and soft computing*, 5, p. 200054. Available at: https://doi.org/10.1016/j.sasc.2023.200054.

Sutherland, G.D. *et al.* (2000) 'Scaling of natal dispersal distances in terrestrial birds and mammals', *Conservation ecology*, 4(1).

Sutherland, W. (2004) 'Diet and foraging behavior', in, pp. 233–250. Available at: https://doi.org/10.1093/acprof:oso/9780198520863.003.0010.

Symstad, A.J. and Jonas, J.L. (2011) 'Incorporating biodiversity into rangeland health: plant species richness and diversity in Great Plains grasslands', *Rangeland Ecology & Management*, 64(6), pp. 555–572.

Székely, T. *et al.* (2014) 'Sex-biased survival predicts adult sex ratio variation in wild birds', *Proceedings: Biological Sciences*, 281(1788), pp. 1–8.

Tallmon, D.A. *et al.* (2010) 'When are genetic methods useful for estimating contemporary abundance and detecting population trends?', *Molecular Ecology Resources*, 10(4), pp. 684–692.

Tang, L.S. *et al.* (2016) 'Genetic structure and diversity of the black-throated finch (*Poephila cincta*) across its current range', *Australian Journal of Zoology*, 64(6), pp. 375–384. Available at: https://doi.org/10.1071/ZO16073.

Tang, L.S. *et al.* (2017) 'Genetic structure and diversity of the black-throated finch (Poephila cincta) across its current range.', *Australian Journal of Zoology*, 64(6), pp. 375–384.

Tang, S.L. (2016) *Conservation genetics of granivorous birds in a heterogenous landscape: the case of the black-throated finch (Poephila cincta)*. PhD Thesis. James Cook University.

Taylor, P.D. *et al.* (2017) 'The Motus Wildlife Tracking System: a collaborative research network to enhance the understanding of wildlife movement', *Avian Conservation and Ecology*, 12(1). Available at: https://doi.org/10.5751/ACE-00953-120108.

Teixeira, D. *et al.* (2022) 'Fledge or fail: Nest monitoring of endangered black-cockatoos using bioacoustics and open-source call recognition', *Ecological Informatics*, 69, p. 101656. Available at: https://doi.org/10.1016/j.ecoinf.2022.101656.

Terrestrial Ecosystems Research Network (2020) *Annual Fire Scars - Landsat*. Available at: http://data.auscover.org.au/xwiki/bin/view/Product+pages/Landsat+fire+scars+Queensland.

Thalinger, B. *et al.* (2021) 'A validation scale to determine the readiness of environmental DNA assays for routine species monitoring', *Environmental DNA*, 3(4), pp. 823–836.

Thomas, B., Holland, J.D. and Minot, E.O. (2011) 'Wildlife tracking technology options and cost considerations', *Wildlife Research*, 38(8), pp. 653–663. Available at: https://doi.org/10.1071/wr10211.

Thomas, L. *et al.* (2010) 'Distance software: design and analysis of distance sampling surveys for estimating population size', *Journal of Applied Ecology*, 47(1), pp. 5–14. Available at: https://doi.org/10.1111/j.1365-2664.2009.01737.x.

Thrash, I. *et al.* (1991) 'The impact of the provision of water for game on the woody vegetation around a dam in the Kruger National Park', *Koedoe*, 34, pp. 121–130.

Tidemann, S.C. *et al.* (1992) 'Comparison of the breeding sites and habitat of two hole-nesting estrildid finches, one endangered, in northern Australia', *Journal of Tropical Ecology*, 8(4), pp. 373–388. Available at: https://doi.org/10.1017/s0266467400006702.

Tidemann, S.C. (1996) 'Causes of the decline of the Gouldian Finch Erythrura gouldiae', *Bird Conservation International*, 6(1), pp. 49–61.

Tischler, M., Dickman, C.R. and Wardle, G.M. (2013) 'Avian functional group responses to rainfall across four vegetation types in the Simpson Desert, central Australia', *Austral Ecology*, 38(7), pp. 809–819. Available at: https://doi.org/10.1111/aec.12065.

Titulaer, M. *et al.* (2018) 'Seed size, bill morphology, and handling time influence preferences for native vs. nonnative grass seeds in three declining sparrows', *The Wilson Journal of Ornithology*, 130(2), pp. 445–456.

Tizard, I. (2004) 'Salmonellosis in wild birds', 13(2), pp. 50–66.

Tognetti, P.M. and Chaneton, E.J. (2012) 'Invasive exotic grasses and seed arrival limit native species establishment in an old-field grassland succession', *Biological Invasions*, 14, pp. 2531–2544.

Tomaszycki, M.L. *et al.* (2009) 'Sexual differentiation of the zebra finch song system: potential roles for sex chromosome genes', *BMC Neuroscience*, 10(1), p. 24. Available at: https://doi.org/10.1186/1471-2202-10-24.

Tothill, J.C. (1977) 'Flowering phenology of some native perennial tropical grasses from north-eastern Australia', *Australian Journal of Ecology*, 2(2), pp. 199–205. Available at: https://doi.org/10.1111/j.1442-9993.1977.tb01137.x.

Towsey, M. *et al.* (2012) 'A toolbox for animal call recognition', *Bioacoustics*, 21(2), pp. 107–125. Available at: https://doi.org/10.1080/09524622.2011.648753.

Tredennick, A.T. *et al.* (2021) 'A practical guide to selecting models for exploration, inference, and prediction in ecology', *Ecology*, 102(6), p. e03336. Available at: https://doi.org/10.1002/ecy.3336.

Truong, A. *et al.* (2019) 'Towards automated machine learning: Evaluation and comparison of AutoML approaches and tools', in *2019 IEEE 31st international conference on tools with artificial intelligence*. IEEE, pp. 1471–1479. Available at: https://doi.org/10.1109/ictai.2019.00209.

Turgeon, P., Van Wilgenburg, S. and Drake, K. (2017) 'Microphone variability and degradation: implications for monitoring programs employing autonomous recording units', *Avian Conservation and Ecology*, 12(1). Available at: https://doi.org/10.5751/ace-00958-120109.

Tyson, C., Loning, H., *et al.* (2024) 'Constant companions: wild zebra finch pairs display extreme spatial cohesion', *Biology Letters*, 20. Available at: https://doi.org/10.1098/rsbl.2024.0519.

Tyson, C., Fragueira, R., *et al.* (2024) 'Fingerprint localisation for fine-scale wildlife tracking using automated radio telemetry', *bioRxiv*, p. 2024.02.15.580447. Available at: https://doi.org/10.1101/2024.02.15.580447.

Vabalas, A. *et al.* (2019) 'Machine learning algorithm validation with a limited sample size', *PLoS One*, 14(11), p. e0224365. Available at: https://doi.org/10.1371/journal.pone.0224365.

Valera, F. *et al.* (2005) 'Dietary specialization on high protein seeds by adult and nestling serins', *The Condor*, 107(1), pp. 29–40.

Van Coller, H. *et al.* (2018) 'Herbaceous responses to herbivory, fire and rainfall variability differ between grasses and forbs', *South African Journal of Botany*, 119, pp. 94–103.

Van Engelen, J.E. and Hoos, H.H. (2020) 'A survey on semi-supervised learning', *Machine Learning*, 109(2), pp. 373–440. Available at: https://doi.org/10.1007/s10994-019-05855-6.

Van Moorter, B. *et al.* (2016) 'Movement is the glue connecting home ranges and habitat selection', *Journal of Animal Ecology*, 85(1), pp. 21–31. Available at: https://doi.org/10.1111/1365-2656.12394.

Van Rooij, E.P. and Griffith, S.C. (2010) 'Are monomorphic species really sexually indistinguishable: no evidence in wild long-tailed finches (Poephila acuticauda)', *Ethology*, 116(10), pp. 929–940.

Van Rooij, E.P. and Griffith, S.C. (2011) 'Breeding ecology of an Australian estrildid, the Long-tailed Finch (Poephila acuticauda)', *Emu-Austral Ornithology*, 111(4), pp. 297–303.

Vanderduys, E.P. *et al.* (2016a) 'Addressing potential cumulative impacts of development on threatened species: The case of the endangered black-throated finch', *PLoS One*, 11(3). Available at: https://doi.org/10.1371/journal.pone.0148485.

Vanderduys, E.P. *et al.* (2016b) 'Addressing Potential Cumulative Impacts of Development on Threatened Species: The Case of the Endangered Black-Throated Finch', *PLOS ONE*, 11(3). Available at: https://doi.org/10.1371/journal.pone.0148485.

Vazquez, C. *et al.* (2019) 'Comparing diel activity patterns of wildlife across latitudes and seasons: Time transformations using day length', *Methods in Ecology and Evolution*, 10(12), pp. 2057–2066. Available at: https://doi.org/10.1111/2041-210x.13290.

Vegvari, Z. *et al.* (2018) 'Sex-biased breeding dispersal is predicted by social environment in birds', *Ecology and Evolution*, 8, pp. 6483–6491.

Vehtari, A., Gelman, A. and Gabry, J. (2017) 'Practical Bayesian model evaluation using leave-one-out cross-validation and WAIC', *Statistics and Computing*, 27(5), pp. 1413–1432. Available at: https://doi.org/10.1007/s11222-016-9696-4.

Vellinga, W.-P. and Planqué, R. (2015) 'The xeno-canto collection and its relation to sound recognition and classification.', in.

Venables, W. and Ripley, B. (2002) *Modern Applied Statistics with S*. Fourth. New York: Springer. Available at: https://www.stats.ox.ac.uk/pub/MASS4/.

Venables, W.N. and Ripley, B.D. (2013) *Modern applied statistics with S-PLUS*. Springer Science & Business Media.

Verner, J. and Milne, K.A. (1989) 'Coping with sources of variability when monitoring population trends', *Annales Zooloogici Fennici*, 26, pp. 191–199.

Verrico, B. and Preston, J.C. (2024) 'Historic rewiring of grass flowering time pathways and implications for crop improvement under climate change', *New Phytologist*, p. nph.20375. Available at: https://doi.org/10.1111/nph.20375.

Vesk, P.A. and Westoby, M. (2001) 'Predicting plant species' responses to grazing', *Journal of Applied Ecology*, 38(5), pp. 897–909. Available at: https://doi.org/10.1046/j.1365-2664.2001.00646.x.

Visser, M.E. and Gienapp, P. (2019) 'Evolutionary and demographic consequences of phenological mismatches', *Nature ecology & evolution*, 3(6), pp. 879–885.

Volodin, I.A. *et al.* (2015) 'Gender identification using acoustic analysis in birds without external sexual dimorphism', *Avian Research*, 6(1), p. 20. Available at: https://doi.org/10.1186/s40657-015-0033-y.

Voros, J. *et al.* (2017) 'Surveying Europe's Only Cave-Dwelling Chordate Species (Proteus anguinus) Using Environmental DNA', *PLOS ONE* [Preprint]. Available at: https://doi.org/10.1371/journal.pone.0170945.

Wallace, G. *et al.* (2022) 'An automated radiotelemetry system (ARTS) for monitoring small mammals', *Methods in Ecology and Evolution*, 13(5), pp. 976–986. Available at: https://doi.org/10.1111/2041-210x.13794.

Wang, G. *et al.* (2020) 'A multiscale approach to identifying spatiotemporal pattern of habitat selection for red-crowned cranes', *Science of the Total Environment*, 739, p. 139980.

Wang, J. *et al.* (2005) 'Relations between NDVI, grassland production, and crop yield in the central Great Plains', *Geocarto International*, 20(3), pp. 5–11.

Wann, G.T. *et al.* (2019) 'Mismatches between breeding phenology and resource abundance of resident alpine ptarmigan negatively affect chick survival', *Ecology and Evolution*, 9(12), pp. 7200–7212. Available at: https://doi.org/10.1002/ece3.5290.

Waples, R.S. and Do, C. (2008) 'LDNE: a program for estimating effective population size from data on linkage disequilibrium.', *Molecular Ecology Resources*, 8(4), pp. 753–756.

Ward, M.P., Sperry, J.H. and Weatherhead, P.J. (2013) 'Evaluation of automated radio telemetry for quantifying movements and home ranges of snakes', *Journal of Herpetology*, 47(2), pp. 337–345. Available at: https://doi.org/10.1670/12-018.

Weidinger, K. (2002) 'Interactive Effects of Concealment, Parental Behaviour and Predators on the Survival of Open Passerine Nests', *Journal of Animal Ecology*, 71(3), pp. 424–437.

Weier, A., Radford, I.J., *et al.* (2017a) 'Frequent fires reduce the nutritional quality of *Sorghum stipoideum* seed, a keystone food resource for the Gouldian finch (*Erythrura gouldiae*)', *The Rangeland journal*, 39(2), pp. 105–112. Available at: https://doi.org/10.1071/RJ16124.

Weier, A., Radford, I.J., *et al.* (2017b) 'Frequent fires reduce the nutritional quality of Sorghum stipoideum seed, a keystone food resource for the Gouldian finch (Erythrura gouldiae)', *The Rangeland Journal*, 39(2), p. 105. Available at: https://doi.org/10.1071/RJ16124.

Weier, A., Radford, I., *et al.* (2017) 'Frequent fires reeuce the nutritional quality of Sorghum stipoideum seed, a keystone food resource for the Gouldian finch (Erythrura gouldiar)'.

Weier, A. *et al.* (2019) 'Seed set in *Sorghum stipoideum*, and not fire, determines the timing of breeding by Gouldian finches (*Erythrura gouldiae*)', *Austral Ecology*, 44(1), pp. 151–162. Available at: https://doi.org/10.1111/aec.12662.

Wenk, E.H. *et al.* (2024) 'A near-complete dataset of plant growth form, life history, and woodiness for all Australian plants', *Australian Journal of Botany*, 72(4).

Werner, E.E. and Hall, D.J. (1974) 'Optimal foraging and the size selection of prey by the bluegill sunfish (Lepomis macrochirus)', *Ecology*, 55(5), pp. 1042–1052.

Westgate, M.J. *et al.* (2021) 'Synergistic impacts of aggressive species on small birds in a fragmented landscape', *Journal of Applied Ecology*, 58(4), pp. 825–835.

Weston, E.D. *et al.* (2013) 'When do young birds disperse? Tests from studies of golden eagles in Scotland', *BMC Ecology*, 13(1), p. 42. Available at: https://doi.org/10.1186/1472-6785-13-42.

White, G.C. and Burnham, K.P. (1999) 'Program MARK: survival estimation from populations of marked animals', *Bird study*, 46(sup1), pp. S120–S139.

White, J.G. *et al.* (2022) 'Can NDVI identify drought refugia for mammals and birds in mesic landscapes?', *Science of the Total Environment*, 851(Part 2). Available at: https://doi.org/10.1016/j.scitotenv.2022.158318.

Wiens, J.A. (1986) 'Spatial scale and temporal variation in studies of shrubsteppe birds', *Community ecology. Harper and Row, New York*, pp. 154–172.

Wiens, J.A. (1989) *The ecology of bird communities*. Cambridge [England]: Cambridge University Press (Cambridge studies in ecology).

Wiens, J.A., Rotenberry, J.T. and Van Horne, B. (1987) 'Habitat Occupancy Patterns of North American Shrubsteppe Birds: The Effects of Spatial Scale', *Oikos*, 48(2), pp. 132–147.

Wildlife Acoustics (2019) 'Kaleidoscope Pro Analysis Software'.

Williams *et al.* (2020) 'Black-throated finch habitat values promoted by patchy fire', *North Queensland Naturalist*, (50), pp. 38–43.

Williams, P.R. (2002) *The effect of fire regime on tropical savannas of north-eastern Australia: interpreting floristic patterns through critical life events*. James Cook University.

Williams, R.J. *et al.* (1999) 'Fire regime, fire intensity and tree survival in a tropical savanna in northern Australia', *Australian Journal of Ecology*, 24(1), pp. 50–59. Available at: https://doi.org/10.1046/j.1442-9993.1999.00946.x.

Williams, R.J., Woinarski, J. and Andersen, A.N. (2003) 'Fire experiments in northern Australia: contributions to ecological understanding and biodiversity conservation in tropical savannas', *International Journal of Wildland Fire*, 12(4), pp. 391–402.

Willson, M.F. and Harmeson, J.C. (1973) 'Seed preferences and digestive efficiency of cardinals and song sparrows', *The Condor*, 75(2), pp. 225–234.

Windle, J. *et al.* (2009) 'A conservation auction for landscape linkage in the southern Desert Uplands, Queensland', *The Rangeland Journal*, 31(1), pp. 127–135.

WIRES (2020) *WIRES Water for Wildlife Project*. Available at: https://www.wires.org.au/blog/wires-water-for-wildlife-project (Accessed: 7 September 2021).

Withers, P.C., Cooper, C.E. and Larcombe, A.N. (2022) 'Relative water economy is a useful index of aridity tolerance for Australian Poephiline finches', *Birds*, 3(2), pp. 172–183. Available at: https://doi.org/10.3390/birds3020012.

Woinarski, J. *et al.* (2011) 'The disappearing mammal fauna of northern Australia: context, cause, and response', *Conservation letters*, 4(3), pp. 192–201. Available at: https://doi.org/10.1111/j.1755-263X.2011.00164.x.

Woinarski, J., Andersen, A.N. and Murphy, B.P. (2020) 'The tropical savannas of northern Australia', in M.I. Goldstein and D.A. DellaSala (eds) *Encyclopedia of the World's Biomes*. Oxford: Elsevier, pp. 827–834. Available at: https://doi.org/10.1016/B978-0-12-409548-9.12023-8.

Woinarski, J. and Catterall, C. (2004) 'Historical changes in the bird fauna at Coomooboolaroo, northeastern Australia, from the early years of pastoral settlement (1873) to 1999.', *Biological Conservation*, 116(3), pp. 379–401.

Woinarski, J. and Legge, S. (2013) 'The impacts of fire on birds in Australia's tropical savannas', *Emu - Austral Ornithology*, 113(4), pp. 319–352. Available at: https://doi.org/10.1071/MU12109.

Woinarski, J. and Tidemann, S. (1991) 'The bird fauna of a deciduous woodland in the wet-dry tropics of Northern Australia.', *Wildlife Research*, 18(4), pp. 479–500.

Woinarski, J.C. and Legge, S. (2013) 'The impacts of fire on birds in Australia's tropical savannas', *Emu-Austral Ornithology*, 113(4), pp. 319–352.

Woinarski, J.C.Z. (1990) 'Effects of fire on the bird communities of tropical woodlands and open forests in northern Australia', *Australian Journal of Ecology*, 15(1), pp. 1–22. Available at: https://doi.org/10.1111/j.1442-9993.1990.tb01016.x.

Woinarski, J.C.Z. *et al.* (2011) 'The disappearing mammal fauna of northern Australia: context, cause, and response', *Conservation Letters*, 4(3), pp. 192–201. Available at: https://doi.org/10.1111/j.1755-263X.2011.00164.x.

Woinarski, J.C.Z. *et al.* (2017) 'The contribution of policy, law, management, research, and advocacy failings to the recent extinctions of three Australian vertebrate species', *Conservation Biology*, 31(1), pp. 13–23. Available at: https://doi.org/10.1111/cobi.12852.

Woinarski, J.C.Z. and Tidemann, S. (1992) 'Survivorship and some population parameters for the endangered Gouldian Finch Erythrura gouldiae and two other finch species at two sites in tropical northern Australia', *Emu*, 92(1), pp. 33–38.

Wood, C.M. *et al.* (2023) 'Challenges and opportunities for bioacoustics in the study of rare species in remote environments', *Conservation Science and Practice*, p. e12941. Available at: https://doi.org/10.1111/csp2.12941.

Woongal Environmental (2021) *Fire Strategy for Carmichael Mine and associated Offsets*.

Wright, B. *et al.* (2020) 'Buffel grass (Cenchrus ciliaris) eradication in arid central Australia enhances native plant diversity and increases seed resources for granivores', *Applied Vegetation Science*, 24(1). Available at: https://onlinelibrary-wiley-com.ezp01.library.qut.edu.au/doi/full/10.1111/avsc.12533.

Wright, B.R. *et al.* (2021) 'Buffel grass (*Cenchrus ciliaris*) eradication in arid central Australia enhances native plant diversity and increases seed resources for granivores', *Applied Vegetation Science*, 24(1), p. e12533. Available at: https://doi.org/10.1111/avsc.12533.

Wright, B.R. and Clarke, P.J. (2007) 'Fire regime (recency, interval and season) changes the composition of spinifex (Triodia spp.)-dominated desert dunes', *Australian Journal of Botany*, 55(7), pp. 709–724.

Wright, B.R. and Fensham, R.J. (2016) 'Relationships between fire severity and recruitment in arid grassland dominated by the obligate-seeding soft spinifex (Triodia pungens)', *International Journal of Wildland Fire*, 25(12), pp. 1264–1272.

Wright, B.R. and Fensham, R.J. (2018) 'Fire timing in relation to masting: an important determinant of post-fire recruitment success for the obligate-seeding arid zone soft spinifex (Triodia pungens)', *Annals of Botany*, 121(1), pp. 119–128. Available at: https://doi.org/10.1093/aob/mcx136.

Wright, B.R., Franklin, D.C. and Fensham, R.J. (2022) 'The ecology, evolution and management of mast reproduction in Australian plants', *Australian Journal of Botany*, 70(8), pp. 509–530.

Xiao, H. *et al.* (2017) 'How much do we know about the breeding biology of bird species in the world?', *Journal of Avian Biology*, 48(4), pp. 513–518.

Yang, L.-X. *et al.* (2022) 'A Survey of Sex Ratios of Raptors at a Rescue Center in China', *Journal of Raptor Research*, 56(2), pp. 237–244.

Yates, C.P., Edwards, A.C. and Russell-Smith, J. (2008) 'Big fires and their ecological impacts in Australian savannas: size and frequency matters', *International Journal of Wildland Fire*, 17(6), pp. 768–781.

Yibarbuk, D.M. *et al.* (2001) 'Fire Ecology and Aboriginal Land Management in Central Arnhem Land, Northern Australia: A Tradition of Ecosystem Management', *Journal of Biogeography - J BIOGEOGR*, 28, pp. 325–343. Available at: https://doi.org/10.1046/j.1365-2699.2001.00555.x.

Zann, R (1976) 'Distribution, status and breeding of Black-throated Finches Poephila cincta in northern Queensland.', *Emu - Austral Ornithology*, 76(4), pp. 201–206.

Zann, Richard (1976) 'Distribution, status and breeding of Black-throated Finches *Poephila cincta* in northern Queensland', *Emu-Austral Ornithology*, 76(4), pp. 201–206.

Zann, R. and Runciman, D. (1994) 'Survivorship, dispersal and sex ratios of zebra finches Taeniopygia guttata in southeast Australia', *Ibis*, 136(2), pp. 136–143.

Zann, R. and Straw, B. (1984) 'Feeding Ecology and Breeding of Zebra Finches in Farmland in Northern Victoria.', *Wildlife Research*, 11(3), pp. 533–552. Available at: https://doi.org/10.1071/WR9840533.

Zann, R.A. (1972) *Ethology of Poephila grassfinches (Estrildidae)*. PhD Thesis. The University of Queensland. Available at: https://doi.org/10.14264/uql.2014.594.

Zekavat, R. and Buehrer, R.M. (2011) 'Automated wildlife radio tracking', in S.A. (Reza) Zekavat and R.M. Buehrer (eds) *Handbook of Position Location*. 1st edn. Hoboken, NJ, USA: Wiley, pp. 1129–1167. Available at: https://doi.org/10.1002/9781118104750.ch33.

Zhang, L. *et al.* (2019) 'A novel CSI-based fingerprinting for localization with a single AP', *EURASIP Journal on Wireless Communications and Networking*, 2019, pp. 1–14. Available at: https://doi.org/10.1186/s13638-019-1371-y.

Zhang, Q. *et al.* (2020) 'Trait–environment relationships differ between mixed-species flocking and nonflocking bird assemblages', *Ecology*, 101(10), p. e03124. Available at: https://doi.org/10.1002/ecy.3124.

Zhu, X. and Goldberg, A.B. (2009) 'Introduction to semi-supervised learning', *Synthesis lectures on artificial intelligence and machine learning*, 3(1), pp. 1–130. Available at: https://doi.org/10.1007/978-3-031-01548-9_7.

Appendix A Compliance with approval requirements

A.1 Compliance with approval requirements

Multiple documents set requirement for the BTFRP to achieve, these include the:

- Environmental Authority (EA) EPML01470513; and
- Black-throated Finch Management Plan (BTFMP; Eco Logical Australia, 2023) and the Black-throated Finch Research Program (BTFRP) contained within.

Table 9.3-1 identifies how this BTFRP final report achieves the requirements specified within these approved documents.

Table 9.3-1: Responses to relevant conditions, reporting requirements and research objectives identified within the EA (EPML01470513), BTFRP and BTFP Project Plan Section 3.5

#	Requirement	Response
Condition I6 of the EA		
a)	A baseline research program on the specific nesting and feeding requirements of the species [SBTF] that will be undertaken prior to and during Project Stage 1.	The BTFRP included an investigation of nesting and feeding requirements of SBTF. Research findings on the nesting requirements of SBTF are reported in Section 4.1, the feeding site selection of SBTF in Section 4.2 and the diet of SBTF using both visual observations and DNA metabarcoding in Section 5.1.
b)	A baseline research program to establish whether the SBTF at the project site are sedentary, locally migratory or regionally migratory.	The BTFRP included an investigation of the movement patterns and site philopatry of SBTF, which included an assessment of whether SBTF within the project site are sedentary, locally migratory or regionally migratory. Research findings on the species': • Short-term movements (<30 days) from radio tracking are reported in Section 3.1 • Long-term movements (up to 3 years) from colour band and mist-netting recapture data are reported in Section 3.3; and • Long-term movements from genetic data of the regional population are reported in Section 3.2.
c)	A description of how the results of baseline research program are to be used to determine any changes of classification of and/or impact on SBTF habitat.	This document includes recommendations to update management actions following the findings of the BTFRP (Section 8). It is proposed that the recommendations be adopted as part of a broader comprehensive review of the BTFMP.
d)	Details of proposed impacts to SBTF habitat from each Project Stage including impacts from clearing, subsidence, ecological function changes, hydrological changes and weed and pest infestation changes.	Not directly relevant to the BTFRP.

e)	Mitigation measures to be undertaken to avoid, mitigate and manage impact resulting from each stage of the project, including rehabilitation of habitat.	This document includes recommendations to update management actions following the findings of the BTFRP (Section 8), which include SBTF diet and habitat information to assist rehabilitation planning (Section 9.1.3).
f)	Monitoring of watering points that must be conducted for a minimum 3 hour period commencing from 9am, to accurately capture SBTF utilisation of watering points.	While this item is focussed on the SBTF Monitoring Program, it is noted that the BTFRP includes a recommendation to maintain water source watches as part of a revised SBTF monitoring program (Table 9.2-2).
g)	Detailed botanical assessment that must occur at all SBTF sighting locations in the project area to record habitat values at those locations.	The research program included detailed botanical assessments at SBTF foraging locations as well as incorporating data from 900 vegetation monitoring sites undertaken over a five year period, which were used to answer research questions on the SBTF's: • Nesting habitat selection (Section 4.1) • Foraging habitat selection (Section 4.2) • Diet (Section 5.1) • Grass seed phenology (Section 5.2); and • Response of habitat to land management factors (fire and grazing) (Section 6.1).
h)	Detailed surveys that must occur across the mining lease area and approved offset areas and must include information on SBTF movements. The survey method and effort must be sufficient to accurately describe the SBTF home range and detail SBTF resource usage patterns between seasons and years (for up to 10 years) and allow robust management actions to be developed for the maintenance of a viable local SBTF population.	The BTFRP included an investigation of the movement patterns and site philopatry of SBTF, which included an assessment of whether SBTF within the project site are sedentary, locally migratory or regionally migratory. Research findings on the species': • Short-term movements (<30 days) from radio tracking are reported in Section 3.1 • Long-term movements (up to 3 years) from colour band and mist-netting recapture data are reported in Section 3.3; and • Long-term movements from genetic data of the regional population are reported in Section 3.2. The BTFRP included survey effort across the mining lease and approved offset areas, as well as additional survey effort on adjacent properties. The BTFRP defined the SBTF home range (Section 3.1), detailed landscape scale resource use (habitat selection) (Section 3.1) and local scale use of foraging and nesting resources (Section 4.1, 4.2 and 5.1). Furthermore, the BTFRP assessed over 10 of SBTF monitoring data, which was incorporated into recommendations for the update of management and monitoring of the local SBTF population (Section 8).

i)	Survey work that should incorporate the usage of call playback and identify all birds present when SBTF are encountered.	While this item is focussed on the SBTF monitoring program, it is noted that the BTFRP includes a recommended update to the SBTF monitoring program (Section 9.2.2). Call playback surveys have not been included within the proposed monitoring design and are recommended to be removed from the EA. This is due to the low detection rate achieved from call playback surveys in comparison to other methods and potential confounding factors that may affect the response of SBTF to calls (e.g. type of call broadcast, local provenance of the call, and seasonality).
j)	Specific surveys that must be undertaken during the SBTF breeding season and include nest location and assessment of the habitat attributes associated with the breeding locations. The survey method and effort must be sufficient to accurately describe the SBTF breeding requirements with consideration to spatial and temporal variation of resources of up to 10 years.	The BTFRP included targeted surveys of SBTF nest locations (including from radio tracking of SBTF to their nests) and assessed the species' nesting habitat preferences (Section 4.1). Through this work, the BTFRP has defined the preferred habitat attributes that SBTF select to construct nests, which includes nests throughout the year, rather than solely the breeding season (Section 4.1).
k)	Survey and monitoring that must be undertaken by experienced ecologists.	The BTFRP surveys were undertaken by suitably qualified ecologists. Research reported within this report has been led by suitably qualified ecologists with substantial input from a broader research team at Griffith University (Associate Professor Guy Castley and Dr Laura Grogan), Macquarie University (for landscape genetics analyses [Section 3.2] and sex-related variation [Section 3.4]; Dr Skye Davies, Professor Simon Griffith and Professor Adam Stowe. Furthermore, the annual report for the BTFRP has been independently peer reviewed each year since 2020, with recommendations incorporated into the research. Independent peer reviewers have included Bruce Mullins and Associate Professor Ben Allen.

Condition I8 of the EA	
The baseline research program must fund a research project to determine the relationship between water sources, woody habitat and the SBTF food sources within the mining lease area and approved offset areas to determine the inter-relationships among these factors.	The BTFRP included multiple lines of evidence to determine the relationship between water sources, woody habitat and SBTF food sources within the mining lease area and approved offset areas, which included: • Radio-tracking to profile the distance from water that SBTF regularly move (Section 3.1) • Assessment of nesting and foraging locations to determine the relative effect and relationships of selection of these habitats to vegetation attributes (including woody vegetation), ground layer composition (BTF food resources) and distance to water (Sections 4.1, 4.2). • Using 1 km transects and remote cameras to assess how SBTF and other bird species are affected by the distribution of permanent water and utilise new artificial water sources (Section 6.2).

Condition I9 of the EA		
a)	The baseline research program under Condition I6 must establish whether the Ten Mile Bore and surrounds are high value habitat for the species.	The BTFRP has identified habitat attributes associated with high value SBTF habitat (Sections 3.1, 4.1 and 4.2). These habitat values broadly align with silver leaved ironbark woodland (*Eucalyptus melanophloia*; RE 10.5.5a) that has a native grass species rich ground layer, sparse shrubs and minimal exotic plant invasion. Such vegetation communities are common within 4 km of Ten Mile Bore and as such Ten Mile Bore and surrounds are considered high value habitat for SBTF. In addition, the SBTF monitoring program results, which are reported in the Annual Report identify that the largest populations of SBTF within the mining lease and offset areas drink at water sources within 4 km of Ten Mile Bore, notably CWAT301, CWAT 302 and Ten Mile Bore itself.
b)	The baseline research program under Condition I6 must establish management actions to maintain the current SBTF population of Ten Mile Bore and surrounds.	This document includes recommendations to update management actions following the findings of the BTFRP (Section 8). It is proposed that the recommendations be adopted as part of a broader comprehensive review of the BTFMP. Specific to this requirement, the recommendations include management actions to: • Promote key SBTF habitat values in habitat surrounding Ten Mile Bore, through optimal fire and grazing regimes; and • Routine water source inspections to ensure permanent water is available to SBTF.

BTFRP Section 6.2 (Reporting) and BTFP Project Plan Section 3.5 (Annual Reporting)		
1	Annual reports summarising the monitoring and Research Program's plans will be prepared and submitted by Adani on 1 July each year and will be made available to the Commonwealth and Queensland Governments.	A summary of research program activities and preliminary findings were included in the BTFMP Annual Report that was prepared and finalised by 1 July each year and lodged onto Bravus' environmental compliance management system. Between the 2020 dry season and the 2023 wet season, the progress and preliminary findings of each research program survey were detailed in reports that were supplied to DETSI.
2	Where results of research are likely to be of interest to the scientific community and/or the public, Adani will make relevant information available through appropriate means, such as the submission of scientific papers to peer-reviewed journals (e.g. scientific community) or the provision of research findings on the Adani website or within community newsletters.	To date, three scientific papers detailing results of the research program have been published in scientific journals (van Osta *et al.*, 2023, 2024b, 2024a). Scientific papers detailing other key findings of the research program are currently in preparation for submission to journals in 2025.
3	Submission of all necessary reports, notices and advices as required by the Department of Agriculture and Fisheries (DAF) Animal Ethics Committee.	Animal Ethics notifications, under the approved Animal Ethics Committee Application CA 2020/07/1392 were lodged for all relevant surveys.
4	Submission of an annual report and return of all data collected under the Scientific Purposes Permit under the *Nature Conservation Act 1992*.	An annual return for the Scientific Purposes Permit under which the Project was conducted (permit number WA0025814), was submitted annually during the research program.
5	Adani will share the findings of the research with the BTF Recovery Team for use and updating of the revised BTF Recovery Plan, of which a revised version is currently being prepared.	A summary of research program activities and preliminary findings were included in the BTFMP Annual Report that was prepared and finalised by 1 July each year and lodged onto Bravus' environmental compliance management system. Between the 2020 dry season and the 2023 wet season, the progress and findings of each research program survey were detailed in reports that were supplied to DETSI.
6	Adani will also provide research findings and data will be provided to DES for inputs to the BTF Bioregional Management Plan.	
7	Provide preliminary research findings.	
8	Identify progress against the project schedule and potential delays.	Progress against research activities identified within the project schedule (Section 4 of the BTFRP Project Plan) and research agenda for future reporting periods, was presented within each of the BTFMP Annual Reports. This final BTFRP Report seeks to complete the research program and identify requirements for ongoing management and monitoring of SBTF.
9	Identify planned research activities yet to be completed with an updated project schedule.	

10	Provide amendments to the Research Program and justification.	The BTFRP and BTFRP Project Plan were revised during the reporting period to include adaptive management updates identified during the first year of the BTFRP. The most recent update to the BTFRP (version 6a) and BTFRP Project Plan (version 2d) were approved by DES in December 2020. The BTFMP was reviewed between 2021 and 2023, with some changes based on recommendations from the research team. The most recent update to the BTFMP (version 9c) was approved by DCCEWW and DES in May 2023. This document includes recommendations to update management actions following the findings of the BTFRP (Section 8). It is proposed that the recommendations be adopted as part of a broader comprehensive review of the BTFMP.
11	Updates to relevant management plans/programs through Adani's adaptive management framework, including but not limited to the: • BTF Management Plan • BTF Monitoring Program • Moray Downs West Offset Area Management Plan; and • Species Management Plan Carmichael Coal Mine and Offsite Infrastructure.	
BTFRP Section 4 Objectives of the Proposed Research Methods		
Aim 1	Determine spatial and temporal variation in the movement patterns and home range of SBTF in the Project Area.	The BTFRP included an investigation of the movement patterns and site philopatry of SBTF, which included an assessment of whether SBTF within the project site are sedentary, locally migratory or regionally migratory. The research program included investigation of the SBTF's • Short-term movements (<30 days), home range and landscape scale habitat selection (including seasonal effects) using radio tracking (Section 3.1) • Long-term movements (up to 3 years), including dispersal patterns using colour band and mist-netting recapture data (Section 3.3); and • Long-term movements, genetic connectivity of sub-populations and effective population sizes estimated using genetic data (Section 3.2).

Aim 2	Identify and describe the breeding locations of SBTF in the Project area. Describe the characteristics of foraging habitat utilised by SBTF within the Project area, spatially and temporally. Characterise the differences between identified SBTF habitat and similar locations which are not used by SBTF. Conduct a detailed botanical assessment at all SBTF sighting locations, to determine the characteristics that make the habitat suitable. Establish whether Ten Mile Bore and surrounds are high value habitat for SBTF. Determine the characteristics of Ten Mile Bore and surrounds that contribute to its frequency of use and compare this with other waterholes and woodlands, through equivalent frequency monitoring.	The BTFRP included assessments of nest and foraging site selection against a comprehensive range of potential explanatory variables (Sections 3.1 and 4.1), which included: • Landscape factors, such as distance to water, elevation, Regional Ecosystem and normalised difference vegetation index; and • Site scale factors, such as individual covers of native grass species, species richness variables and vegetation composition variables. Furthermore, tree-scale variables were included within the nest-site selection model to account for factors such as tree species and height in the fine-scale selection of nest sites. The statistical analyses detailed in Sections 3.1 and 4.1 were used to identify the key habitat values that drive SBTF habitat selection. These habitat values broadly align with silver leaved ironbark woodland (*Eucalyptus melanophloia*; RE 10.5.5a) that has a native grass species rich ground layer, sparse shrubs and minimal exotic plant invasion. Such vegetation communities are common within 4 km of Ten Mile Bore and as such Ten Mile Bore and surrounds are considered high value habitat for SBTF. In addition, SBTF monitoring program results, which are reported in the Annual Report identify that the largest populations of SBTF within the mining lease and offset areas drink at water sources within 4 km of Ten Mile Bore, notably CWAT301, CWAT 302 and Ten Mile Bore itself.
Aim 2 cont.	Determine the population substructures including age structure.	The BTFRP included population genetics analysis of 158 SBTF captured from within the mining lease and offset areas as well as on Lignum and Hyde Park Stations. Genetic analyses presented in Section 3.2 characterised the structure of the SBTF population within and adjacent to the Project Area. Specifically, two sub-populations were identified, which comprised a northern and southern sub-population. Furthermore, all SBTF captured through mist-netting were aged (as juvenile, immature or adult) and the presence of juveniles have been noted in all SBTF sighting data collected throughout the BTFRP. Age data have been used to assess age related dispersal from banding data (Section 3.3).

Aim 2 cont.	Identify critical areas to manage SBTF populations and habitat in different seasons.	The BTFRP included a radio-tracking program of SBTF that included nine surveys over three years, with surveys aligned to key seasons relevant to SBTF (early wet, wet and dry seasons; Section 3.1). The radio tracking data informed a habitat selection model, which included an assessment of season differences in the habitat selection of SBTF. While a slight diversification of habitat use was identified, the results identified that broad similarities in the habitat preferences of SBTF among seasons (Section 3.1). However, the foraging habitat analysis identified that at a finer, local habitat selection scale, SBTF preferentially forage in areas of ground cover productivity (measured by normalised difference vegetation index), which are likely to be largely affected by local site topography, soil and ground cover composition (Section 4.2). While SBTF may adjust their daily movement patterns in response to such localised area of high productivity, there is no strong indication of broad-scale seasonal shifts in the home ranges or habitat use of SBTF (Section 3.1). Therefore, considering critical areas of SBTF habitat at a larger landscape scale is appropriate (e.g. areas of silver-leaved ironbark woodland with a richness of native tussock grasses and low shrub cover).
Aim 3	Collect further data on the dietary requirements and foraging preferences of SBTF within the Project area. Identify which plants species are most critical for the provision of seeds for SBTF throughout the year. Identify critical sources of seeds at critical bottleneck periods when the availability of seeds is low.	The BTFRP included multiple lines of evidence to understand the diet of SBTF within the Project Area, including: • Visual observation of SBTF foraging on grass seed inflorescences throughout the research period; and • DNA metabarcoding of crop and faecal samples that were collected over four seasonal surveys. The diet and foraging preferences of SBTF are characterised in Section 5.1. Included within this is an assessment of the seasonal shifts in SBTF diet, which provides information on the diet of SBTF when seeds are readily available (wet season) and when seed availability is low (dry season).

Aim 3 cont.	Identify patterns in seed availability throughout the year, and the factors affecting it, particularly during times of seed scarcity. Determine seed phenology, temporal continuity and shortfall and the relationship of these factors to grazing. Determine how vegetation dynamics are affected by environmental and edaphic factors. Identify which environmental and edaphic factors affect seed quality, quantity and spatial distribution, local variation in distribution and interspecific competition	Grass phenology and seed availability data were collected during all BTFRP habitat assessments undertaken from late 2022 to 2024. The patterns of seed availability and grass phenology are examined in Section 5.2, which includes an assessment of the effects of soil moisture, rainfall and season on seed phenology and availability. Furthermore, the effect of grazing on grass species cover, as well as consideration of the effects of grazing on seed availability is included within Section 6.1.
Aim 4	Understand the relative effects of grazing, fire, water distribution and their interactions on SBTF habitat, and the current spatial and temporal variation and impacts of these factors on SBTF. Determine the effect of cattle grazing on SBTF habitat by undertaking small scale experimental trials to understand how absent, low and high cattle stocking densities and different strategies effect ground cover and grass composition (positively or negatively). Determine effective management that will reduce the likelihood of exotic perennial pasture species spreading in the Project Area. Determine the effects of various fire regimes on the quality of SBTF habitat, and the best fire regimes to enhance or protect existing high quality habitat, under different weather scenarios. Determine spatial and temporal variation in floristic composition and how management actions limit or promote SBTF food availability.	The BTFRP included a land management trial that aimed to understand the relative effects of grazing, fire and additional water sources on SBTF habitat and the SBTF population. The land management trials included; • Experimental trials of different cattle stocking rates and fire regimes. • Addition and monitoring of 6 temporary raised artificial water sources and 3 cattle troughs. • Monitoring of 900 habitat monitoring sites over the research period to understand the response of key SBTF habitat attributes to these factors. The effects of fire and grazing on SBTF habitat are presented in Section 6.1 and include the effect of different fire and grazing regimes on: • Native ground cover components • Exotic perennial pasture species (e.g. buffel grass) • Floristic composition (e.g. changes in grass, forb and shrub species composition); and • Species richness among different vegetation components. Through the research presented in Section 6.1, the BTFRP identifies optimal fire regimes and stocking rates to promote SBTF habitat quality. The practical applications of fire and stocking rate regimes to the Project Area are detailed in the recommendations in Section 9.1.1, which include consideration of how the burning program may be adapted under different weather scenarios (e.g. drought vs successive good wet seasons).

Aim 4 cont.	Determine the influence of establishing new artificial water sources on the utilisation of adjacent habitat by SBTF, via cautious small-scale experimental trials. Investigate whether livestock exclusion from some water sources provides benefits to SBTF.	The BTFRP included the addition and monitoring of 6 temporary raised artificial water sources and 3 cattle troughs, with the response of SBTF to these additional water sources assessed in Section 6.2. Further to this, detailed habitat monitoring, radio tracking (Section 3.1) and analysis of SBTF foraging (Section 5.1) and nesting (Section 4.1) sites has built an understanding of the relationship of SBTF with water sources and the potential benefits of new artificial water sources for SBTF, particularly within the offset areas. Recommendations for the provision of additional water sources within the Project Area are provided in Section 9.1.2 and the optimal design of water sources in Section 6.3.
Aim 4 cont.	Use the information gained in these experimental trials to guide management actions that will maintain or enhance SBTF habitat across the Project area, and in key, high priority SBTF populations and locations (e.g. Ten Mile Bore and surrounds).	This document includes recommendations to update management actions following the findings of the BTFRP (Section 8). It is proposed that the recommendations be adopted as part of a broader comprehensive review of the BTFMP. Specific to this requirement, the recommendations include management actions to: • Promote key SBTF habitat values throughout the Project Area, including in high priority habitat such as in habitat surrounding Ten Mile Bore, through optimal fire and grazing regimes • Additional water sources; and • Monitoring actions to ensure the maintenance or enhancement of the SBTF population and associated habitat attributes.

Appendix B Assay optimisation and eDNA analysis for black-throated finch

Assay optimisation and eDNA analysis for black-throated finch *(Poephila cincta)*.

Wednesday, 29 May 2024	
Project number:	ED2415CR1
Client:	E2M Consulting
Prepared for:	Ed Meyer, John van Osta
Lab analysis by:	Haylo Roberts
Report prepared by:	Josh Griffiths
Assay(s):	Black-throated finch (*Poephila cincta*)
Filter used:	1.2 μm PES syringe filter

Highlights

- A species-specific qPCR assay was developed for *Poephila cincta* using local tissue samples and online sequences.
- One or two replicate water samples were collected from 18 natural and artificial watering points in arid Queensland during February 2024.
- All samples (25) were analysed for black-throated finch (*Poephila cincta*) DNA using qPCR.
- Black-throated finch DNA was detected in water samples from 2 sites.
- Contamination was not detected in laboratory controls.

Background

Environmental DNA (eDNA) methods are being used routinely to monitor aquatic animals including fish, amphibians and mammals across waterways, estuaries and wetlands throughout Australian catchments. Analysis of environmental samples for eDNA can be a highly sensitive and cost-effective method for detecting species presence, particularly for rare or elusive species (Biggs et al. 2015; Smart et al. 2015; Thomsen et al. 2012; Valentini et al. 2016). Here, we develop a species-specific eDNA assay for black-throated finch (*Poephila cincta*), and then use to screen 25 water samples collected from natural and artificial isolated watering points (one or two replicate samples at each site) throughout arid central Queensland.

Methods

Sampling

During February 2024, 25 water samples were collected from 18 sites by E2M Consulting staff following EnviroDNA protocols. At each site, 1 or 2 replicate samples were collected by passing up to 450 mL of water (mean = 208 mL) through a 1.2 μm PES syringe filter. Filtration was undertaken on-site to reduce DNA degradation during transport of water samples (Yamanaka *et al.* 2016). Clean sampling protocols were employed to minimise contamination risk between sites. A preservative was added to each filter following sampling to minimise DNA degradation during transportation. Filters were stored out of sunlight and at ambient temperature before being transported to the laboratory for processing.

To assist with assay development, DNA samples (bloods) from local black-throated finch (*n* = 3) and local off-target finch species (Zebra finch *Taeniopygia guttata n* = 1, double barred finch *Taeniopygia bichenovii n* = 1, plum-headed finch *Neochmia modesta n* = 1) were collected and stored on FTA cards. DNA was extracted from the reference samplesusing an altered Qiagen DNeasy Blood & Tissue kit protocol, with the alteration of an extended (overnight) incubation in lysis buffer and sequenced using primers from Rollins et al. (2012). Sequences were aligned and visualised using Geneious software and NCBI Blast and primer design tool used to assess for off-targets with sympatric species. Limit of detection (LOD) and limit of quantification (LOQ) were calculated using synthetic DNA for the *P. cincta* probe region (Thalinger *et al.* 2021).

Analysis

DNA was extracted from filters using a commercially available DNA extraction kit (Qiagen Power Soil Pro Kit) that minimises compounds in environmental samples that can inhibit PCR reactions. Real-time quantitative Polymerase Chain Reaction (qPCR) assays were used to amplify the target DNA, using a species-specific probe targeting a small region of the mitochondrial DNA for the target species. Assays were performed in triplicate on each sample. Positive and negative controls were included for all assays.

Results

Assay development

A qPCR assay targeting the control region (CR) for black-throated finch was developed using available sequences on Genbank (www.ncbi.nlm.nih.gov) as well as supplied blood samples. Two probes were originally developed that performed similarly for the target species. However, Probe CR1 amplified all local off-targets at similar PCR cycles (Ct values) to the target species and was therefore discarded. Probe 2 did not amplify *Neochmia modesta*, but did amplify *Taeniopygia*

guttata and *Taeniopygia bichenovii* at approximately 10 cycles higher Ct value than the target species (i.e. requires 2^{10} higher DNA concentrations to be effective). It is presumed that off-target amplification would not occur with the low concentration that eDNA samples typically yield. The LOD for this assay is determined to be 2 copies/microlitre and the LOQ is determined to be 10 copies/microlitre.

Data spreadsheet

Results of the eDNA analysis for all samples are summarised in the attached spreadsheet (ED2415CR2_E2M_BTF_data). Black-throated finch DNA was positively detected at one site (TT02; 3 out of 3 assays positive for the sample) while one other site returned an equivocal result (CWAT99; 1 out of 3 assays positive for the sample). All other sites returned negative results (0 positive qPCRs in samples).

Two or three positive qPCRs (out of three technical replicate assays undertaken for each sample) strongly suggests that a sample is positive for the presence of the target species. Samples with only one positive qPCR indicate lower levels of target DNA and are classified as “possible detections”. This can signal that the target species is present in low abundance or is transient. Alternatively, low levels of target DNA may arise from sample contamination through field sampling or laboratory screening processes (minimised through strict protocols and negative controls), facilitated movement of DNA between waterbodies (e.g., water birds, recreational anglers, water transfers, predator scats), or eDNA dispersal from other sites (e.g., flow, floods). Possible detections in multiple replicate samples from a site provide greater confidence regarding presence of the target species. If greater confidence is required, further sampling is recommended at sites where there are possible detections to confirm or deny the presence of the target species.

Quality assurance

- Included in each assay plate were control reactions containing a range of *Poephila cincta* DNA concentrations.
- The R^2 value from a regression of qPCR cycle number vs DNA concentration was 0.9779, indicating the assay performed as expected.
- No contamination was detected in negative controls (n = 3).

References

Biggs J., Ewald N., Valentini A. *et al.* (2015) Using eDNA to develop a national citizen science-based monitoring programme for the great crested newt (*Triturus cristatus*). *Biol. Conserv.* doi: 10.1016/j.biocon.2014.11.029. [online].

Lugg W. H., Griffiths J., van Rooyen A. R., Weeks A. R. & Tingley R. (2018) Optimal survey designs for environmental DNA sampling. *Methods Ecol. Evol.* **9** , 1049–1059.

Smart A. S., Tingley R., Weeks A. R., Van Rooyen A. R. & McCarthy M. A. (2015) Environmental DNA sampling is more sensitive than a traditional survey technique for detecting an aquatic invader. *Ecol. Appl.* **25** , 1944–1952.

Thalinger B., Deiner K., Harper L. R. *et al.* (2021) A validation scale to determine the readiness of environmental DNA assays for routine species monitoring. *Environ. DNA* **3** , 823–836.

Thomsen P. F., Kielgast J., Iversen L. L. *et al.* (2012) Monitoring endangered freshwater biodiversity using environmental DNA. *Mol. Ecol.* **21** , 2565–2573.

Valentini A., Taberlet P., Miaud C. *et al.* (2016) Next-generation monitoring of aquatic biodiversity using environmental DNA metabarcoding. *Mol. Ecol.* **25** , 929–942.

Yamanaka H., Motozawa H., Tsuji S., Miyazawa R. C., Takahara T. & Minamoto T. (2016) On-site filtration of water samples for environmental DNA analysis to avoid DNA degradation during transportation. *Ecol. Res.* **31** , 963–967.

Disclaimer

The professional analysis and advice in this report has been prepared for the exclusive use of the party or parties to whom it is addressed (the addressee) and for the purposes specified in it. This report is supplied in good faith and reflects the knowledge, expertise and experience of the consultants involved. The report must not be published, quoted or disseminated to any other party without prior written consent from EnviroDNA pty ltd.

EnviroDNA pty ltd accepts no responsibility whatsoever for any loss occasioned by any person acting or refraining from action as a result of reliance on the report. In conducting the analysis in this report EnviroDNA pty ltd has endeavoured to use what it considers is the best information available at the date of publication including information supplied by the addressee. Unless stated otherwise EnviroDNA pty ltd does not warrant the accuracy of any forecast or prediction in this report.