School of Molecular and Life Sciences
Faculty of Science and Engineering

Improving translocation management for restricted range reptiles
Case Study: The Western Spiny-tailed Skink (*Egernia stokesii badia*)

Holly Sydelle Bradley
ORCID: 0000-0002-7937-857X

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Declaration:

To the best of my knowledge and belief this thesis contains no material previously published by any other person except where due acknowledgement has been made.

This thesis contains no material which has been accepted for the award of any other degree or diploma in any university. Several chapters of this thesis have been accepted for publication or are under peer review at the time that the thesis has been submitted for examination. These are detailed in the Statement of Contributions.

The research presented and reported in this thesis was conducted in compliance with the National Health and Medical Research Council Australian code for the care and use of animals for scientific purposes 8th edition (2013). The proposed research study received animal ethics approval from the Curtin University Animal Ethics Committee, Approval number ARE2018-28 (Chapters 3-5).

This project received funding from Karara Mining Ltd. and the Gunduwa Regional Conservation Association WA. HSB was additionally supported by the Australian Research Council Industrial Transformation Training Centre for Mine Site Restoration (ICI150100041) and the 2019 Postgrad Australia scholarship.

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Statement of contributions

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The study presented in **Chapter 2** was published in the peer-reviewed journal, *Conservation Biology* on November 19th, 2020:

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All authors conceived the ideas and designed the methodology; HSB collected the data; HSB, MDC and ST analysed the data; HSB wrote the manuscript; all authors contributed to the revisions of the manuscript.

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HSB and PWB designed the experiment. HSB carried out the fieldwork and undertook the digitisation of LiDAR 3D models. HSB, ATC, MDC, and ST conducted the statistical analysis and interpretation of the data. HSB wrote the main manuscript text, with all authors contributing to subsequent drafts.

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Name: Assoc/Prof Philip W. Bateman

Signature: **Date:** 28 November 2021

Name: Dr Michael D. Craig

Signature: **Date:** 29 November 2021

Name: Dr Sean Tomlinson

Signature **Date:** 28 November 2021

Name: Dr Adam Cross

Signature: **Date:** 26 November 2021

Name: Dr Mike Bamford

Signature: **Date:** 30 November 2021

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General abstract

Translocation has become one of the main conservation tools used to help safeguard threatened species around the globe. The increase in the number of translocations has not, however, necessarily corresponded with advancement in translocation procedure, nor success. This is particularly the case for translocations used to offset the impacts of anthropogenic disturbance: “mitigation translocations”. It is high priority for future mitigation translocations to follow the same strategic framework as other conservation translocations, with significant, long-term investment into understanding the ecological requirements and conservation status of the target species. Maximising the efficacy of translocations is critical given limited global conservation funding, especially if the objective of biodiversity conservation is undermined by failing to save individuals. Improving translocation procedures is particularly relevant to reptile translocations in Australia, which is a global hotspot for reptile diversity, yet suffers a significant knowledge gap surrounding the conservation status and ecological requirements of its diverse reptilian fauna.

This thesis uses the Western Spiny-tailed Skink (*Egernia stokesii badia*) as a case study to highlight how identifying and addressing knowledge gaps on the ecological requirements of a threatened reptile in Australia is key to informed, effective management. In this thesis a range of methodologies and technologies was used to understand the ecological requirements of *E. s. badia*, including plasticine model experiments, camera trapping, unbounded point count surveys, terrestrial LiDAR modelling and DNA metabarcoding. Using plasticine models, point count surveys, and camera traps to understand predation risk, corvids, native predators of *E. s. badia*, were found to preferentially forage around log piles containing skinks, leading to a potential issue for skinks if corvid populations are artificially augmented around anthropogenic food sources such as landfill sites on mining tenements. Cats were also a confirmed predator of both adult and juvenile skinks and, therefore, a significant threat to an entire colony. The novel application of terrestrial LiDAR was shown to be an effective tool in quantifying the structural microhabitat requirements of skink colonies. Skinks were found to be selective in occupying longer log piles with an average of two logs, and with overhanging vegetation, preferably at mid-storey height. Quantifying this has implications for the design of potential translocation sites for the species. Moreover, using LiDAR in this way has extensive applications for the assessment of complex microhabitat types essential to the persistence of other threatened and cryptic species, such as rock piles and cave structures. Lastly, the complementary use of visual identification and DNA metabarcoding was also successful at identifying different invertebrate and plant families comprising the diet of *E. s. badia.* Springtime feeding on moth larvae and the plant families Asteraceae and Crassulaceae appeared particularly important components of skink diet, likely supporting the rapid growth and development of juveniles. The close proximity of these plant and invertebrate resources to translocation sites is, therefore, likely to be an important contributor to translocation success. The detailed ecological information provided within this thesis substantially increases the ecological knowledge base for this endangered subspecies of skink. It helps inform best practice for mitigation translocations and overall species conservation, plus provides a clear strategic framework for the targeted research of meaningful ecological factors that influence on-ground translocation decision making and is adaptable for the improved management of a range of other fauna, including other threatened restricted range endemic reptiles.

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List of abbreviations

ACT	Australian Capital Territory
ANOSIM	Analysis of Similarity
ARC	Australian Research Council
CMSR	Centre for Mine Site Restoration
COI	Cytochrome *c* Oxidase subunit I
CSIRO	Commonwealth Scientific and Industrial Research Organisation
CWD	Coarse Woody Debris
DTM	Digital Terrain Model
EIA	Environmental Impact Assessments
GLMER	Generalised Linear Mixed-Effects (Regression) model
GRCA	Gunduwa Regional Conservation Association
IBRA	Interim Biogeographic Regionalisation for Australia
IUCN	International Union for Conservation of Nature
KML	Karara Mining Ltd
LiDAR	Light Detection and Ranging
LRM	Logistic Regression Model
MID	Multiplex Identifier
OHS	Occupational Health and Safety
OUT	Operational Taxonomic Unit
PCR	Polymerase Chain Reaction
PloS	Public Library of Science
SVL	Snout-Vent Length
ZOTU	Zero-radius Operational Taxonomic Units

Chapter 1. General introduction

1.1 Introduction

Translocation biology is a growing biological field (Seddon et al. 2007) and is one of the main conservation tools used to help safeguard threatened species (Fischer and Lindenmayer 2000, Rout et al. 2007). However, a large increase in the number of translocations has not corresponded with an advancement in translocation procedure (Seddon et al. 2007, Teixeira et al. 2007, Taylor et al. 2017). This has been attributed to the largely *ad hoc* nature of translocation methods (Seddon et al. 2007), which could be improved through integrating translocation biology projects within the more holistic scientific frameworks of restoration ecology, which focusses on restoring functional ecosystems (Lipsey et al. 2007), and conservation biology, which focusses on retaining priority species in a conservation context (Caughley 1994). A more strategic approach to translocation biology can be established through selecting *a priori* goals to form testable hypotheses to help understand the target species in its broader ecological context (Taylor et al. 2017). A framework of key questions to help develop these goals has been developed by Armstrong and Seddon (2008), which includes an understanding of population (establishment and persistence) level, metapopulation level and ecosystem-level characteristics of the translocation. Although there has been an increase over time in the proportion of research addressing *a priori* hypotheses, there are still few studies that incorporate the direct comparison of different management actions used pre- or post-translocation, and an integration of this research into translocation management of the species (Taylor et al. 2017). General improvement of species recovery and ecosystem restoration requires research to avoid *post hoc* interpretation of data and retrospective analyses, and rather focus on gaining reliable information on pre-selected key questions in translocation biology, comparing different management techniques for continued future improvement in translocation practices (Seddon et al. 2007, Armstrong and Seddon 2008, Taylor et al. 2017).

Herpetofauna, in particular, have been largely overlooked within larger translocation reviews (Germano and Bishop 2009), with a research bias towards mammals and birds in the literature (Seddon et al. 2005). Debate within the scientific community has also disagreed upon the suitability of reptiles for translocation (Burke 1991, Dodd Jr and

Seigel 1991, Reinert 1991, Germano and Bishop 2009). However, there has been a two-fold increase in the number of successful amphibian and reptile translocations published in the last thirty years (Germano and Bishop 2009), indicating reintroduction biology to be a viable conservation tool for reptile species, as has long been suggested for other taxonomic groups (Griffith et al. 1989). Despite this positive trend, there are still significant knowledge gaps which can impede translocation success, leading to a high frequency of reptile translocations ending in failure (McCoy et al. 2014, Tingley et al. 2016). Of all reptile species assessed using IUCN Red List Criteria, 19% are Data Deficient (Tingley et al. 2016). There is a particular lack of data surrounding movement and habitat requirements, and these are two of the greatest contributing factors towards reptile translocation failure (Germano and Bishop 2009). Therefore, there is a major knowledge gap associated with reptile species globally, critical to be addressed in order to improve reintroduction techniques.

1.2 Focal species and site selection

While detailed information has been collected on *Egernia stokesii*, with a distribution spanning across Australia (Figure 1.1; Pearson 2012), little research has been undertaken on the subspecies endemic to Western Australia, *Egernia stokesii badia* (Figure 1.2). This endangered subspecies is at risk of extinction from a number of threatening processes (Pearson 2012). A recommended recovery action listed under the Western Spiny-tailed Skink Recovery Plan is the development of a strategy to translocate at-risk populations (Pearson 2012). However, mitigation-driven translocations (moving individuals away from an immediate, deadly threat) run the risk of failure without applying a more structural approach to address key translocation questions (Armstrong and Seddon 2008, Germano et al. 2015). Although there have been no successful translocations of this subspecies published, there are anecdotal reports of unsuccessful attempts, indicating complex requirements for establishment and persistence. *E. s. badia* is also at particular risk from inappropriate mitigation translocations as much of its habitat lies on mining tenements. There is, therefore, strong reason to investigate the biotic and behavioural requirements of the skinks to improve translocation management protocols for this subspecies. For this reason, *E. s. badia* was selected as a case study organism for this thesis, to exemplify how investigating key ecological questions prior to translocation can help address the knowledge gaps typical of reptile translocations, and highlight best practice techniques for mitigation translocations, while providing useful ecological knowledge for on-the-ground management of an endangered species.

Egernia stokesii badia is endemic to the central region of Western Australia, in the semi-arid Geraldton Sandplains and Yalgoo IBRA regions (Pearson, 2012). Individuals are typically reddish-brown in colour, with obvious dorsal white or cream irregular stripes or blotches and spinose scales, particularly on their tails (Pearson 2012). Along with several other members of the *Egernia* genus, *E. s. badia* is among the most social of squamate reptiles, living in stable social aggregations of closely related individuals (Chapple 2003). The colonies live in log piles, comprised of fallen tree trunks and branches, in *Acacia*-dominated shrubland and open eucalypt woodland (Pearson 2012).

For the investigation of the specific ecological requirements of *E. s. badia* for improved management, the research area selected was on a mining tenement (Figure 1.1), located approximately 450km north-east of Perth in the Mid West region of Western Australia (29°10'54.0"S, 116°32'55.1"E). The climate of this area is semi-arid, with a yearly mean rainfall of approximately 300mm (Australian Bureau of Meteorology 2020; Figure 1.4). The tenement also contains *E. s. badia* habitat consisting of open eucalypt woodland (Fig 1.3) with fallen log piles for shelter (Fig 1.4).

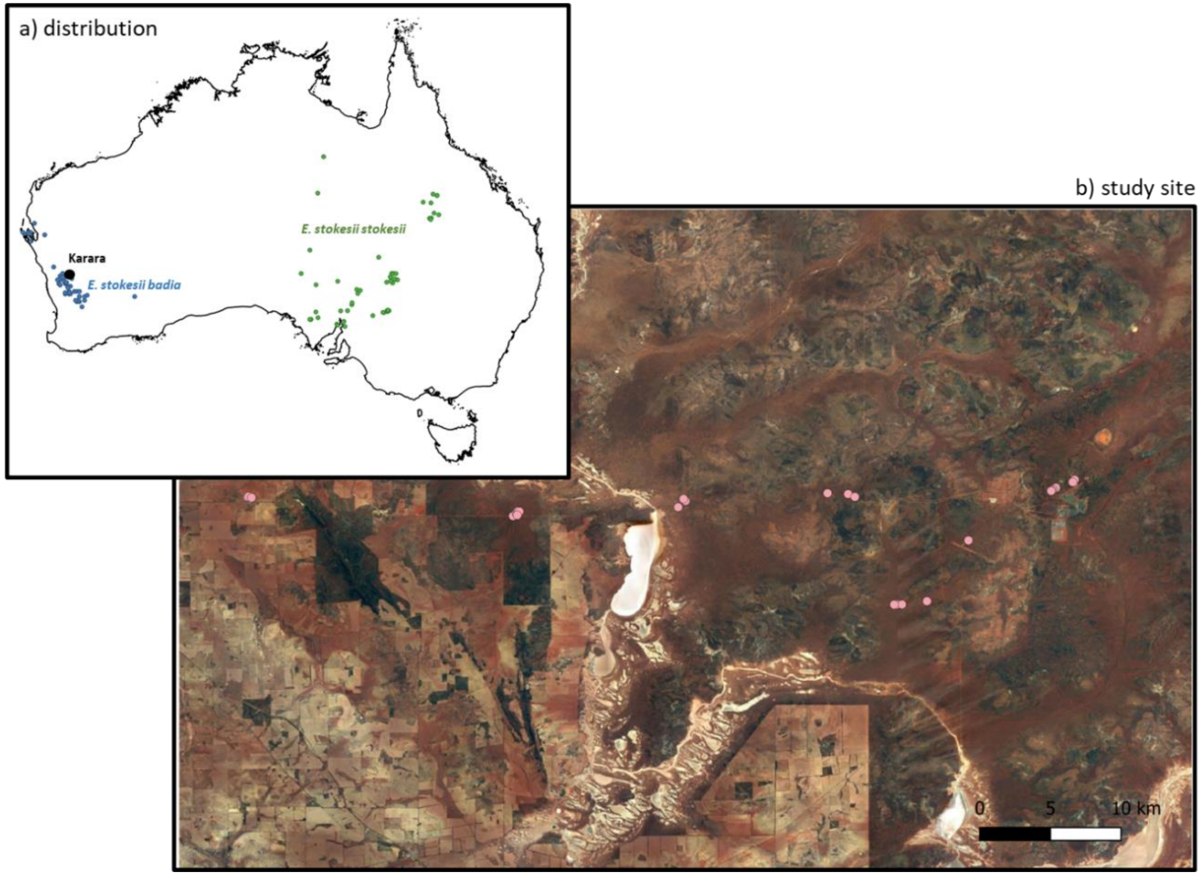


Figure 1.1 (A) Distribution of two subspecies of *Egernia stokesii* according to records maintained by the Atlas of Living Australia (https://www.ala.org.au/, accessed 19 Sept 2021), and the location of the Karara study site. **(B)** Locations of log pile sites (pink) including those occupied with *Egernia stokesii badia* colonies, sampled within the study, throughout the mining tenement in the Mid West region of Western Australia.

Figure 1.2 Examples of adult *Egernia stokesii badia* **(A & B)**, and subadults **(C & D)**, highlighting the variation in colour and patterning. Photos taken by Holly Bradley.

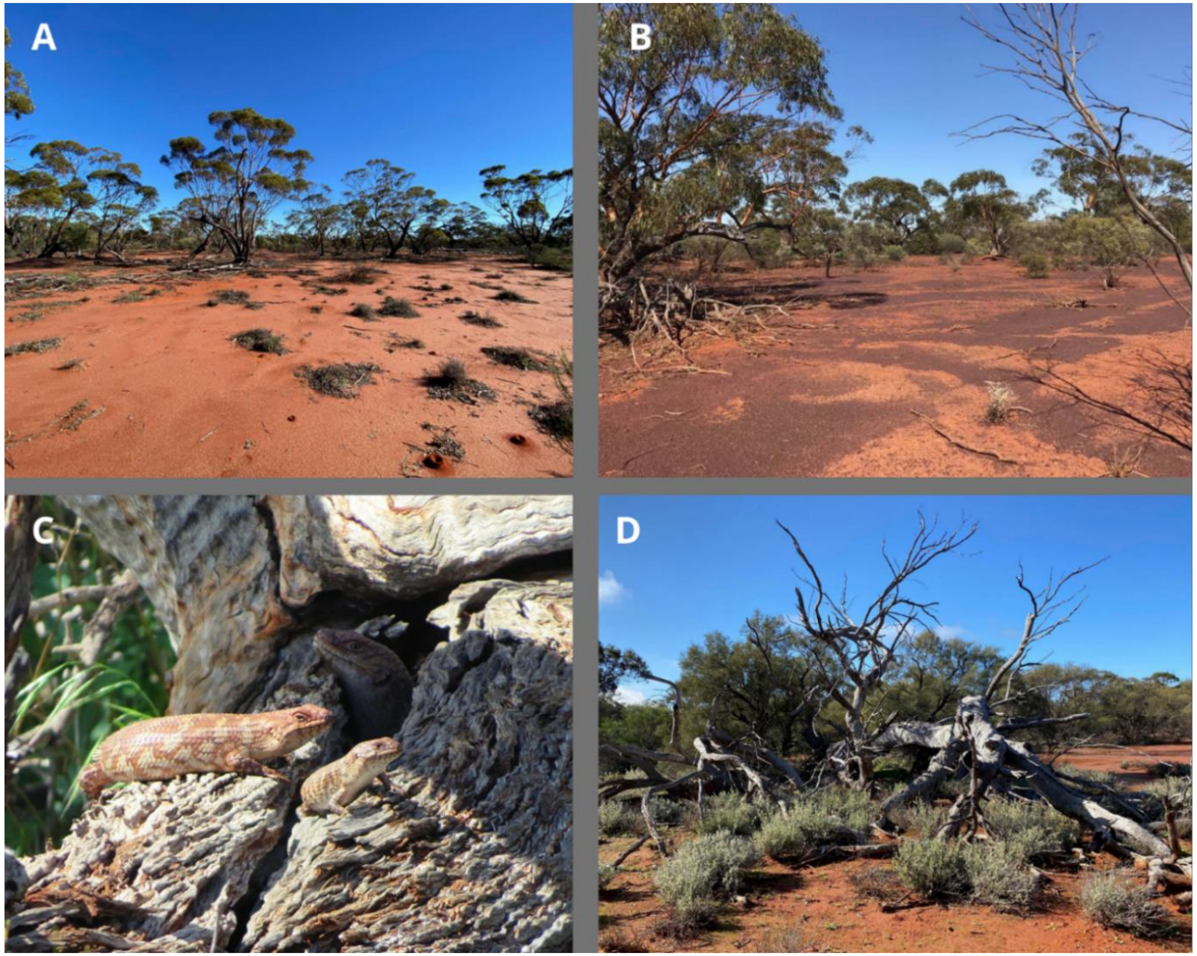


Figure 1.3 Typical habitat structure of open eucalypt woodland **(A & B)**, example of various sized skinks sharing the same log pile microhabitat (**C**), and an example of a log pile structure occupied by a colony of *Egernia stokesii badia* (**D**). Photos taken by Holly Bradley.

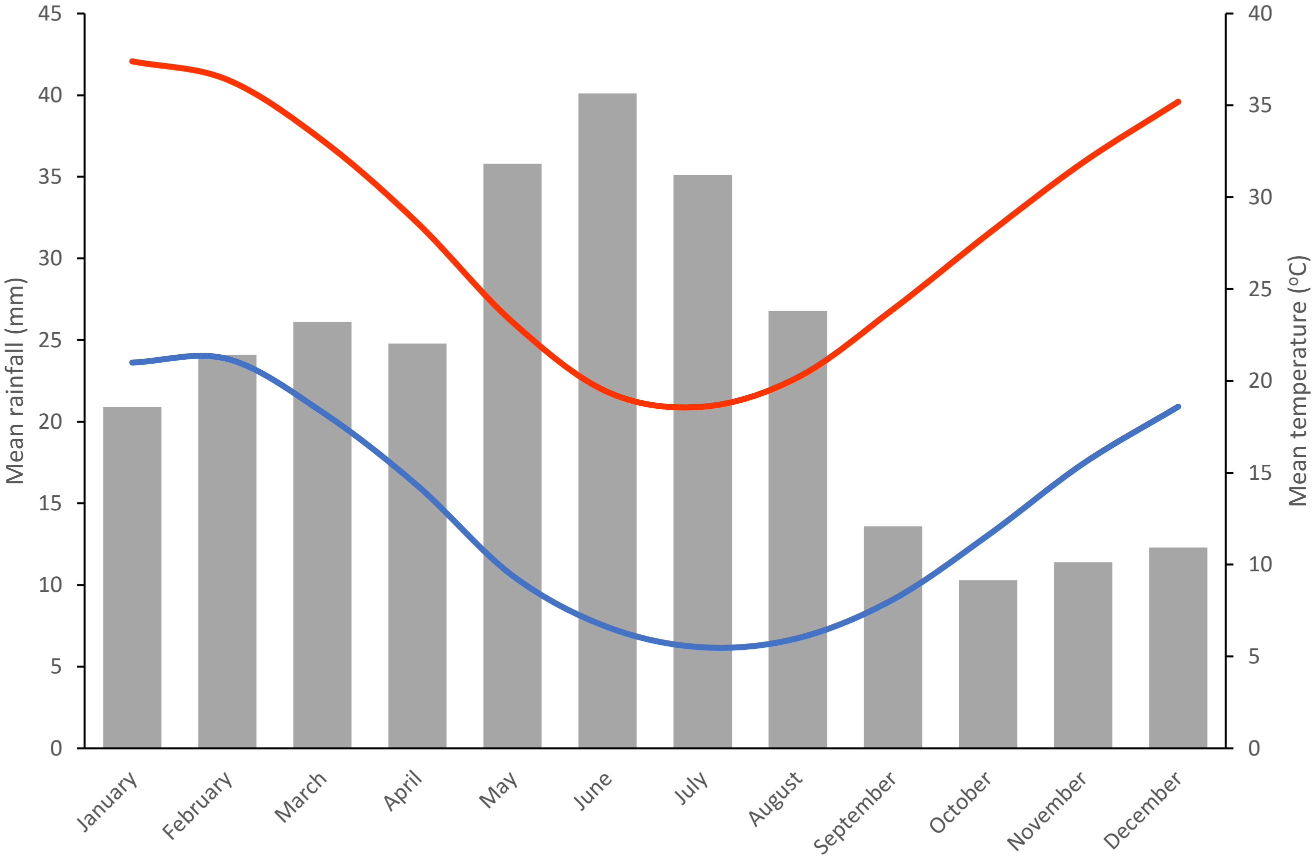


Figure 1.4 Mean rainfall (grey), mean maximum temperature (red), and mean minimum temperature (blue), representative of the semi-arid Mid West region, taken from the Paynes Find weather station, approximately 80 kilometres from the study site (Australian Bureau of Meteorology 2021). Means taken from the years 1975-2021.

1.3 Thesis overview

This thesis will follow a strategic approach towards the conservation of a threatened Western Australian endemic reptile subspecies, to highlight how research into key ecological questions prior to translocation can address knowledge gaps typical of reptile translocations (Figure 1.5). The thesis chapters are structured and formatted as journal articles, with one published (Chapter 2), two currently under review (Chapters 3 and 4) and one that is in preparation (Chapter 5). The focal species is *Egernia stokesii badia*, a subspecies of *E. stokesii* restricted to Western Australia, which will most probably require mitigation translocations in the future to prevent mortality of colonies, due to habitat clearing for ore extraction (Pearson 2012).

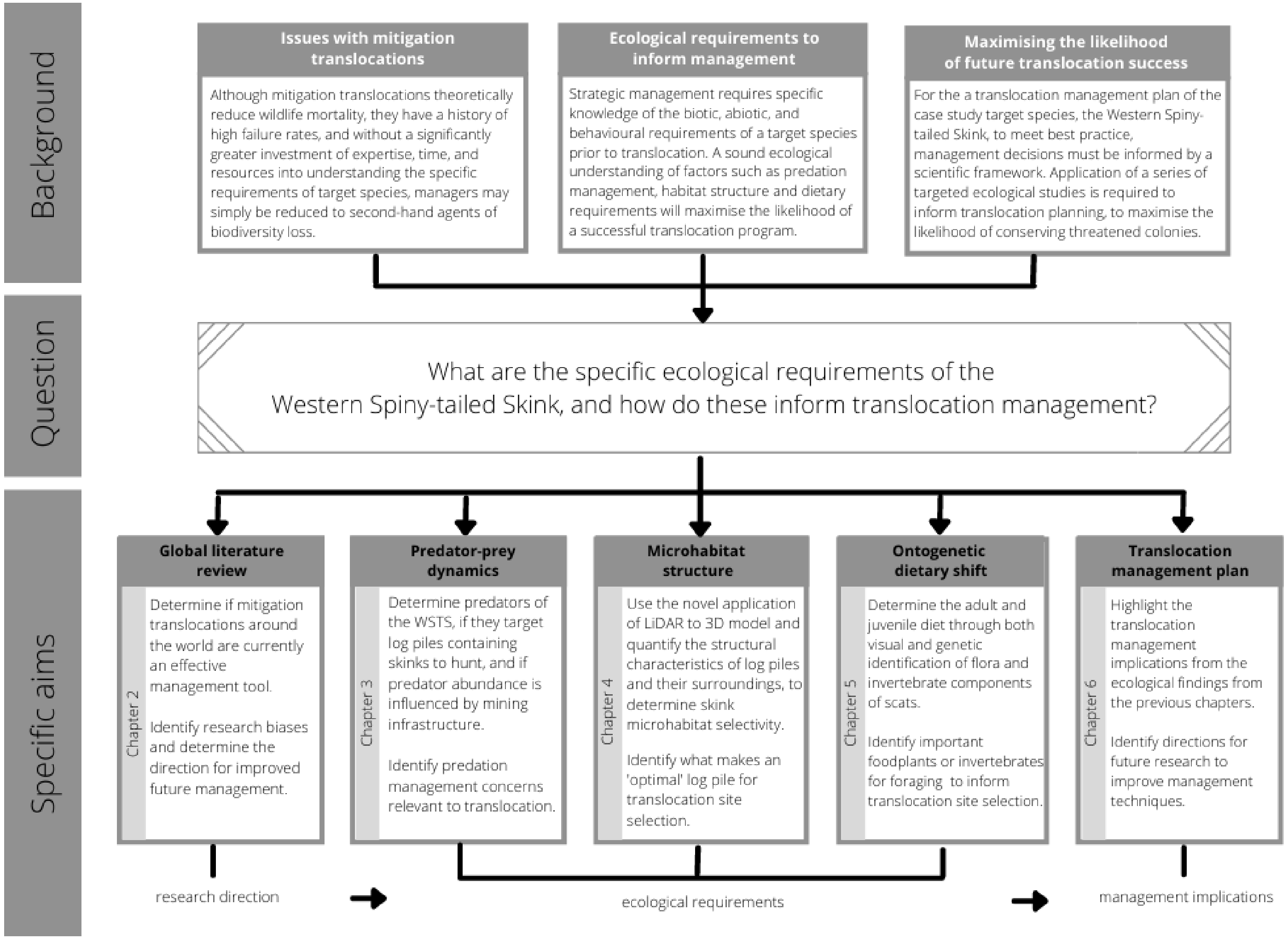
Chapter 2 reports a global review of the mitigation translocation literature, to investigate if mitigation translocations are currently being implemented as an effective management tool and to decipher the path to achieve best practice in mitigation translocation science. The findings of this review help to inform the strategic, scientific framework used to examine the specific ecological requirements of the study species explored in the remainder of the thesis, to maximise the likelihood of successful future translocations.

Chapter 3 investigates the predator-prey dynamics of the semi-arid ecosystem where *E. s. badia* is located. Predator relative activity, hunting behaviour, prey selectivity, and response to mining infrastructure are investigated over multiple seasons through multiple experimental methods, including plasticine model experiments, predatory bird surveys, and camera trapping. Understanding the dynamics between habitat structure and predator-prey interactions is critical for informed management decisions, such as the targeted control of invasive or generalist species and providing critical predator refuge structures in the implementation of mitigation translocation programs in the future.

Chapter 4 quantifies the structural characteristics of log piles and their surrounding habitat using terrestrial LiDAR (light detection and ranging) technology, to determine the features of log piles that are critical in their selection by *E. s. badia* colonies. Determining if the target subspecies has high selectivity of log pile habitat is important in informing managers of what structural features are required for optimal translocation site selection or modification.

Chapter 5 investigates the ontogenetic diet differences of *E. s. badia,* using both visual identification and genetic analysis of vegetation and invertebrates found in scats. Understanding the diet of *E. s. badia* is important in identifying any foodplants or invertebrate groups that comprise a significant component of their diet, and are, therefore, important for consideration when selecting or developing appropriate translocation sites in the future.

Chapter 6 is the thesis discussion, which synthesises all the gathered ecological data from the previous chapters to provide an informed translocation strategy for *E. s. badia* colonies in the future. Areas requiring further investigation to maximise the likelihood of future translocation management are also highlighted.



Issues with mitigation translocations

Although mitigation translocations theoretically reduce wildlife mortality, they have a history of high failure rates, and without a significantly greater investment of expertise, time, and resources into understanding the specific requirements of target species, managers may simply be reduced to second-hand agents of biodiversity loss.

Ecological requirements to inform management

Strategic management requires specific knowledge of the biotic, abiotic, and behavioural requirements of a target species prior to translocation. A sound ecological understanding of factors such as predation management, habitat structure and dietary requirements will maximise the likelihood of a successful translocation program.

Maximising the likelihood of future translocation success

For the a translocation management plan of the case study target species, the Western Spiny-tailed Skink, to meet best practice, management decisions must be informed by a scientific framework. Application of a series of targeted ecological studies is required to inform translocation planning, to maximise the likelihood of conserving threatened colonies.

What are the specific ecological requirements of the Western Spiny-tailed Skink, and how do these inform translocation management?

Global literature review

Determine if mitigation translocations around the world are currently an effective management tool.

Identify research biases and determine the direction for improved future management.

Predator-prey dynamics

Determine predators of the WSTS, if they target log piles containing skinks to hunt, and if predator abundance is influenced by mining infrastructure.

Identify predation management concerns relevant to translocation.

Microhabitat structure

Use the novel application of LiDAR to 3D model and quantify the structural characteristics of log piles and their surroundings, to determine skink microhabitat selectivity.

Identify what makes an 'optimal' log pile for translocation site selection.

Ontogenetic dietary shift

Determine the adult and juvenile diet through both visual and genetic identification of flora and invertebrate components of scats.

Identify important foodplants or invertebrates for foraging to inform translocation site selection.

Translocation management plan

Highlight the translocation management implications from the ecological findings from the previous chapters.

Identify directions for future research to improve management techniques.

Figure 1.5 Thesis overview, highlighting the overall research question, specific aims, and background justification.

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Chapter 2. Mitigation translocation as a management tool

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Western Spiny-tailed Skink adult and juvenile, photo by Holly Bradley.

2.1 Abstract

Mitigation translocation is a subgroup of conservation translocation, categorized by a crisis-responsive time frame and the immediate goal of relocating individuals threatened with death. However, the relative successes of conservation translocations with longer time frames and broader metapopulation- and ecosystem-level considerations have been used to justify the continued implementation of mitigation translocations without adequate post hoc monitoring to confirm their effectiveness as a conservation tool. Mitigation translocations now outnumber other conservation translocations and understanding the effectiveness of mitigation translocations is critical given limited global conservation funding especially if the mitigation translocations undermine biodiversity conservation by failing to save individuals. We assessed the effectiveness of mitigation translocations by conducting a quantitative review of the global literature. A total of 59 mitigation translocations were reviewed for their adherence to the adaptive scientific approach expected of other conservation translocations and for the testing of management options to continue improving techniques for the future. We found that mitigation translocations have not achieved their potential as an effective applied science. Most translocations focused predominantly on population establishment- and persistence-level questions, as is often seen in translocations more broadly, and less on metapopulation and ecosystem outcomes. Questions regarding the long-term impacts to the recipient ecosystem (12% of articles) and the carrying capacity of translocation sites (24% of articles) were addressed least often, despite these factors being more likely to influence ultimate success. Less than half (47%) of studies included comparison of different management techniques to facilitate practitioners selecting the most effective management actions for the future. To align mitigation translocations with the relative success of other conservation translocations, it is critical that future mitigation translocations conform to an established experimental approach to improve their effectiveness. Effective mitigation translocations will require significantly greater investment of time, expertise, and resources in the future.

2.2 Keywords

biodiversity conservation, human-wildlife interaction, mitigation hierarchy, phased destruction, translocation biology

2.3 Article impact statement

In the absence of high standards of planning and monitoring, mitigation-translocation managers may be secondhand agents of biodiversity loss.

2.4 Introduction

One of the hallmarks of the Anthropocene is that wildlife extinctions are occurring at a rate thousands of times greater than background species losses (Ceballos et al. 2010). This rate is predicted to increase (Johnson et al. 2017), suggestive of a mass extinction event. A major contribution to this rate of extinction is the loss of populations due to habitat loss and land-use change (Foley et al. 2005). Intensive conservation actions, such as translocations (Beeton et al. 2010), have been recommended to mitigate the magnitude of these losses (Thomas 2011; Boyer et al. 2016). According to the IUCN (2013), conservation translocations are a demand-driven practice concerning "the deliberate movement of organisms from one site for release in another [that] must yield a measurable conservation benefit at the levels of a population, species or ecosystem, and not only provide benefit to translocated individuals." Mitigation translocation is a supply-driven subset of conservation translocation and is "implemented in response to legislation or governmental regulation, with the intent of reducing a development project's effects on animals or plants inhabiting the site" (Germano et al. 2015). Therefore, compared with the goal of augmenting or enhancing the viability of recipient populations for long-term conservation benefit, the trigger for mitigation translocations is to prevent the mortality of the at-risk founder individuals (e.g., nuisance animals) (Massei et al. 2010), populations (e.g., at a development site) (Germano et al. 2015; Nally and Adams 2015), or a threatened taxon with the known global population threatened by human activity. Despite only slight differences in triggers and time scales between mitigation and other conservation translocations, the International Union for Conservation of Nature (IUCN 2013) states that: "Rigorous analysis and great caution should be applied when assessing potential future conservation benefits [of mitigation translocations] and using them to mitigate or offset current development impacts." What remains to be understood is whether mitigation translocations currently adhere to such rigorous analysis and design prior to implementation, and if they do not, whether it is possible for mitigation translocations to absorb the extra requirements of time and finances to do so. It is, therefore, disadvantageous to global conservation to ignore these 2 subgroups of translocations, and the relative success of other conservation translocations should no longer be used to justify the continued use of mitigation translocations (Germano et al. 2015).

Translocation is often expensive (Caldecott and Kavanagh 1983; Maunder 1992; Carter and Newbery 2004; Seddon et al. 2005), and the success of translocations must be maximised to best use the limited global funding available for biodiversity conservation (Waldron et al. 2013). To facilitate better translocation outcomes, Armstrong and Seddon (2008) developed a list of key questions to address during a reintroduction event as one strategy to improve translocation success around the globe (Table 2.1). Their questions focus on how a project addresses the population, metapopulation, and ecosystem implications of a translocation (Armstrong and Seddon 2008). At the population level, they consider a site's capacity to support a species and the viability of the founder group to maintain a self-sustaining population (Armstrong and Seddon 2008). This includes considerations of founder behavioural plasticity (Page et al. 2019), pre-release predator exposure (Frair et al. 2007), and habitat quality (Johnson and Swift 2000). At the metapopulation level, translocations require the optimal allocation of individuals (Wolf et al. 1998), or population reinforcement at translocation sites (Armstrong and Seddon 2008). At the ecosystem level, translocations also have the potential to introduce parasites (Schaffer et al. 1981; Fernández-de-Mera et al. 2003; Thompson et al. 2010), disease (Caldecott and Kavanagh 1983; Woodford and Rossiter 1994; Kock et al. 2010), and non-native species (Ruesink et al. 1995; Manchester and Bullock 2000; Olden et al. 2006) into the recipient ecosystem (Armstrong and Seddon 2008). Approximately 1% of translocations result in subsequent environmental harm (Williamson and Fitter 1996). Factors at each of these 3 levels, therefore, interact to influence the likelihood of long-term translocation success and persistence. Notably, these guiding principles imply that success is characterised not simply by the self-sustained persistence of the translocated individuals or population, but that their persistence is also not damaging to the recipient ecosystem or to the metapopulation structure of the focal species as a whole (Armstrong and Seddon 2008).

Table 2.1 Variables and questions searched for in the mitigation translocation article data set and explanations on how criteria were met

Variable or question	**Question or definition in Armstrong and Seddon (2008)**	**Article included in review (?) considered to address variable or question if:**
Question 1	How is establishment probability affected by size and composition of the release group?	Article mentions either population size or composition and how this influenced the survival or establishment of translocated individuals (through post hoc analysis or experimentation).
Question 2a	How are post-release survival and dispersal affected by pre-release management?	Article mentions a pre-release management technique (e.g., different types of soft-release structures) and experimental testing or comparison with another technique to determine the benefit to survival or dispersal.
Question 2b	How are post-release survival and dispersal affected by post-release management?	Article mentions a pos-release management technique (e.g., supplementary feeding and experimental testing or comparison with another method to determine the benefit to survival or dispersal.
Question 3	What habitat conditions are needed for persistence of the reintroduced population?	Article mentions habitat conditions considered when selecting the translocation site, including temperature or climate, vegetation, predator abundance or management, soil, geology, and slope.
Question 4	How will genetic makeup affect persistence of the reintroduced population?	Article mentions genetic testing, modelling, or monitoring, for example relating to inbreeding, ancestry, genetic diversity, and the need for future population supplementation.
Question 5	How heavily should source populations be harvested?	Article states reason for the number of founders selected (e.g., if construction threatened an entire population as many as possible were removed) or modelled how many founders would be sustainable to remove.
Question 6	What is the optimal allocation of translocated individuals among sites?	Article >1 release site and how a decision was made to allocate individuals between them.
Question 7	Should translocation be used to compensate for isolation?	All articles considered to answer this question due to the assumption stated by Armstrong and Seddon (2008) that all translocations implicitly consider this question.
Question 8	Are the target species or taxon and its parasites native to the ecosystem?	Article states the translocation is a reintroduction (meaning it is within the original range of the species) or that habitat is available..

Variable or question	**Question or definition in Armstrong and Seddon (2008)**	**Article included in review (?) considered to address variable or question if:**
Question 9	How will the ecosystem be affected by the target species and its parasites?	Article mentions how the translocation will affect the surrounding ecosystem of the translocation site, such as grazing pressure on vegetation, function as an ecosystem engineer, or ecological replacement for a locally extinct species.
Question 10	How does the order of reintroductions affect the ultimate species composition?	Multiple species translocations occurred and article mentions how the order of translocations was decided, and how this influenced species composition in the system.
A priori goals	Research and monitoring targets are identified before translocation takes place.	Article states goals in the text (e.g., quantify demographic parameters related to survival and reproduction) that were not made after collection of data (e.g., genetic study to look at years of bottlenecks after translocation because these goals did not appear to influence the original translocation design.
Mitigation translocation	supply-driven translocations, where the current population is under threat of extirpation and translocation is required to mitigate the impending threat (IUCN/SSC 2013; Germano et al. 2015).	Article says mitigation translocations occurred both as a response to threatened individuals (e.g., nuisance animals) and populations (e.g., at a development site), as well as in response to an immediate crisis related to the preservation of a threatened taxon, where the known global population is threatened by human activity.
Testing or comparing management actions	One or more management actions are tested either by *a priori* predictive modelling or a posteriori analysis of field data (Taylor et al. 2017). .	Article mentions at least one trial of a management action, other than the translocation itself.

To facilitate an adaptive approach, translocation projects are encouraged to adopt a scientific rationale to address the key questions in translocation biology (Armstrong and Seddon 2008; Moseby et al. 2014; Daniels et al. 2018). The selection of *a priori* goals prioritizes translocation design to answer key questions and assess success through targeted monitoring. Targeted monitoring (such as recording the survival rates of individuals with or without supplementary feeding) is more efficient than unfocused monitoring (such as collecting data for a suite of variables, e.g., survival, habitat features, and predator density with no prior plan) because it avoids collecting purely descriptive data unrelated to management (Nichols and Williams 2006; Taylor et al. 2017). Targeted monitoring is also more likely to identify causes of failure and inform future translocation practice to maximise success (Sutherland et al. 2010; Taylor et al. 2017). *A priori* goals, therefore, promote systematic adaptive management practices and increase the chance of future translocation success (Taylor et al. 2017; Daniels et al. 2018) or recovery of a translocation program in the event of a suboptimal beginning.

The propensity for translocations to consider *a priori* goals (Armstrong and Seddon 2008), as well as other key questions in translocation biology (Table 2.1), has been recently investigated (Taylor et al. 2017). The trend from translocation studies over the last two decades shows a promising increase in testing *a priori* hypotheses, although there remains a focus on assessing the short-term establishment of populations rather than the long-term persistence-level and wider metapopulation- and ecosystem-level dynamics (Taylor et al. 2017). However, Taylor et al. (2017) did not distinguish among the different types of translocations or examine how each responded to the call for more adaptive management.

Mitigation translocations generally receive greater financial support in aggregate than other conservation translocations (Germano et al. 2015), although individual programs are often less well-funded and less coordinated in their planning than other types of conservation-oriented programs (such as ACT Government 2017). Despite caution from the IUCN (2013) regarding mitigation translocations in offsetting development impacts, the number of mitigation translocations undertaken has increased substantially within the last 20 years (Miller et al. 2014; Romijn and Hartley 2016), and now outnumber other conservation translocations (Germano et al. 2015). Mitigation translocations are still regarded by the public as a more humane, species-specific, and effective solution to human-wildlife conflict than traditional culling programs, and this has contributed significantly to their increasing frequency (Massei et al. 2010). Although mitigation translocations theoretically reduce wildlife mortality, they have a history of high failure rates (Sullivan et al. 2015). Proponents often fail to monitor the long-term success of such translocations (Massei et al. 2010), and there is often a lack of publicly accessible results (Nash 2017; Silcock et al. 2019). Furthermore, without the same conservation-oriented goals as other conservation translocations, they often fail to follow scientific best practice (Germano et al. 2015).

Consequently, mitigation translocations are rarely represented in the scientific literature (Armstrong and Seddon 2008; Germano et al. 2015; Taylor et al. 2017), perhaps due to a reluctance to report failures (Germano et al. 2015). There is, therefore, a lack of scientific evidence to assess the effective use of mitigation translocation for their intended purpose of reducing anthropogenic wildlife mortality and promoting biodiversity conservation. With high failure rates, a lack of monitoring, and minimal scientific rationale, the efficacy of mitigation translocations is questionable. Are the majority of cases simply removing wildlife for a socially acceptable death out of the public eye (phased destruction; Jackson et al. 1983) and, therefore, not effective management tools (Germano et al. 2015)?

The value of translocations that follow a strategic experimental framework has been strongly advocated (Armstrong and Seddon 2008; Taylor et al. 2017). However, because mitigation translocations are globally under-represented within the literature (Germano et al. 2015), the increased adoption of a scientific framework is largely informed by other conservation translocations, and it remains unclear whether mitigation translocations follow the same recent recommendations for best practice (Armstrong and Seddon 2008; IUCN SSC 2013). If mitigation translocations fail to follow accepted scientific best practice (Germano et al. 2015), it is unlikely that their success is being maximised, leading to a waste of conservation dollars. We aimed to determine whether mitigation translocations have adopted a strategic, systematic approach to management and to assess their efficacy as a management tool. We aimed to evaluate whether published mitigation translocations considered the population-, metapopulation-, and ecosystem-level repercussions of a translocation event; included *a priori* hypotheses (Schaffer et al. 1981); compared management techniques to inform future management and allow an adaptive approach; and were more likely to result in a self-sustaining translocated population if each of these factors were considered.

2.5 Methods

A quantitative review of the translocation literature was undertaken with Scopus (24 March 2019) to identify all articles that cited Armstrong and Seddon (2008) for the years 2008 through 2019 (Fig. 2.1). This search produced 486 publications, which were reduced to 283 by exclusion of nonempirical data sets, publications that could not be sourced in English (e.g., Apollonio et al. 2001; Azeredo and Simpson 2004;

Yoshio et al. 2009), and publications that were not publicly available (Wacher 1986). The 283 articles were then separated into mitigation-motivated or other conservation-motivated translocations. Publications describing the same translocation event were synonymized, ultimately resulting in a data set of 59 reported mitigation translocations.

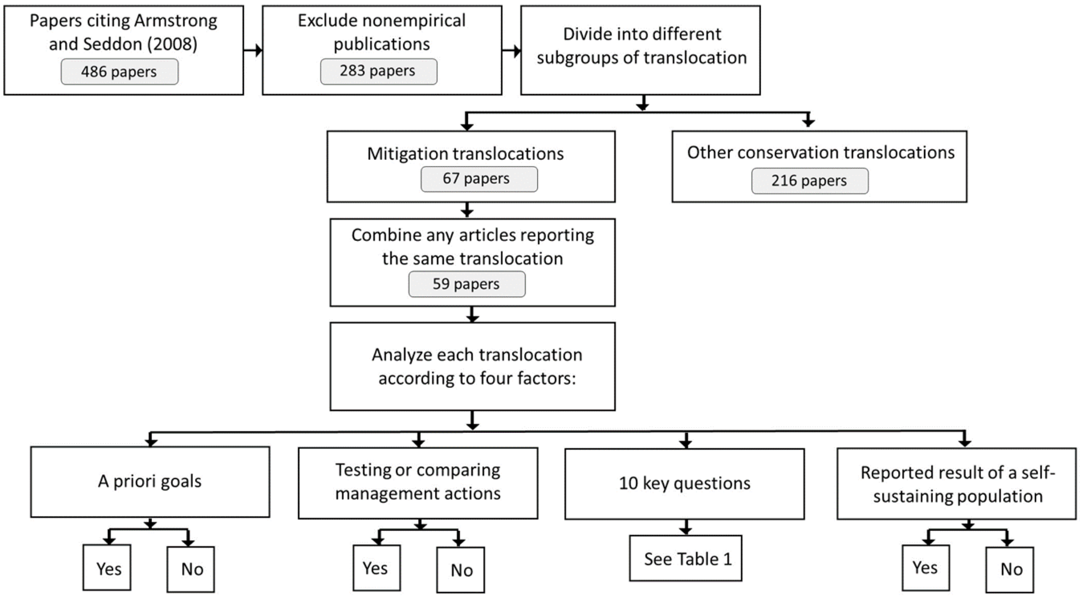

Figure 2.1 PRISMA Flowchart illustrating how the data were subdivided and analysed. Refer to Table 2.1 for the listed ten key questions in translocation biology from Armstrong and Seddon (2008).

An article search using the search terms mitigation AND translocation on the Scopus database (29 July 2020) returned 200 articles published from 2008 to 2020. This number was reduced to 198; 1 article could not be sourced in English (Born 2015), and another could not be sourced (Box et al. 2019). Only 28 of the 198 articles were actually mitigation translocations according to our rubric (Table 2.1). Only one of these 29 articles cited Armstrong and Seddon (2008), the seminal work in the last 20 years on how to maximise the success of a translocation. Therefore, for the purposes of this review, the data set was derived only from articles that cited Armstrong and Seddon (2008), as a way to more easily locate articles that involved wildlife translocations, rather than studies where the term translocation was used in another sense. Furthermore, we believed it was critical to identify projects that were clearly aware of the importance of an evidence-oriented project framework. In the context of wildlife translocations, the most highly regarded of such frameworks, judging by citations, is that of Armstrong and Seddon (2008), and we assumed that awareness of this framework underpinned the test of how many studies actually followed the suggested adaptive management framework.

Because the IUCN (2013) encouraged mitigation translocations to follow the same protocols as other conservation translocations, we analysed whether mitigation translocation articles addressed the 10 key questions developed by Armstrong and Seddon (2008) said to be critical in maximising translocation success. The whole body of text from each publication was searched for any consideration of Armstrong and Seddon's (2008) 10 key questions in reintroduction biology (Table 2.1). Question 7 was removed from the analysis due to the assumption stated by Armstrong and Seddon (2008) that all translocations implicitly consider this question, and question 10 was removed because none of the studies addressed this question; most included only a single species translocation. Question 2 was divided into pre-release management and post-release management. Translocations were only considered self-sustaining if reported as such within the publication. Articles reporting model predictions or recommendations were removed from analyses that tested the influence of variables on the self-sustaining nature of a population because this factor was not applicable. This removal reduced the data set to 54 articles. Articles were also categorized according to taxa. The percentage of articles per taxon group was used to determine which groups had the most mitigation translocations undertaken.

To determine whether the questions answered influenced translocations becoming self-sustaining, we analysed a 9 × 2 contingency table to see if the number of translocations that answered the nine questions (questions 7 and 10 removed and question 2 split into two questions) differed between translocations that were self-sustaining and those that were not. Questions were then grouped into four categories (population establishment [Q1-2], population persistence ([Q3-4)], metapopulation [Q5-6], and ecosystem [Q8-9]) and analysed using a 4 × 2 contingency table to determine if translocations that resulted in self-sustaining populations included an equal number of questions from each category.

Mitigation translocation publications were also classified according to the *a priori* hypotheses. Publications that stated goals for the article but for which an existing data set (the goals had no influence on the design or structure of the original translocation event) was used were not considered to test *a priori* hypotheses. To determine whether project success was influenced by the inclusion of *a priori* goals, we used a Fisher's exact test.

To assess whether mitigation translocations applied an active adaptive management approach (Palmer et al. 2016), the data set was divided into two categories: publications that included one or more management techniques in addition to the translocation, such as supplementary feeding and that did not obviously include a management action other than the translocation intervention. This distinction follows the analysis by Taylor et al. (2017), who investigated whether reintroductions in general were effective as an applied science and considered the comparison of management actions as “studies that directly assist decisions by explicitly comparing alternative management actions.” Because the selection of mitigation translocation over inaction in the face of anthropogenic disturbance represents an *a priori* expectation that translocation can avoid wildlife mortality, we considered management actions as only those that included additional management efforts applied in conjunction with the translocation to assist decision makers to determine the most effective means of mitigation translocation for the future. To determine if implementing management techniques affected a translocated population's ability to become self-sustaining, we used a Fisher's exact test. A one-tailed p value was selected, because we assumed management actions would improve, rather than decrease, the probability of a translocation being self-sustaining.

Chi Square analyses (expected values calculated as 50% of the total) were employed to determine if mitigation translocations were divided equally into self-sustaining and not self-sustaining translocated populations. Chi-squared analyses were also used to determine if more key questions or more categories (establishment, persistence, metapopulation, and ecosystem) were addressed by different studies. For these analyses, expected values were based on the assumption that all questions were equally likely to be addressed.

2.6 Results

Most publications focused on mammals (37%), birds (29%), and herpetofauna (19%); invertebrates (2%) were the least considered taxa. Significantly fewer mitigation translocations resulted in self-sustaining populations than nonself-sustaining populations (χ^2_1 = 21.41, *p* <0.001) (Fig. 2.2A). No difference was found in the questions addressed between self-sustaining and nonself-sustaining translocated populations (χ^2_8 = 2.18, *p* = 0.975).

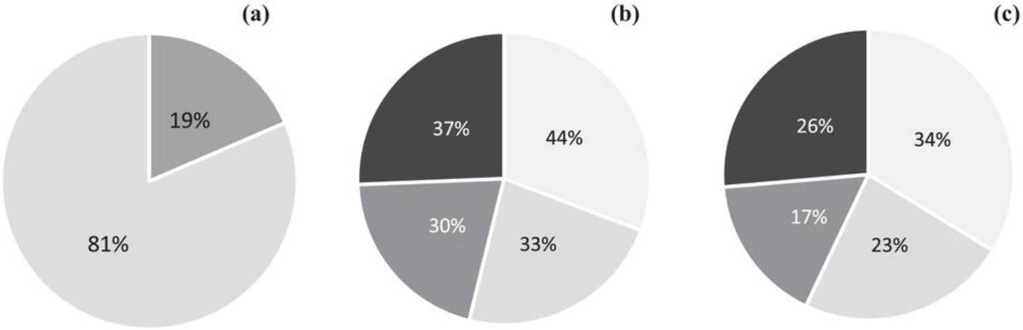

Figure 2.2 The percentage of **(A)** mitigation-translocation articles that reported the result of a self-sustaining (dark grey) or non-self-sustaining population (pale grey), **(B)** self-sustaining populations resulting from mitigation translocations that addressed questions at the establishment (off-white), persistence (pale grey), metapopulation (dark grey), and ecosystem (black) levels, and **(C)** non-self-sustaining populations resulting from mitigation-translocation articles that addressed establishment (off-white), persistence (pale grey), metapopulation- (dark grey), and ecosystem- (black) level questions.

There was no difference in the number of questions answered within the four categories (population establishment, population persistence, metapopulation, and ecosystem) between self-sustaining and nonself-sustaining populations (χ^2_3 = 4.78, *p* = 0.188) (Figs. 2.2B & 2.2C). There was also no overall difference in the number of questions addressed in each of the four categories (χ^2_3 = 2.98, *p* = 0.395). In total, 66% (39 studies) of all mitigation translocations addressed establishment-level questions in translocation biology, whereas 78% (46 studies), 58% (34 studies), and 88% (52 studies) addressed persistence-, metapopulation-, and ecosystem-level key questions, respectively (nonexclusive). The inclusion of *a priori* goals had no influence on production of self-sustaining translocations (1-tailed *p* = 0.550) (Figs. 2.3A & 2.3C). Testing of management techniques did not produce more self-sustaining translocations than not testing management techniques (1-tailed *p* = 0.611) (Figs. 2.3C & 2.3D).

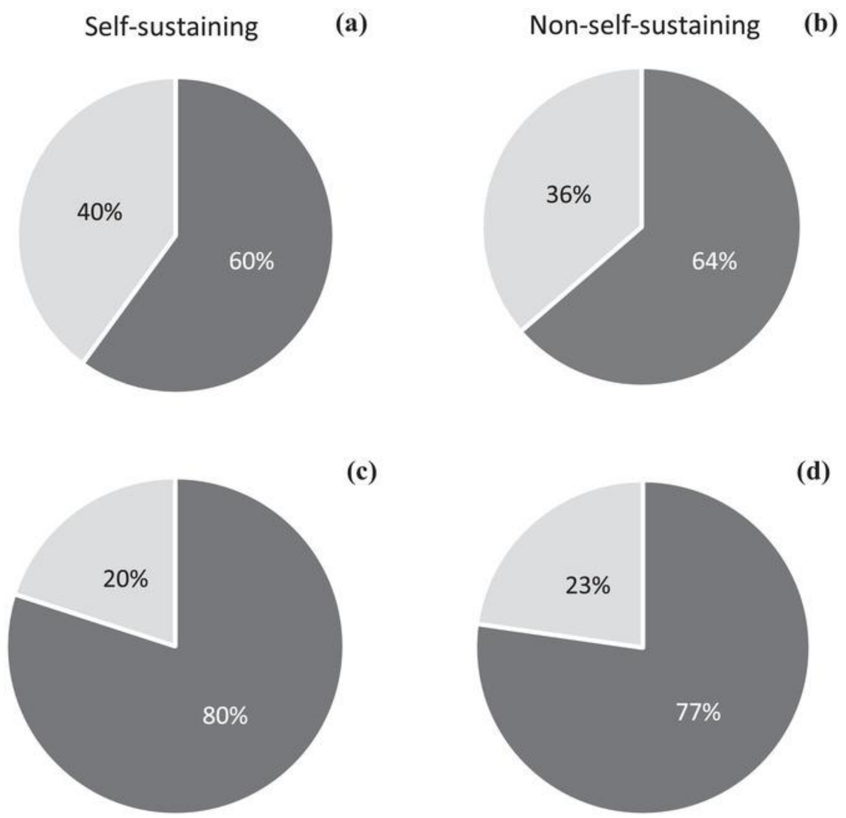

Figure 2.3 The proportion of **(A)** self-sustaining and **(B)** nonself-sustaining mitigation translocations that did (dark grey) or did not (pale grey) state *a priori* goals and the proportion of **(C)** self-sustaining and **(D)** nonself-sustaining mitigation translocations that tested (dark grey) or did not test (pale grey) at least one management technique.

2.7 Discussion

Less than one-quarter of mitigation translocations resulted in the establishment of self-sustaining populations. Because there is likely a strong bias by consultants contracted by private companies against publishing unsuccessful translocation efforts, the overall proportion of self-sustaining populations is likely to be lower than we found. This supports previous evidence that successful outcomes are less likely for translocations driven by factors such as economic motivations or human-wildlife conflict rather than primarily conservation motivations (Fischer and Lindenmayer 2000; Germano et al. 2015; Sullivan et al. 2015; Wolfe et al. 2018). Success in mitigation translocations is also less likely to be considered as the establishment of a self-sustaining population. For many mitigation translocations, the end result is considered the effective resolution of the human-wildlife conflict rather than any longer-term goals of establishing new populations (Massei et al. 2010).

Although many mitigation translocations can involve threatened species, as we found (89% of studies), they can also involve abundant or relatively common species, where conservation-oriented factors, such as population viability, are less likely to be considered (Massei et al. 2010). We found that mitigation translocations largely ignored the wider implications of a translocation event at the metapopulation and ecosystem levels. Where specific research questions were included in planning, mitigation translocations focused on a small number of readily answerable questions principally around whether the focal species was native to the recipient ecosystem (question 8), the appropriate population size (question 1), and presence of habitat (question 3). Establishment-level questions are generally easier to answer but are not usually the ultimate determinant of translocation success (Armstrong and Seddon 2008). Although the majority of mitigation translocations are generally not focused on improving science, they still need to be conducted following the same process of design and implementation as conservation translocations if they are to successfully prevent species mortality in response to land-use change or human-wildlife conflict (IUCN SSC 2013). Therefore, whether or not mitigation translocations also have a conservation objective beyond addressing a human-wildlife conflict, or varying conservation objectives according to different taxa, the translocation requires strategic planning and adaptive management to be considered an effective translocation with sound ethics, according to IUCN (2013) guidelines.

Despite the need for mitigation translocations to follow an adaptive management approach, technical difficulties and resource demands limit the likelihood of the critical metapopulation- and ecosystem-level questions being addressed during mitigation translocations (Taylor et al. 2017), perhaps explaining their lack of success. Because mitigation translocations are crisis responsive, the short time frame over which they operate probably limits any ability to address these more challenging questions (Berg 1996). However, failure to address questions at the metapopulation and ecosystem levels will likely lead to continued project failure as reported for the majority of studies within this review due to inappropriate distribution of individuals among sites, introduction of non-native parasites, or flow-on effects for the ultimate species composition (Waldron et al. 2013). In the future, managers, therefore, need to be more strategic with their use of conservation dollars; a broader acceptance of the time and money required to achieve effective mitigation translocations is needed. With the more reliable, effective reduction of wildlife mortality from land-use change through translocation, mitigation translocations can play a larger role within conservation planning.

For mitigation translocations to become integrated within conservation planning, it is also necessary to reduce ambiguity around what constitutes a mitigation translocation. As a supply-driven method (Germano et al. 2015), mitigation translocations can include those aligned with preventing species extinction, as was the case for many studies within this review, but can also include the removal of nuisance animals, commonly snakes, or large carnivores near urban areas, as well as consultants walking in front of bulldozers relocating wildlife disturbed in a development footprint. The latter two forms of translocation are performed by people with a range of expertise, adhering to an ambiguous range of legislative and policy controls. Wildlife translocated as the result of these human-wildlife conflicts, particularly herpetofauna, often do not survive (Reinert and Rupert Jr 1999; Nowak et al. 2002; Sullivan et al. 2015; Devan-Song et al. 2016; Wolfe et al. 2018). Government regulators may also be unaware of the ethics criteria set by the IUCN (2013) and so may fail to impose them on the proponents of the translocations. The added challenge is that proponents may be reluctant to embrace the genuine costs of conducting rigorous mitigation translocations unless required to do so by legislation. Even in countries, such as Australia, which has a much more rigorous legislative and policy framework than many places in the world, the appropriate legislation (at least at the national level), the Environment Protection and Biodiversity Conservation Act 1999, imposes no strict and specific requirements for a translocation plan or post-translocation monitoring. Also, with no clear strategy for recipient site selection, state regulators have to work out what to do with populations with no prior management plan and no overarching strategy, which can lead to ineffective conservation outcomes. Even for mitigation translocations aligned with broader conservation objectives, as were the majority in our study, there is little evidence for mitigation translocations following a sound scientific paradigm (Germano et al. 2015; Lennon 2019). Because the grey literature on mitigation translocations is impenetrable, the scale of the problem is likely much larger than observed within the scientific literature. Therefore, what we and others found within the literature (Germano et al. 2015; Sullivan et al. 2015; Lennon 2019) shows a shuttered glimpse into an action that may appear to be conservation in principle, but in actuality, it may threaten wildlife. Until there are real numbers and a real understanding within science and policy and outside it, this is a major challenge to conservation.

The inclusion of *a priori* goals and testing or comparing management options did not influence the result of self-sustaining mitigation translocations. However, this does not negate the value of including these in translocation design because their primary reason for inclusion was to use the least amount of conservation dollars and maximise the knowledge gained for future translocations (Armstrong and Seddon 2008). To continue improving translocation techniques and facilitate practitioners selecting the most appropriate management actions, it is critical to compare management techniques (Taylor et al. 2017). Broadly speaking, because many of the mitigation translocation studies we analysed were not built around a sound scientific paradigm, there is still improvement required in linking translocation science with project implementation. Yet, this lack of comparative approach did not appear to hinder the success of these translocations, raising the question of whether such comparisons are as relevant for mitigation translocations as those motivated by other intentions. We posit that they are, but that success is often poorly defined and poorly assessed in mitigation translocations (Germano et al. 2015). The desire of developers to continue justifying mitigation translocation as a management tool for conservation in lieu of protecting natural areas from development has hindered long-term monitoring and resolution of a consistent definition of success. We found that the questions that require long-term monitoring, including the impacts to the recipient ecosystem (question 9) and carrying capacity of different recipient sites (question 6), were addressed the least (including question 10 which was never addressed because the majority of translocations were for single species). We argue that a mere assessment of success as the resolution of the original land-use conflict through relocation is not enough (Massei et al. 2010) and that the aim to establish a self-sustaining population integrated with the recipient ecosystem (Griffith et al. 1989), and the larger metapopulation structure (Armstrong and Seddon 2008) should be the end goal of all translocations.

Translocation is very much a conservation tool of the Anthropocene (Corlett 2015), encompassing everything from mitigation translocations and reintroductions to assisted colonisation (Lunt et al. 2013) and rewilding (Jørgensen 2015; Sweeney et al.

2019). With bold aspirations comes substantial capacity for unintended consequences for both ecology (May and Spears 2011; Abbott and Haynie 2012; Colman et al. 2014) and evolution (Laikre et al. 2010). Avoidance of this risk is the motivation for a robust scientific rationale to underpin all translocations and reintroductions (Armstrong and Seddon 2008). Mitigation translocations have in the past been criticized for lacking this robustness (Germano et al. 2015; Sullivan et al. 2015), and there has been an ongoing call for better evidence to support translocation biology over the last two decades (Sutherland et al. 2010; Kemp et al. 2015; La Haye et al. 2017; Taylor et al. 2017). In common with other forms of conservation translocation (Taylor et al. 2017), mitigation translocations have not yet reached their potential as an effective applied science.

There are a number of factors that limit the capacity of a mitigation translocation to meet the same strict scientific rationale as is common for other conservation translocations. In situations with no effective means of diverting development away from significant natural areas, managers are left to make the best of a bad situation (Berg 1996). The speed and scope of infrastructure development can lead to developers removing individual organisms from a site and conducting translocations in an *ad hoc* manner without any feasibility analysis (Gardner and Howarth 2009). We argue that without the capacity to conduct a well-planned translocation, at least addressing as many key questions in translocation biology as possible, such translocations should be avoided wherever possible. It is also critical to address the apparent imbalance between management actions and monitoring. Monitoring as standard practice for mitigation translocations will help inform best practice and reduce the loss of conservation dollars spent on ineffective management techniques. However, of greatest importance is the adequate protection of natural areas through application of the mitigation hierarchy of avoidance, minimisation, and compensation or offsetting when implementing disturbance activities (Gardner et al. 2013; Ekstrom et al. 2015). Translocation should be the final option within a hierarchical decision framework for mitigating biodiversity loss, and all other options for avoidance and minimisation of disturbance should be exhausted prior to the selection of translocation as a management option.

In the event that mitigation translocation is the most, or only, appropriate course of action, then it is critical to maximise its efficacy as a management tool. It is, therefore, of high priority for future mitigation translocations to follow the same strategic framework as other conservation translocations, namely to promote and monitor long-term success through planned experimental research at the population, metapopulation, and ecosystem levels of the translocation. This raises questions as to the appropriate agency, time frame, and investment required to conduct these programs. With the recognition that mitigation translocations are not so simple as the altruistic (or, in some cases, mandated) aim of capturing the animals and releasing them into whatever habitat is available nearby, comes the recognition that they probably require greater investments of time, resources and, most importantly, expertise than they are currently provided. Without significantly greater investment, many mitigation translocations will continue to simply change the location in which their target animals are killed.

2.8 Acknowledgements

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2.10 Supplementary material

Mitigation translocation papers included within the analysis

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Chapter 3. Predators in a mining landscape: threats to a behaviourally unique, endangered lizard

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Photo: Holly Bradley

3.1 Abstract

Patchy resource distribution can cluster predator activity around areas of highest productivity in ecosystems. For the endangered Western Spiny-tailed Skink (*Egernia stokesii badia*) in Western Australia, the log piles that they permanently inhabit in an otherwise patchy, arid landscape, represent a potentially reliable, high abundance food source for predators. Not only are encounter rates by potential predators of *E. s. badia* likely to be influenced by vegetation structure at the microhabitat scale, but *E. s. badia* also occurs in a region where mine sites and associated infrastructure, such as landfill sites, likely concentrate generalist predators (e.g. Feral Cats and corvids). We assessed the influence of the presence of coarse woody debris (CWD) and distance to landfill on predator behaviour towards *E. s. badia* through plasticine model experiments, unbounded point count bird surveys and camera trapping. We found that CWD inhabited by *E. s. badia* attracted a greater relative activity of corvids compared with uninhabited CWD, or control sites without CWD. The relative activity of corvids and predatory birds combined increased with decreasing distance from landfill. Preferential hunting by corvids at CWD inhabited by *E. s. badia* compared to both uninhabited CWD and open sites suggests that inhabited CWD may be targeted by generalist predators in the region, and that adaptive management may be required for species conservation around active mining areas.

Keywords

Egernia stokesii, mitigation translocation, optimal foraging, predation, threatened species management

3.2 Introduction

Predator-prey interactions can be influenced by the structure of the surrounding environment, which can both improve and reduce predation success, influencing either the ability of predators to catch prey, or of prey to avoid predators (Heithaus et al., 2009; Schmidt & Kuijper, 2015). For example, open spaces may improve the ability of prey to forage, but also increase their predation risk (Hernández & Laundré, 2005; Hebblewhite & Merrill, 2009; Rieucau et al., 2009). Alternatively, for predators, areas with the greatest quantity of prey may also have the lowest catchability (ease of prey capture; Hopcraft et al., 2005). Predators must, therefore, make a trade-off between hunting in areas where prey is more easily caught but are potentially less abundant and areas where prey encounter rates are highest but where their capture may be more challenging (Schmidt & Kuijper, 2015).

As rainfall can be a major factor limiting ecological processes, ecosystems such as arid regions and deserts often have low primary productivity (Pianka, 1967), plus a patchy distribution of resources (Aguiar & Sala, 1999; McAllister et al., 2011). Predators are, therefore, likely to target areas of highest prey activity to optimise foraging; in arid landscapes this includes areas of higher productivity, such as watering holes, which attract congregations of prey (Valeix et al., 2010; Brawata & Neeman, 2011). Many small to medium-sized animals rely upon log piles, or coarse woody debris (CWD), as shelter, thermal refuges and as refuges from predators, particularly in Australian ecosystems (Chapple, 2003; Sumner, 2006; Jacobs et al., 2007; Christie et al., 2012). Indeed, CWD frequently harbours higher faunal abundance and diversity compared with surrounding habitats in a number of ecosystems (Loeb, 1999; Lohr et al., 2002; Kappes et al., 2006; Craig et al., 2012). Many species reliant upon CWD have small home ranges (Sumner, 2006; Christie et al., 2012), with some skinks travelling as little as 0–5m per month (Sumner, 2006).

The Western Spiny-tailed Skink (*Egernia stokesii badia*) is an endangered Australian lizard dependent upon CWD for long-term shelter and predator refuge (Pearson, 2012). The skinks produce latrine piles just outside inhabited logs (Lanham, 2001), which may create a long-lasting olfactory cue to potential predators. This could make inhabited CWD even more attractive to potential predators of *E. s. badia* which use olfactory cues to hunt (Garrett & Card, 1993; Hughes et al., 2010), such as Feral Cats, *Felis catus*, Foxes, *Vulpes vulpes* (Desmond & Chant, 2001; Pearson, 2012), snakes, and varanid lizards (Arida & Bull, 2008). The creation of adjacent communal latrines by *E. stokesii* and their behaviour of often spending their time at their core refuge site (Duffield & Bull, 2002) therefore potentially creates a more predictable target for predation. These skinks also do not employ predator-avoidance tactics that have been observed for other log-dwelling species, such as lizards which defecate distantly from their home crevices (Lanham, 2001), or that are solitary (Chapple, 2003). Instead, *E. s. badia* have keeled scales and highly spinose tails which are probably anti-predator adaptations, preventing removal of the animal from crevices and hollows by predators (Arida & Bull, 2008). The skinks are also highly cryptic, with a colour pattern camouflaging against the red earth of their open woodland habitat. CWD supporting *E. s. badia* may, therefore, represent a high abundance, but low-detectability and low-catchability prey resource for their predators within the patchy, low-productivity heterogeneous landscape of the semi-arid Mid West region in Western Australia, in which the subspecies occurs.

In addition to small-scale habitat structure, such as CWD, influencing predator-prey dynamics, wide-scale habitat modification due to mining activity can also impact species interactions across the landscape. For example, the development of linear infrastructure including roads can act as predator highways and alter how introduced predators (such as Feral Cats and Foxes) utilise the landscape (Raiter et al., 2018). In addition to direct clearing for ore extraction, mining has multiplier effects on the degradation of the environment, such as general edge effects (Majer, 2014; Cross et al., 2021). We predicted, therefore, that the encounter rate of predators, particularly introduced and generalist species (e.g. native corvids and introduced Feral Cats) with CWD will increase with decreasing proximity to the active mine site and associated infrastructure, in particular the landfill site. For this study, we hypothesized that in semi-arid open eucalypt woodland typical of the Mid West region of Western Australia, areas of highest abundance and lowest catchability primarily include fallen log piles (i.e., CWD). We predicted that the optimal hunting strategy of predators would be to target areas of highest abundance and lowest catchability and that predators would target CWD with olfactory cues marking occupancy. This translates to the prediction that predators would be more likely to hunt at sites occupied by *E. s. badia* than sites where individuals were not resident, and that predator encounter rates with CWD would be greater at log piles in closer proximity to landfill. To test these predictions, we examined (a) if predators were more likely to actively hunt at inhabited CWD compared with uninhabited CWD or open sites (no CWD); (b) if predator relative activity was higher at inhabited CWD, compared with uninhabited CWD or open sites; (c) if dispersal behaviour by *E. s. badia* between areas of CWD was likely to significantly increase mortality risk; and (d) if predator relative activity significantly increased with proximity to landfill. To gain a comprehensive understanding of the complex predator-prey dynamics within this system we employed three different sampling techniques: plasticine model experiments, unbounded point count surveys and camera trapping.

3.3 Methods

3.3.1 Study area

The study area was within an iron ore mining tenement in the Mid West region of Western Australia (29°10'54.0"S, 116°32'55.1"E; Figure S3.1). Available sites were limited by the location of inhabited CWD within eucalypt woodland of the arid to semi-arid zone of the Mid West, and could be no more than 500m from the access track (stretching approximately 45km long east to west), due to logistical constraints. Inhabited and uninhabited CWD (confirmed by the presence/absence of a latrine pile) for the following studies were randomly selected from monitoring maps developed by the mine site environmental team, and were of similar structure, generally a dead fallen tree with a number of hanging branches (Figure 3.1). As the inhabited CWD ranged in length from approximately 7-20m and height approximately 1-10m, this variation was similarly captured in the selected uninhabited CWD.

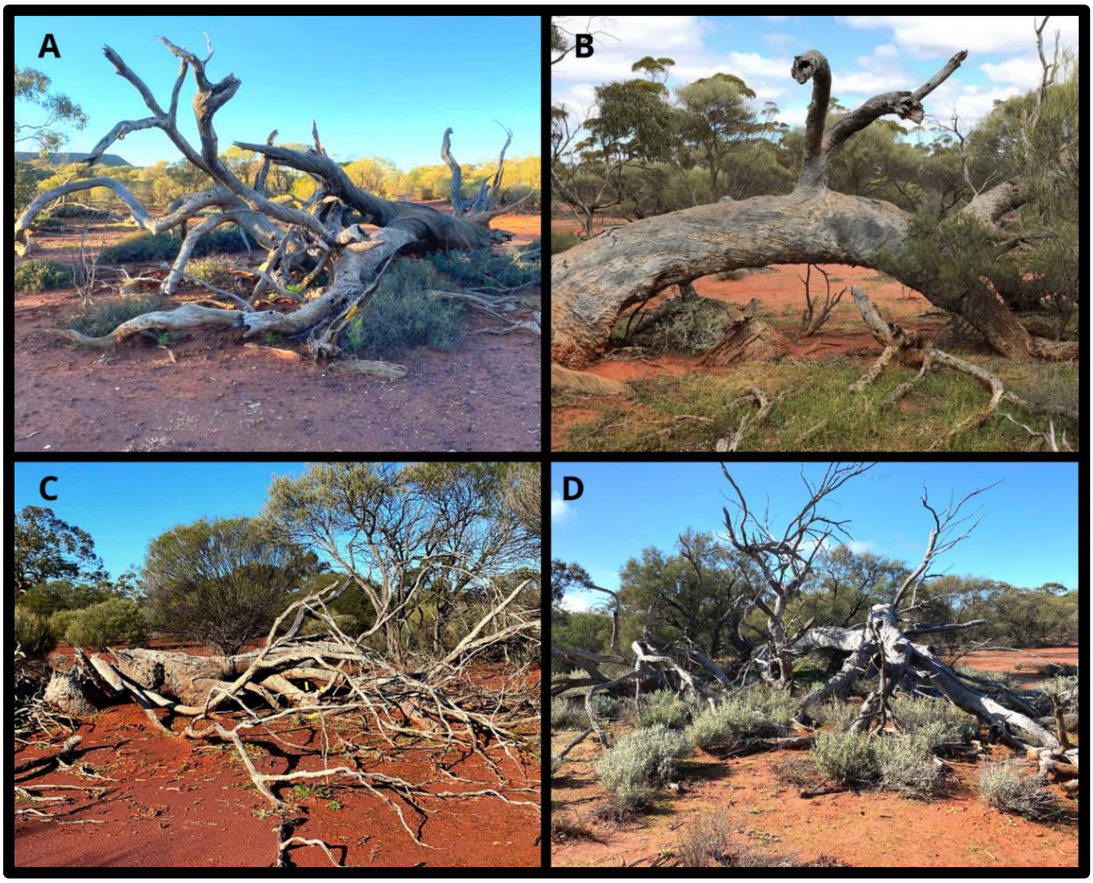

Figure 3.1 Examples of typical CWD classified as uninhabited **(A & B)** and inhabited **(C & D)**, with the general structure of a single fallen dead tree, with numerous hanging branches and hollows/crevices.

Open sites (no CWD) were selected at random, also within 500m of the track, and with similar vegetation to the sites with CWD. To identify the area in which we could safely work within OHS guidelines, a polygon shapefile of the area within 500m of all vehicle-accessible tracks on the site was generated using the "buffer" algorithm in QGIS. In order to avoid spatial autocorrelation with log piles, all identified log pile sites were also buffered by a distance of 100m, again using the "buffer" algorithm in QGIS, and then removed from the candidate area using the "clip" algorithm in QGIS. Twenty points were then generated within the candidate area using the "random points inside polygons" algorithm in QGIS. All random points were generated a minimum of 100m apart, and ground-truthed to match the same open eucalypt woodland habitat surrounding the CWD sites.

3.3.2 Unbounded point count bird surveys

We used unbounded point count bird surveys to determine predator relative activity at sites with inhabited and uninhabited CWD, and open sites. We surveyed five inhabited CWD sites, five uninhabited CWD sites and five open sites (15 sites total), all at least approximately 100m apart to ensure independent sampling points. Ten-minute unbounded point counts of potential predatory bird species (Table 3.1) were recorded at each site, repeated daily for six days, during spring 2018 (9th to 15th October) and again during autumn 2019 (2nd to 7th May). Each survey was preceded by a two-minute waiting period to avoid any disturbance influencing the survey outcomes. Order of sites visited was randomised each day to capture the range of bird activity times, from 07:00 to 17:00. For each recording, the same observer was situated at a count station, and recorded the number of individuals per species detected (through either call or visual observation) in an unbounded direction. This method follows the widely used five-minute bird count method (Hartley, 2012), extended for a further five minutes to increase the probability of detection.

3.3.3 Plasticine models

This experiment investigated the effect of skink position (‘open’, ‘beneath vegetation’, and ‘exposed’) and habitat selection (CWD versus open sites) on predation frequency and severity (fatal versus non-fatal). To assess predation pressure in relation to CWD, we created replica models of skinks simulating different behaviours, and placed them at sites with and without CWD. Plasticine models have been widely used to determine predation rates on reptiles (Vervust et al., 2007; Daly et al., 2008; Sato et al., 2014; Bateman et al., 2017). Our methods follow those of Wuster et al. (2004), Niskanen and Mappes (2005) and Sato et al. (2014), using non-toxic sculpting clay (Plastiplay; Brian Clegg, Rochdale, Lancashire, OL12 0HQ, United Kingdom), moulded to simulate the mean size, shape, and general appearance of adult *E. s. badia*. Post-construction, models were coated with red dirt (retrieved on-site), to achieve approximate colouration of field specimens, increase the structural integrity of models by reducing the likelihood of clay softening, and to remove as much of the human scent as possible from the models.

The experiment followed a two-factor design, with five models per position: ‘beneath vegetation’ (under mid- or understorey vegetation cover), ‘exposed’ (on a live/dead branch or log surface) and ‘open’ (in open space), placed at five different CWD sites, and five models per position ‘beneath vegetation’ and ‘open’ at five separate open sites (due to the absence of CWD at open sites for ‘exposed’). 125 models were used in total. Positioning of models was limited by the habitat structure. Models in the ‘beneath vegetation’ position were placed beneath the cover of understorey vegetation where possible, and in the absence of understorey, under the lowest mid-storey cover available. ‘Exposed’ models were placed on living and dead branches, depending upon the nature of the CWD, if part of the tree was still alive. The height of models placed on branches/logs varied from approximately 0.3-1.5m from the ground. Models in the ‘open’ position were limited in range from the log pile from approximately 0.3-2.0m, and was limited to the availability of shade, as at least partial shade was required to prevent the clay models from softening. The site types were defined as: (A) CWD with a log pile similar to those inhabited by *E. s. badia*; and (B) open sites with no CWD present. CWD inhabited by *E. s. badia* were avoided to prevent encouraging predator activity.

Models were left out over seven days in winter/spring 2018 (23rd August to 1st September), and six days in autumn 2019 (11th to 17th April). Each model was checked daily and the following recorded: a) if an attack occurred; b) where on the model any attack was evident; c) the form of predation (e.g. model removal or visible indentations); d) what predator made the attack (confirmed through camera trap records and comparison of peck/bite marks); and e) the severity of attack. Severity scores followed the definitions by Smithies (2016): (0) no attack; (1) non-fatal attack – light scratch/peck on back, tail or flipped with no markings; (2) fatal attack - deep wound on head or back, or model removed. Each model was smoothed after each daily recording to remove previous attack marks, and interactions with non-predator species such as rodents were excluded. Corvid predators (*Corvus orru, C. bennetti,* and *C. coronoides*) were combined as the single observation '*Corvus* sp.' (corvids) for this experiment as well as the unbounded point count surveys and camera recordings, due to the difficulty in differentiating peck marks and distant observations between corvid species.

3.3.4 Camera traps

A single motion-activated camera trap (Reconyx Hyperfire 2 and Reconyx HC500) was placed at each of nine sites during the first survey in 2018 (three uninhabited, three open, and three inhabited) and were placed at all sites (ten uninhabited, ten open, and ten inhabited) during the second round of plasticine experiments and bird surveys in 2019. Cameras were attached to a metal fence post, facing south, depending on the presence of potential vegetation triggers which could activate the motion sensor. Cameras were placed approximately 10m from each area of CWD and angled downwards to capture as much of the CWD as possible, as well as any animals moving on the ground in front of the camera. Camera images were used to identify the predator species responsible for model attacks. Not all models were able to be captured within the camera scope due to their placement around all sides of the CWD, and peck/bite measurements and shape were compared with other previous attacks to help identify the likely responsible predator.

Camera traps were used to determine the predator relative activity, diversity, and hunting behaviour (actively hunting or not) at inhabited CWD sites, compared with both uninhabited CWD and open sites with no log piles. As species of *Egernia* are known to have a large range of mammal, reptile and bird predators (Chapple, 2003), all vertebrates large enough to consume an adult or juvenile skink and that were known to hunt vertebrates were considered potential predators (Table 3.1). Overall, 30 cameras were placed at randomly selected inhabited CWD, uninhabited CWD, and open sites. Sites were scattered randomly, between approximately 100m and 46km apart, due to the scattered pattern of *E. s. badia* colonisation of CWD within the area

(Figure S3.1). Sites were all selected within open eucalypt woodland habitat, to reduce variation between sites. Photos of potential *E. s. badia* predators were recorded between 20th August 2018 and 19th May 2020, with a total trapping effort of 16,057 days (approximately 385,000 hours). Total trap nights for inhabited (5231), uninhabited (5499) and open (5327) slightly varied due to camera malfunctions, such as from water damage. SD cards were downloaded and batteries replaced in the field approximately every three months. Photos were analysed for behaviour using ethograms modified from a similar behavioural study (Meek et al. (2016): Table S3.1). Due to the low likelihood of capturing actual predation events on camera, assumptions of behaviour were based on predators within the camera view. Classification of potential ‘active hunting’ behaviour was surmised from the display of observation, movement, and action responses listed in Table 3.1.

Table 3.1 List of species considered potential *E. s. badia* predators recorded in bird surveys and observed on camera footage, with listed references supporting their known/likely previous predation upon lizards/reptiles.

Predator Species		Point Count Survey	Camera	Reference
Birds				
Australian Magpie	*Gymnorhina tibicen*	X	X	Veltman and Hickson (1989)
Black-breasted Buzzard	*Hamirostra melanosternon*	X		Debus and Czechura (1992), Nunn and Pavey (2014)
Brown Goshawk	*Accipiter fasciatus*		X	Aumann (1988) Aumann (1990)
Bush Stone-curlew	*Burhinus grallarius*		X	Michael and Lindemayer (2010)
Collared Sparrowhawk	*Accipiter cirrocephalus*		X	Czechura et al. (1987)
Corvus sp.	*Corvus bennetti/Corvus orru/Corvus coronoides*	X	X	Stewart (1997), Stuart-Fox et al. (2003), Troscianko et al. (2008)
Grey Butcherbird	*Cracticus torquatus*	X	X	Walters (1980), Nordberg and Schwarzkopf (2019)
Grey Currawong	*Strepera versicolor*	X	X	Stapley (2004)
Grey Shrike-thrush	*Colluricincla harmonica*	X	X	Kutt and Kemp (1997), Baxter (2015)
Pied Butcherbird	*Cracticus nigrogularis*	X		Michael and Lindemayer (2010), Hansen et al. (2019)
Reptiles				
Sand Goanna	*Varanus gouldii*		X	Pianka (1994), Cross et al. (2020)
Black-headed Monitor	*Varanus tristis*		X	Pianka (1994), Cross et al. (2020)
Perentie	*Varanus giganteus*		X	King et al (1989), Pianka, (1994)
Mammals				
Wild Dog/Dingo	*Canis lupus/Canis lupus dingo*		X	Doherty (2015), Doherty et al. (2018)
Feral Cat	*Felis catus*		X	Pearson (2012), Stobo-Wilson et al. (2021)

3.3.5 Data analysis

Analyses were conducted in *R 4.04* statistical environment (R Core Team, 2016). To determine whether the number of individual predatory birds recorded during unbounded point count surveys differed according to site type and distance to the landfill, we ran a generalised linear mixed effects model (GLMM) with a Poisson distribution, in the package lme4 (Bates et al. 2011), with the number of individual predatory birds as the dependent variable, site type (uninhabited CWD, inhabited CWD, and open sites) and distance to landfill as the fixed effects and season (autumn or spring), site (replicates 1-5 of each site type) and day (1-6) as the random effects. Due to the scattered nature of sites (because of the natural variability in availability of log piles), distance to landfill was a continuous variable. This analysis was repeated for the number of corvids observed according to site type and distance to landfill.

To determine whether differences in the number of fauna attacks on plasticine models (dependent variables) varied according to site type (uninhabited CWD and open sites), model position (‘beneath vegetation’, ‘open’ and ‘exposed’), and distance to landfill, we used a GLMM with the number of fauna attacks on plasticine models as the dependent variable, site type (uninhabited CWD and open sites), model position (‘beneath vegetation’’, ‘open’ and ‘exposed’) and distance to landfill as the fixed effects and site (five sites per CWD or open site type), day surveyed (as the surveys were conducted over a week), and season (spring and autumn) as the random effects. To determine if the severity of plasticine model attacks varied according to site type, we used a Wilcoxon rank-sum test with attack severity as the ordinal dependent variable and site type (uninhabited CWD vs open sites) as the independent variable.

To explore whether the number of predators recorded on cameras differed according to site type and distance to landfill, we ran a GLMM with a Poisson distribution, with the number of predators (reptiles, mammals and birds combined) as the dependent variable (adjusted to a relative abundance index, the number of sightings per 100 trap nights), site type (uninhabited CWD, inhabited CWD, and open sites) and distance to landfill as the fixed effects and site (replicate) as the random effect. We then repeated this analysis on the number of mammals alone, birds alone, reptiles alone, then corvids alone and Feral Cats alone, in five separate GLMMs.

To assess whether the proportion of predators detected on camera traps that (a) actively hunted, and (b) did not actively hunt, differed according to site type, we calculated the proportion of predator sightings which exhibited potential active hunting behaviour at each site, and analysed these data using a Kruskal-Wallis rank sum test with site type (uninhabited CWD, inhabited CWD, and open sites) as the independent variable. If there was a significant effect of site type, we used a post-hoc Dunn test to determine between which site types there was a difference. We ran this analysis for bird predators (all species combined), Feral Cats alone (42 sightings) and corvids alone (80 sightings). The latter two categories were the two predator taxa with large enough sample sizes to separately compare potential hunting activity at different site types.

3.4 Results

3.4.1 Predator relative activity

Unbounded point count surveys found predatory bird relative activity to vary with site type, with more predatory birds found at inhabited CWD compared to open sites (z = 2.59, p = 0.010). However, differences between inhabited and uninhabited CWD were non-significant (z = -1.87, p = 0.061; Figure 3.2A). Predatory bird relative activity did not differ between uninhabited CWD and open sites (z = 0.86, p = 0.391). The mean number of predatory birds observed also decreased with increasing distance from landfill (z = -3.25, p = 0.001). Unbounded point count surveys also found corvid relative activity to vary with site type, with more corvids found at inhabited CWD compared to both uninhabited CWD (z = -128.8, p < 0.001) and open sites (z = -142.6, p < 0.001), and more corvids at uninhabited CWD than open sites (z = 13.76, p < 0.001; Figure 3.2B). The mean number of corvids observed also decreased with increasing distance from landfill (z = -147.9, p < 0.001).

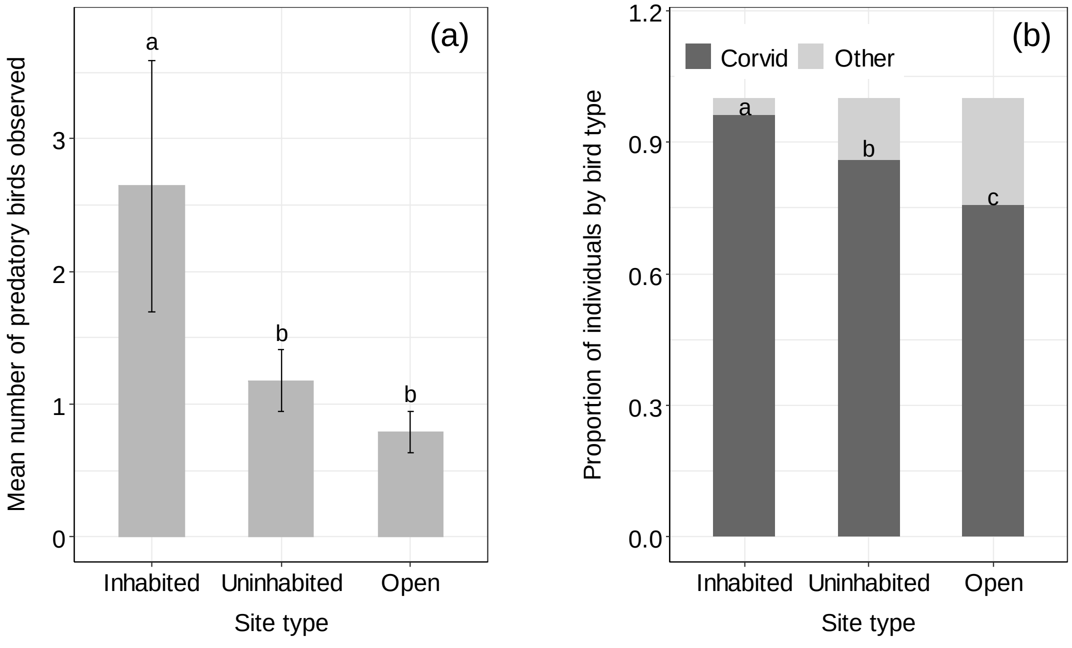

Figure 3.2 (A) The mean number of predatory bird sightings at coarse woody debris (CWD) inhabited by *Egernia stokesii badia,* CWD uninhabited by *E. s. badia,* and open sites, and **(B)** the proportion of predatory bird sightings that were corvids (dark grey) and other (pale grey; Grey Butcherbirds, Grey Shrike Thrush, Pied Butcherbirds and Black-breasted Buzzards combined), observed at CWD inhabited by *E. s. badia,* CWD uninhabited by *E. s. badia,* and open sites. Bird sightings were recorded from unbounded point count surveys, pooled across autumn and winter survey events. Black letters above bars indicate significant differences ($p < 0.05$).

The overall relative activity of predators (bird, mammal and reptile predators combined; Table S3.4) captured on cameras did not differ between inhabited CWD and either uninhabited CWD (z = 0.57, p = 0.572) or open sites (z = -0.99, p = 0.322), or between uninhabited CWD and open sites (z = -0.43, p = 0.667) nor with distance to landfill (z = -0.14, p = 0.886). Neither the number of predatory mammals, predatory birds, predatory reptiles, number of Feral Cats nor number of corvids (separately) differed between inhabited CWD, uninhabited CWD and open sites and none showed any relationship with distance to landfill (Table S3.2).

3.4.2 Attack rates on lizards

The number of attacks on the plasticine models did not differ according to the presence or absence of CWD (z = 0.08, p = 0.936), or distance to landfill (z = -0.29, p = 0.936). Model position also had no effect, with no difference between the number of predator attacks on 'open' models compared with 'exposed' (z = 0.65, p = 0.514) or 'beneath vegetation' models (z = 0.67, p = 0.501), and no difference between attacks on 'exposed' compared with 'beneath vegetation' models (z = 0.16, p = 0.246; Table S3.3). The attack severity on plasticine models also did not vary according to site type (W = 316409; p = 0.887).

3.4.3 Predator behaviour

The proportion of predators exhibiting potential active hunting behaviour did not differ between site types (H_2 = 4.73, p = 0.094). However, the proportion of combined avian predators actively hunting was found to differ between site types (H_2 = 6.49, p = 0.039), with a greater proportion of birds actively hunting at inhabited CWD compared to open sites (Z = 2.53, p = 0.011). There was no difference in the proportion of birds actively hunting at inhabited compared to uninhabited CWD (Z = 1.09, p = 0.276; Figure 3.3) or at uninhabited CWD compared to open sites (Z = -1.41, p = 0.160).

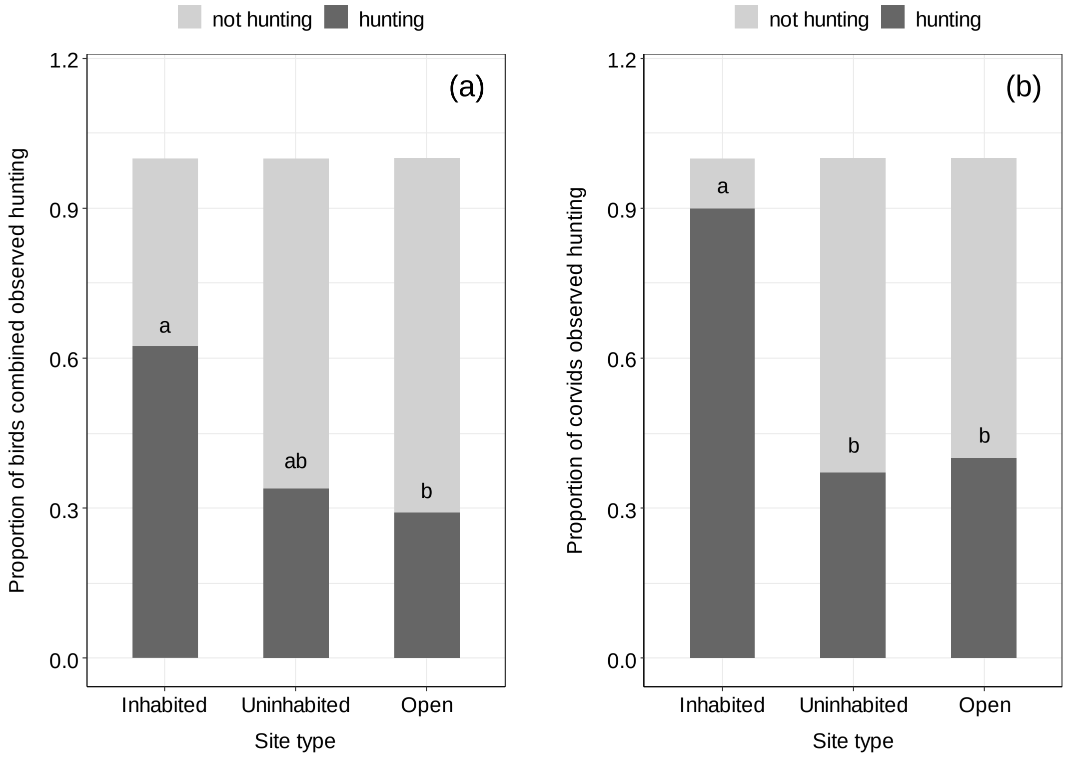

Figure 3.3 The proportion of **(A)** combined bird predators and **(B)** corvids classified as hunting (dark grey) or not hunting (pale grey) when visiting CWD inhabited and uninhabited by *Egernia stokesii badia*, as well as open sites, as observed on camera trap recordings. Black letters above bars indicate significant differences ($p < 0.05$).

The difference in the proportion of Feral Cats exhibiting potential active hunting behaviour at different site types was non-significant ($H_2 = 4.76$, $p = 0.092$). A Feral Cat was also recorded capturing an adult *E. s. badia* on our cameras on one occasion (Figure 3.4). The proportion of corvids actively hunting did differ between site types ($H_2 = 7.04$, $p = 0.030$), where more corvids actively hunted at inhabited CWD, compared to both uninhabited CWD ($Z = 2.18$; $p = 0.029$) and open sites ($Z = 2.49$; $p = 0.013$), and no difference was found between uninhabited CWD and open sites ($Z = -0.44$; $p = 0.662$).

Figure 3.4 Camera trap image of a Feral Cat (centre) at coarse woody debris with an adult *Egernia stokesii badia* in its mouth.

3.5 Discussion

The plasticine model study and presence of predators recorded during the camera trap survey failed to produce significant results or details on predator relative activity or behaviour, due to the low capture rate of individuals and abundance of zeros in the data. This study, therefore, highlights the value of a multi-faceted approach to understanding predator behaviour and predator-prey dynamics, as, in isolation, these two survey methods failed to reveal any significant patterns. Sample sizes are nearly always an issue in ecology, particularly when targeting scarce apex predators (e.g. Feral Cats) in a low productivity arid zone in an El Niño-Southern Oscillation (ENSO) dominated, highly variable climate such as the Mid West of Western Australia. However, supplementing the study with the addition of unbounded point count surveys and behaviour analysis of camera trap images enabled observation of significant patterns in predator behaviour and relative activity, providing an important insight into the predation risk for *E. s. badia.*

The greater relative activity and exhibition of potential hunting behaviour of corvids at sites with inhabited CWD compared to other sites supports the hypothesis that some predators may actively target CWD containing long-term group-living reptile colonies, potentially due to the appeal of a high abundance, reliable food source. The hypothesis that CWD acts as a greater focus for predators than the open landscape surrounding them was partially supported, with the observed higher rate of active hunting and relative activity of predatory birds at inhabited CWD compared with open sites. These results support previous literature suggesting CWD are sites of high species activity (Loeb, 1999; Lohr et al., 2002; Kappes et al., 2006; Craig et al., 2012) and, therefore, potentially centres of activity for predators in a patchy heterogeneous landscape.

Egernia stokesii badia live in a heterogeneous matrix of *Acacia*-dominated shrubland and open eucalypt woodland (Pearson, 2012). Such a patchy landscape gives an advantage to predators that can travel long distances in short amounts of time (Valeix et al., 2011; Carter et al., 2012; McGregor et al., 2016), and are able to target certain habitat patches for improved hunting success (Shettleworth et al., 1988; Doniol-Valcroze et al., 2011; Schmidt & Kuijper, 2015). The most frequent predators seen at all site types over the two years of camera trapping were Feral Cats and corvids, both generalist predators (Dickman, 1996; Piper & Caterall, 2006; Fielding, et al. 2020). Both Feral Cats (Moseby et al., 2009; McGregor et al., 2017), and corvids (Rowley, 1973) have been observed travelling large distances to forage.

Corvids were also abundant across the landscape. As a visually-oriented predator (Stuart-Fox et al., 2003), they were responsible for all attacks on plasticine models, and were the most abundant predatory bird at all site types, especially at inhabited CWD. Corvids, including the Little Crow, *Corvus bennetti* (Stuart-Fox et al., 2003), and the Australian Raven, *C. coronoides* (Stewart, 1997) attack lizards (Troscianko et al., 2008), and our results are consistent with recorded attacks by Australian Ravens on models imitating the Rottnest Island Bobtail (*Tiliqua rugosa konowi*), a lizard of similar size (~27cm in length) to *E. s. badia* (Oversby et al., 2018). Australian Ravens have also been observed attacking and feeding on the tails of live Australian Water Dragons (*Physignathus lesueurii*), of almost one metre in length (Pérez, 2013). The Little Crow is also one of the main predators of Rock Dragons (e.g. *Ctenophorus decresii*), an agamid with an SVL of approximately 78mm (Stuart-Fox et al., 2003). Consistent with these previous reports, and the observed high relative activity of corvids in proximity to inhabited CWD, *E. s. badia* are likely a component of the corvid diet.

Although *C. bennetti, C. orru* and *C. coronoides* are native to the study area, corvid populations have been found to increase in or around human settlements and modified landscapes, such as landfills (Coates & Delehanty, 2004; Marzluff, McGowan, Donnelly, & Knight, 2001; Preininger, Schoas, Kramer, & Boeckle, 2019), as was found in this study. Corvids are known to travel tens of kilometres to access anthropogenic food sources within modified landscapes (Marzluff & Neatherlin, 2006), and anecdotal records from 2004-2020 show that *C. bennetti* in particular were scarce in the area until the construction of the landfill site (M. Bamford personal observation, 2021). As with introduced fauna, overabundant synanthropic species also negatively impact native species through different mechanisms such as disease and predation (Côté, Rooney, Tremblay, Dussault, & Waller, 2004; Peery & Henry, 2010), and can cause devastating impacts to less adaptable, rarer species (Garrott, White, & White, 1993). The global pattern of population expansion by some corvids is of particular concern (Janiszewski, Włodarczyk, & Wojciechowski, 2005; Jerzak, 2001; Marzluff, Boone, & Cox, 1994; Storch & Leidenberger, 2003), as they threaten less abundant native species through predation (Marzluff et al., 1994; Peery & Henry, 2010). An overabundance of corvids within a landscape of anthropogenic disturbance, therefore, risks the suppression and decline of sympatric rarer, less adaptable species such as *E. s. badia*.

Feral Cats have been suggested as a potential threat to *E. s. badia* populations in the past (Desmond & Chant, 2001), particularly to juveniles (Pearson, 2012), and a range of anecdotal evidence indicates they may be one of the skinks' main predators (Lee-Steere 2008). Reptiles in general are a significant proportion of the Feral Cat diet (Dickman, 1996; Paltridge et al., 1997; Doherty et al., 2015; Woinarski et al., 2018), particularly medium-sized reptiles (Stobo-Wilson et al., 2021). Although Feral Cats were not found to actively target CWD, their capacity to hunt adult (as well as the smaller juvenile) skinks was confirmed through camera trap imaging. Therefore, the conclusion that Feral Cats do not target or alter their behaviour according to the presence of skink colonies or CWD must be interpreted with caution as, whilst untargeted, their suppressive effect may still be significant.

As well as identifying the predators of apparent greatest threat to *E. s. badia*, we also explored differences in the behaviour of these predators at different sites. As group-living and scale morphology of *E. s. badia* are potentially adaptive in offsetting high predator activity at CWD, dispersal away from the CWD was predicted to be the riskiest activity for *E. s. badia* to undertake. However, there was no difference observed in the prevalence or severity of predator attacks in the plasticine model study according to their position of ‘open’, ’beneath vegetation’, or ‘exposed’. Nonetheless, this result needs to be interpreted with caution. We only definitively identified corvids attacking our models and although identification of individual corvids was not possible, it appeared as though, once a detection was made, corvids revisited the same site on multiple days to attack models. Similar corvid behaviour has been observed in other studies, where the design of baits and traps required altering to avoid the repeated incidental capture, or disturbance by corvids (Matlack et al., 2006; Way, 2009; Page et al., 2013). Therefore, whilst this study found no trends in attack rate or severity according to model placement, we recommend further testing to understand the level of risk associated with dispersal by *E. s. badia* individuals, particular in the context of future translocations.

3.6 Conclusion

Understanding predator-prey interactions is likely critical to inform the management and conservation of *E. s. badia* populations occurring on mining tenements. Management effort to ameliorate the loss of skink populations through mining activity will likely occur, in part, in the form of mitigation translocations. Our study suggests that predator control is likely to be important for translocation success. Two identified predators in this investigation, Feral Cat and corvids, are attracted to explore novel objects and sites (Church et al., 1994; Heinrich, 1995; Bradshaw et al., 2000; Reina, 2010; Miller et al., 2015), such as translocation sites. As high mortality of individuals is often observed immediately post-translocation likely due to an unfamiliarity with the surroundings (Letty et al., 2000; Pinter-Wollman et al., 2009), and reptiles often move large distances and exhibit homing behaviour post-relocation (Germano & Bishop, 2009), translocated skinks are likely to be more at risk from predator attacks. Our results suggest that the control of predators, including introduced predators, may be important to facilitate the success of future translocations of the endangered *E. s. badia*.

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3.8 Data availability statement

The datasets generated during and/or analysed during the current study are available from the corresponding author on reasonable request.

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3.9 Supplementary material

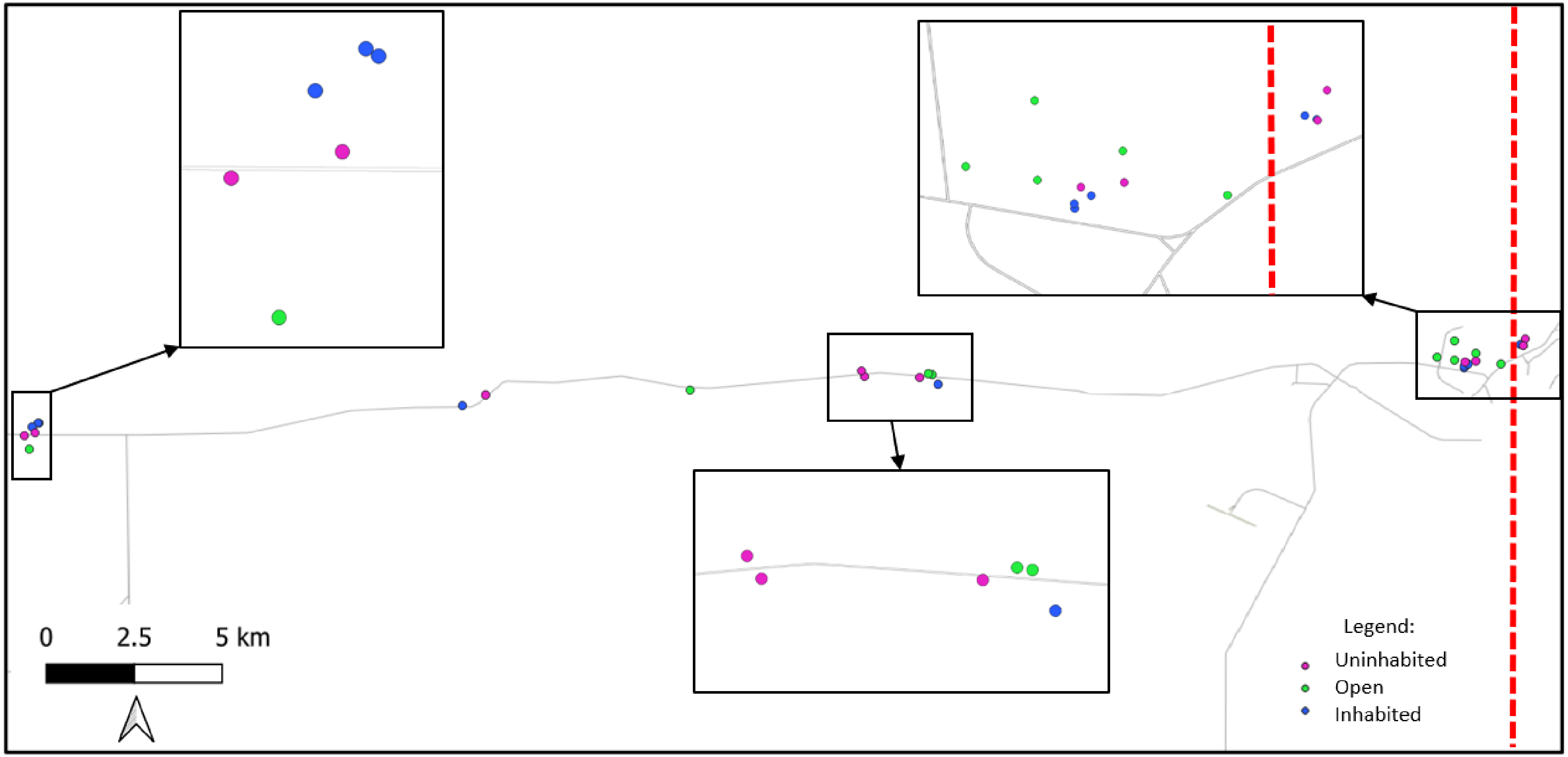

Figure S3.1 Map of the location of all 30 camera traps, including 10 uninhabited CWD (pink), open sites with no CWD (green), and inhabited CWD (blue). The red dotted line represents distance zero (the location of the landfill, from which the distance of sites is recorded). Grey lines represent major tracks.

Table S3.1 Ethograms of the possible quantitative behaviours of potential *E. s. badia* predators recorded on camera traps, when considered potentially 'active hunting'. Predators exhibiting any of the observe/move/act responses were classified as potentially 'actively hunting'.

Predator type	**Observe response**	**Move response**	**Act response**
Mammals, reptiles, and birds	Look/stare at ground, CWD, or model bait	Move towards model bait or CWD/through open space, stalking	Looks down (and sniffs if a mammal) towards ground or log surface, investigates log or ground surface, attacks model bait or prey item

Table S3.2 GLMER results of the effect of site type and distance to landfill on the number of predatory animals. Shown are the values of the *z* statistic and their corresponding *p*-values.

Variable	**Inhabited vs uninhabited CWD**		**Inhabited CWD vs open**		**Uninhabited CWD vs open**		**Distance to landfill**	
	z	*p*	*z*	*p*	*z*	*p*	*z*	*p*
Predatory mammals	0.06	0.949	-0.59	0.558	0.59	0.558	-0.70	0.482
Predatory birds	0.62	0.536	0.87	0.387	-0.26	0.799	0.29	0.774
Predatory reptiles	0.61	0.539	0.57	0.568	0.06	0.951	-0.42	0.678
Feral Cats	0.20	0.839	-0.71	0.479	-0.52	0.605	-0.06	0.950
Corvids	1.01	0.315	-1.02	0.310	-0.01	0.992	-0.09	0.930

Table S3.3 Summary information of the total number of models attacked per model position ('beneath vegetation', 'open', and 'exposed on log'), in each treatment (sites with and without logs), and the proportion of days each model was found attacked, out of the available 325 days.

Treatment		**Model position**					
		Beneath vegetation		Open		Exposed on log	
Site type	Severity	Total	Proportion	Total	Proportion	Total	Proportion
Log	0	313	0.96	306	0.94	303	0.93
	1	3	0.01	10	0.03	11	0.03
	2	9	0.03	8	0.02	11	0.03
No Log	0	307	0.94	302	0.93		
	1	7	0.02	12	0.04		
	2	11	0.03	11	0.03		

Table 3.4. The number of sightings per predatory species observed on camera trap footage during the trapping period, at CWD inhabited by *E. s. badia*, CWD uninhabited by *E. s. badia*, and sites with no CWD.

Predator	**Species name**	**Number of individuals observed**		
		Inhabited	**Uninhabited**	**No CWD**
Australian Magpie	*Gymnorhina tibicen*	4	6	6
Bush Stone Curlew	*Burhinus grallarius*	0	0	1
Brown Goshawk	*Accipiter fasciatus*	1	1	0
Corvid	*Corvus sp.*	10	40	29
Feral Cat	*Felis catus*	9	12	20
Grey Butcherbird	*Cracticus torquatus*	2	3	2
Grey Currawong	*Strepera versicolor*	3	7	8
Grey Shrike Thrush	*Colluricincla harmonica*	4	0	0
Sand Goanna	*Varanus gouldii*	0	0	3
Black-headed Monitor	*Varanus tristis*	0	4	1
Perentie	*Varanus giganteus*	1	1	0
Wild Dog/Dingo	*Canis lupus/Canis lupus dingo*	6	5	5

Chapter 4. Revealing microhabitat requirements of an endangered specialist lizard with LiDAR

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Photo: Holly Bradley

4.1 Abstract

A central principle of threatened species management is the requirement for detailed understanding of species habitat requirements. Difficult terrain or cryptic behaviour can, however, make the study of habitat or microhabitat requirements difficult, calling for innovative data collection techniques. We used high-resolution terrestrial LiDAR imaging to develop three-dimensional models of log piles, quantifying the structural characteristics linked with occupancy of an endangered cryptic reptile, the Western Spiny-tailed Skink (*Egernia stokesii badia*). Inhabited log piles were generally taller with smaller entrance hollows and a wider main log, had more high-hanging branches, fewer low-hanging branches, more mid- and understorey cover, and lower maximum canopy height. Significant characteristics linked with occupancy were longer log piles, an average of three logs, less canopy cover, and the presence of overhanging vegetation, likely relating to colony segregation, thermoregulatory requirements, and foraging opportunities. In addition to optimising translocation site selection, understanding microhabitat specificity of *E. s. badia* will help inform a range of management objectives, such as targeted monitoring and invasive predator control. There are also diverse opportunities for the application of this technology to a wide variety of future ecological studies and wildlife management initiatives pertaining to a range of cryptic, understudied taxa.

4.2 Keywords

Cryptic, *Egernia stokesii,* habitat selection, LiDAR, threatened species

4.3 Introduction

Contemporary wildlife extinctions are occurring at a rate thousands of times greater than background species loss (Ceballos et al. 2010) and are predicted to increase as a result of anthropogenic threats increasing in range and intensity commensurate with increasing human population pressures (Johnson et al. 2017). As a result, threatened species recovery is a major conservation focus around the world (Scott et al. 2010). Fundamental to a broad range of species recovery and wildlife management initiatives is understanding the two main factors that influence habitat use: habitat availability and habitat choice (Johnson et al. 2006). The first restricts the distribution of species only through the quantity of options within the surrounding environment, while the latter is linked to specific adaptations to particular habitats, regardless of what broader spectrum of habitat is available (Johnson et al. 2006). Species limited by habitat choice generally exhibit narrow niche breadth, meaning that they are constrained by the physical conditions under which they can survive and reproduce (Gaston et al. 1997; Devictor et al. 2010). The more specialised the habitat/microhabitat requirements of a species, the more targeted habitat selection is necessary for successful species recovery.

Distribution maps based on historical sightings and species distribution models are often a useful starting point for developing a broad sense of habitat requirements, due to their ability to identify patterns at a coarse scale (Jetz et al. 2008; Razgour et al. 2011). However, the dynamic nature of ecological systems means that species-specific microhabitat and habitat suitability assessments may be critical for finer understanding of a species' requirements (Seddon 2010), and such data are difficult to obtain from conventional distribution models (Tomlinson et al. 2018; Tomlinson et al. 2019). For example, in developing habitat selection protocols for narrow-range plants, Tomlinson et al. (2019) noted that the resolution of many distribution maps was unable to identify the specific microhabitats required. For animals, these can be influenced by numerous factors including refuge from predators (Glen et al. 2010), thermoregulatory requirements (Limberger et al. 1986; Parmenter et al. 1989; Kleckova et al. 2014), and suitability for camouflage (Lovell et al. 2013; Marshall et al. 2016). However, in some cases, the nature of the environment (such as dense jungle and deep ocean), or the shy or cryptic nature of the target species can make habitat assessments by direct observation difficult (Wrege et al. 2017). Such challenges call for innovative approaches, such as the use of acoustic monitoring (Lambert and McDonald 2014; Measey et al. 2017; Wrege et al. 2017; Picciulin et al. 2019), camera trapping (Carbone et al. 2001; Balme et al. 2009; Linkie et al. 2013), and tracking tunnels (Russell et al. 2009; Watts et al. 2013; Jarvie and Monks 2014). The choice of technology is species-specific, set by the limitations of the species' cryptic nature (Williams 2016).

The microhabitat requirements of many animals are relatively subtle, and relate to small differences in localised habitat structure (Valentine 2006; Hacking et al. 2014), especially for relatively sessile species (Hawkins et al. 2000; Mason et al. 2018). A novel option for assessment of localised habitat structure is LiDAR (light detection and ranging (Dassot et al. 2011), a non-destructive tool that rapidly and precisely digitises an object or site into a three-dimensional (3D) point cloud (Dassot et al. 2011; Weber 2018). LiDAR has been applied to the broad-scale assessment of numerous fauna habitats, including forests (Lim et al. 2003; Anderson and Burgin 2008; Michel et al. 2008), tidal flats (Hannam and Moskal 2015), subtidal coastal zones (Zavalas et al. 2014), and rivers (Laize et al. 2014; Mandlburger et al. 2015), but most of these have been at relatively large scales at square metre resolutions. At a smaller scale, terrestrial LiDAR allows for detailed scanning of microhabitat structure without obstruction from overhanging canopy or vegetation. We propose that the ultra-high-resolution (10 mm) precise characterisation of the physical environment provided by terrestrial LiDAR scanning provides a unique opportunity to gain an in-depth understanding of log pile microhabitat requirements for fauna of conservation concern.

Coarse woody debris, such as fallen log piles, are often critical habitat features for threatened fauna such as numbats (*Myrmecobius fasciatus*; Cooper and Withers 2005), chuditch (*Dasyurus geoffroii*; Orell and Morris 1994), and the Western Spiny-tailed Skink (*Egernia stokesii badia*; Pearson 2012). However, not all log pile sites are equally suitable for species habitation, and species-specific preferences for features such as log diameter, canopy cover and presence of adjacent trees can influence site suitability (McPeek et al. 1983). Here, we report the first study using ultra high-resolution terrestrial LiDAR to quantify the microhabitat characteristics of fallen log piles, using this approach to estimate the suitability of log piles for an endangered cryptic reptile subspecies (western spiny-tailed skinks). We aimed to determine if the skinks exhibited a high selectivity for certain structural features of log pile habitat, such as structural complexity with multiple logs providing potential refuge options, or associated features such as degree of vegetation cover (e.g., canopy gaps for basking) through comparison of log piles known to be inhabited and uninhabited by the species. Such detailed analysis of log pile characteristics and understanding skink microhabitat specificity will provide crucial information in the design of future habitat improvement for management efforts, highlighting the applicability of the technology for the assessment of other complex microhabitat structures, potentially including the specific structures of specialised roosting habitats in some species of bats (Armstrong 2001; Mancina et al. 2007) or nesting hollows or sites of endangered parrots (Webb et al. 2012; Watson et al. 2014) to better understand sites for protection, translocation, or replication in restoration and other threatened species management.

4.4 Methods

4.4.1 Study species

Western spiny-tailed skinks live together in social groups and these social aggregations occupy large fallen log piles (Pearson 2012), inhabiting the hollows and crevices in the wood. The skinks are at risk of extinction and are recognised under both Australian Federal legislation (Endangered; *Environment Protection and Biodiversity Conservation Act 1999*) and Western Australian state legislation (Schedule 1; *Biodiversity Conservation Act 2016*). One of the major threats identified for this subspecies is habitat loss and modification through mining and grazing; the translocation of specific populations threatened by local extinction is a recommended recovery option (Pearson 2012). Although there have been no successful translocations of this subspecies published in the scientific literature, there are anecdotal reports of failed attempts which may indicate that the skink has specific log pile requirements to ensure establishment and persistence.

Egernia s. badia are a large skink, with both sexes reaching a mature size over 170 mm snout-vent length (SVL; Duffield & Bull 1996). Females bear live young, with some *E. stokesii* subspecies producing litters of up to eight individuals at a time (Duffield & Bull 1996). The family groups are social and live together in groups comprised of different sexes and age classes. These social aggregations occupy the same large fallen log pile (Pearson 2012) over years, inhabiting the hollows and crevices in the wood. The juvenile skinks also take over five years to reach mature size, with many skinks remaining in the same social group as their parents even after reaching maturity (Duffield & Bull 2002). Beyond this basic information, the ecology of *E. s. badia* is largely unknown, compared to comparatively well-studied subspecies of *E. stokesii* occurring in states outside of Western Australia.

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4.4.2 Study area

The study area is located approximately 450 km northeast of Perth, Western Australia, on a mining tenement in the Mid West region (29°10'54"S, 116°32'55"E). The site occurs in a semi-arid region within the distribution of the skinks and comprises mainly open eucalypt woodland on loam or clayey loam flats, predominantly York Gum (*Eucalyptus loxophleba*) over a sparse understorey (Ecoscape 2016; Silver Lake Resources 2021; Figure 4.1A). Log piles inhabited by the skinks (determined from previous fauna surveys) were randomly selected for study (Figure 4.1B, C & D) although, due to site access limitations, log pile selection was restricted to within 1000 m of a 55 km access track (Figure 4.2).

Figure 4.1 Typical habitat of the skinks in the Mid West region of Western Australia; **(A)** open *Eucalyptus* woodland in which log piles were sparsely distributed; **(B & C)** examples of log piles inhabited by skink colonies; and **(D)** juvenile skinks basking by one of the hollows of an occupied log pile. Photos taken by Holly Bradley.

Figure 4.2. Distribution records of *Egernia stokesii badia* (green) according to records maintained by the Atlas of Living Australia (https://www.ala.org.au/, accessed 13 January 2022, OpenStreetMap contributors), and the location of the study site (black circle), with different LiDAR scanning locations (pink). Maps generated using QGIS 3.18.2 (QGIS.org, 2021. QGIS Geographic Information System. QGIS Association. http://www.qgis.org).

4.4.3 LiDAR technology

All research and animal observational experiments were carried out and approved by the Animal Ethics Office of Curtin University (ARE2018-28) and conformed with all relevant guidelines and regulations. Scans were taken during the end of the austral winter and beginning of spring, to capture the peak abundance of annual plants. A total of 39 log piles (22 inhabited and 17 uninhabited), was scanned and three-dimensional (3D) point clouds generated for quantification of the 14 structural characteristics. The laser scanning data for this investigation were collected using a terrestrial LiDAR scanner, the Maptek™ I-Site™ 8800 (Maptek, Adelaide, Australia), which has a resolution of 10 mm (McPeek et al. 1983) and a range of up to 2000 m (Maptek 2010). The LiDAR scanner was set up using a tripod on a standard tribrach mount, and a marked post was installed at each log pile site so that the scanner could be manually aligned to the top of the post using the survey alignment telescope for each scan. The scanner was placed in three to five positions around each log pile, depending on how large the log pile was, to create overlapping scans for development of a full 360-degree view of target log piles. Scan positions were targeted to ensure scanner positioning maximised capture of internal log structure within hollows. The LiDAR system position was coupled with a differential GPS system so that points were recorded with an xyz coordinate (SoilWater Group 2018).

4.4.4 Data processing and analysis

Consecutive scans of each log pile were merged into a single point cloud oriented using known GPS coordinates (Kemeny and Turner 2008). Point clouds were then processed using the I-Site Studio software package on Maptek v5 Point Studio. High-resolution digital images taken with each scan were ‘draped’ over each point cloud to produce a 3D digital terrain model (DTM) of each log pile scene (Kemeny and Turner 2008). Only landscape features within a 10 m radius of each log pile were included and the model then divided into three sections: above 2 m (canopy cover), between 30 cm and 2 m (mid-storey cover), and below 30 cm (understorey; Figure 4.2A & B).

Surface area along a single plane was calculated using a topographical model post-filtering for the canopy cover, mid-storey cover, and understorey layers. Manual filtering out of logs, trunks and branches prevented overestimation of vegetation cover. The processing software CloudCompare (version 2.12, 2021 retrieved from http://www.cloudcompare.org/) was used to develop system learning to isolate ‘bare ground’ versus ‘vegetation’ for understorey estimates.

Point clouds of each site were filtered to isolate each log pile system (Figure 4.2C), and the physical characteristics of each log pile were measured discretely: (i) maximum canopy height, (ii) number of logs, (iii) length of log system, (iv) number of branches above and below/adjacent to the main log, (v) log structure height, (vi) diameter of widest hollow, (vii) the presence of overhanging vegetation, (viii) the position of the log pile (majority resting on ground or raised), (ix) orientation of the log pile, and (x) the diameter of the widest section of log (Table S4.1).

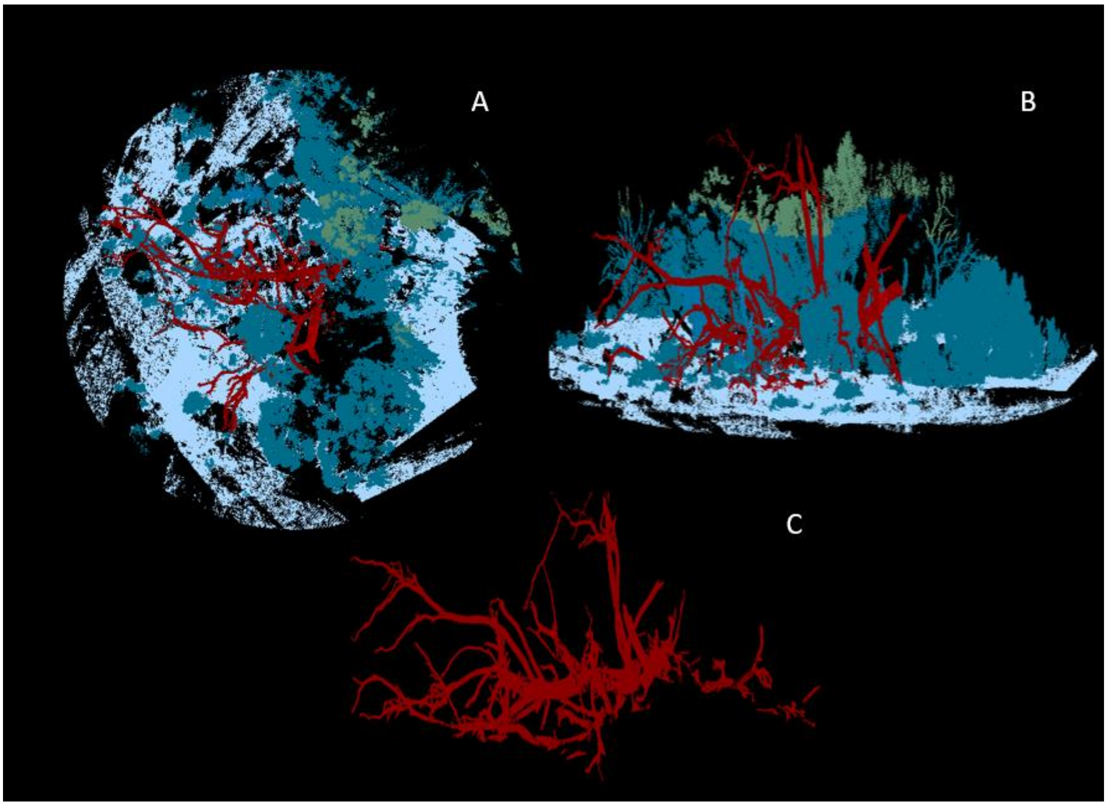

Figure 4.2 Example outputs of LiDAR scans; **(A & B)** the point cloud restricted within a ten-metre radius of the central log pile including the canopy cover (green), mid-storey cover (dark blue), and understorey (pale blue); and **(C)** the isolated log pile (red) from within the ten-metre radius point cloud.

Analyses were conducted in the *R 4.04* statistical environment (R Core Team 2016). To determine whether the presence or absence of skinks within log piles (binary dependent variable) was predicted by log pile characteristics, multiple logistic regression models (LRMs) with continuous predictor variables (number of high branches, number of low/adjacent branches, orientation, canopy cover, mid-storey cover, understorey, number of logs, length of log system, height of log system, canopy maximum height, diameter of the widest section of log, diameter of widest hollow entrance, presence of overhanging canopy, presence of logs raised above the ground) were fitted. Missing values were replaced with the global mean of each appropriate parameter. The “dredge” function from the “MuMIn” package (Bartoń 2020) was used to apply a drop one protocol, to retain a candidate set of LRMs with a lower AICc (Akaike’s Information Criterion corrected for small sample sizes) than the global model. From that candidate set of models we selected the models with the lowest AICc (≤Δ2 of the lowest AICc) and calculated the weight (ωi) of each model, which is the probability of that model being the best model. To assess the importance of individual variables we summed the weights of all models containing each variable and considered all variables with a summed model weight >0.4 to be well supported (Converse et al. 2006).

4.5 Results

Compared with uninhabited log piles, inhabited log piles were generally taller with smaller entrance hollows and a wider main log, had more high-hanging branches, fewer low-hanging branches, more mid- and understorey cover, lower maximum canopy height (Figure 4.3), most often faced in a SE direction (36%), and had some or most of the log pile raised off the ground (81%). The most parsimonious LRM indicated that log pile occupation by skinks was significantly predicted by increasing length of log piles ($\sum\omega_i = 0.9$), decreasing number of logs per pile ($\sum\omega_i = 0.70$), reduced canopy cover ($\sum\omega_i = 0.76$; Figure 4.3), and the presence of overhanging vegetation ($\sum\omega_i = 0.69$; Figure 4.4; Table S4.1).

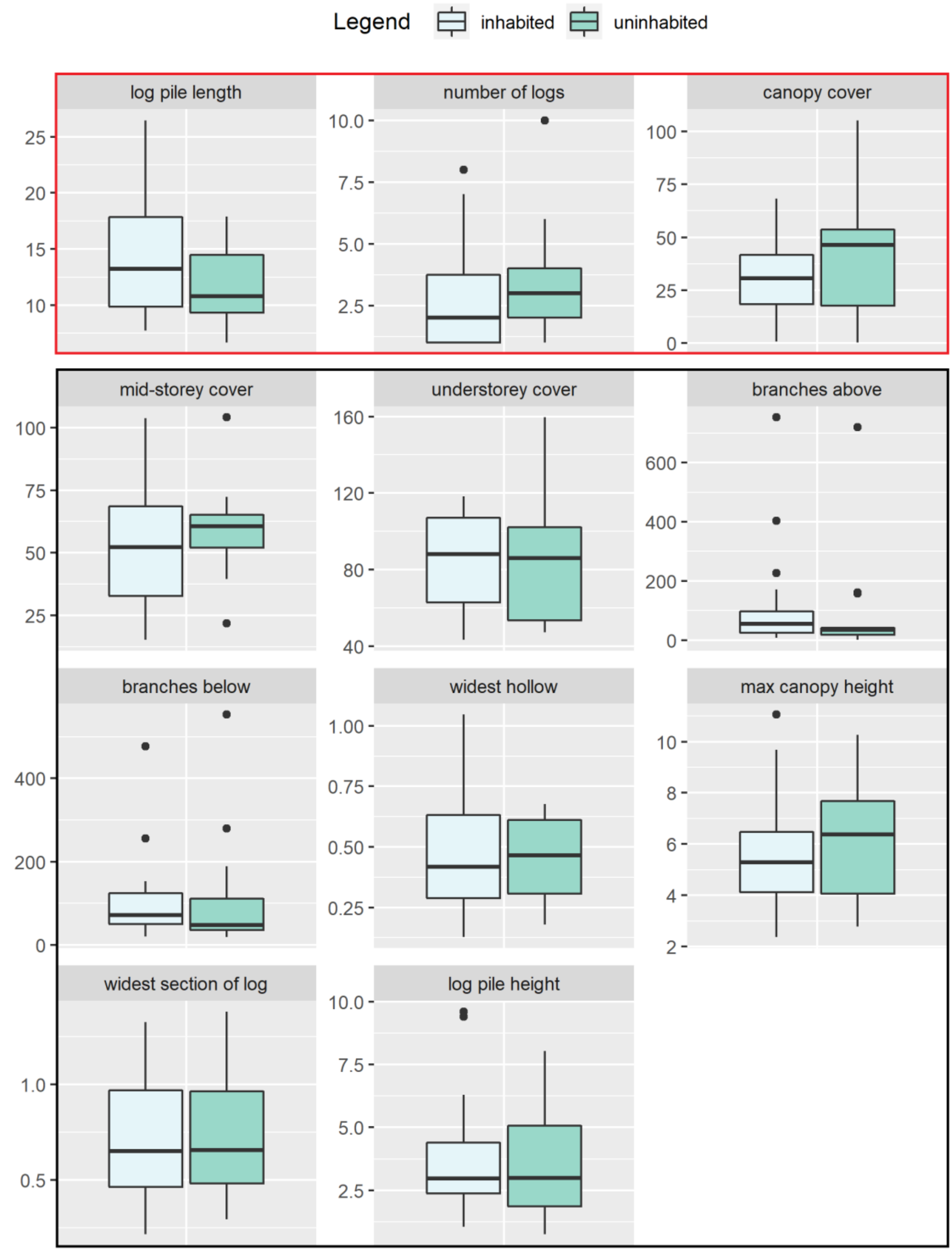

Figure 4.3 Boxplots showing the average log pile characteristics at both inhabited and uninhabited log piles. Thick horizontal lines indicate the median, boxes represent the 2nd and 3rd quartiles, and whiskers represent the 1st and 4th quartiles. Individual points represent outliers. Variables well supported to influence log pile occupancy (log pile length, number of logs and canopy cover) are bordered in red.

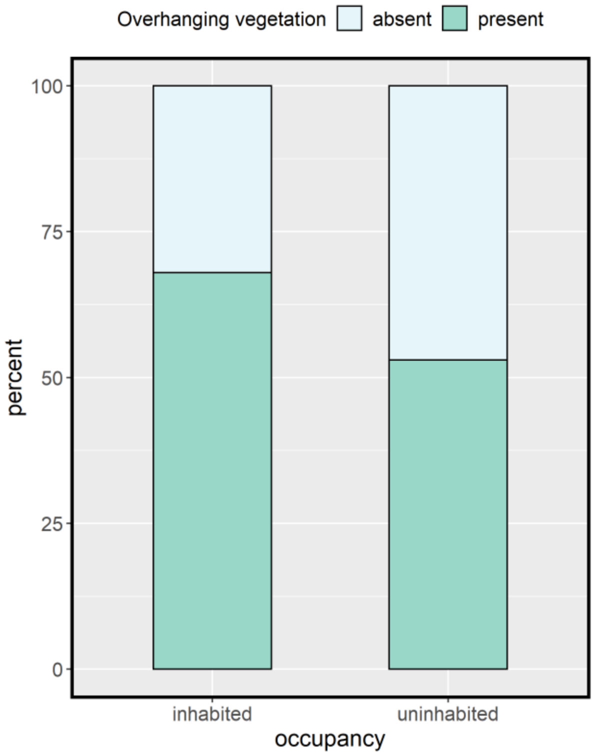

Figure 4.4 Stacked bar chart showing the differing percentage of inhabited and uninhabited log piles with vegetation overhanging the log pile (green) and with no vegetation overhanging the log pile (blue).

4.6 Discussion

4.6.1 Microhabitat selectivity

The novel use of LiDAR to study microhabitat provided a high accuracy and resolution of structural characteristics otherwise unattainable by traditional techniques and determined four log pile characteristics to significantly influence skink presence. The most significant variable was length of the log pile, with skinks more commonly occupying longer logs. One possible reason for this trend is that taller trees may be older (Vieira et al. 2003), and larger and older trees are more likely to have hollows usable by fauna (Whitford and Williams 2002; Salmona et al. 2018). Tall trees (and the log piles they become) are also more likely to house a mixture of different sized cavities and hollow branches, and greater refuge options for skinks (Lindenmayer et al. 1993; Craig et al. 2011). Therefore, before log decomposition can contribute to the production of further hollows, log piles from taller trees begin with more crevice/hollow options. Longer log piles are also likely to have the space for segregation between members of the skink colony. Due to the size variability between juveniles, adults, and gravid females, a range of crevice options is more likely to support an entire colony’s requirements (Schwarzkopf et al. 2010), as well as greater structural complexity to allow for social separation and predator refuge (Cooper Jr and Whiting 2000). Other reptiles also non-randomly select trees with more branches and hollows, which is predicted to provide increased opportunity for behavioural thermoregulation (Webb and Shine 1997; Fitzgerald et al. 2002).

Skink occupation was also found to be linked with log pile composition. However, while occupancy was significantly linked with log piles averaging three logs, this was also the average for unoccupied log piles. This is likely a result of how log piles form within the landscape. The open eucalypt woodland habitat contains sparsely distributed trees or stands of trees (Ecoscape 2016; Silver Lake Resources 2021) which form isolated ‘habitat islands’ (Cooper Jr and Whiting 2000) when single or a few trees fall and decay to become log piles. The ‘logs’ within these fallen log piles were defined as the major trunks off which branches emanate. As trees are often forked and have more than one trunk, three logs are likely the average available within the landscape, or are at least the average number to provide the structural complexity (through hollow options and associated increase in branches) to support a colony.

Skinks also generally occupied log piles with overhanging vegetation combined with reduced canopy cover, indicating that the presence of vegetation, particularly at mid-storey height, adjacent to and overhanging log piles is important. Microhabitat variability helps to facilitate behavioural thermoregulation of ectotherms, and vegetation cover at a site of long-term residence is likely to be particularly important in an arid environment where vegetation is highly scattered (Grimm-Seyfarth et al. 2017). Microhabitats that provide complex shading have been found to increase the activity budget of other arid-dwelling lizard species during hot weather, with vegetation also acting as a temperature buffer during cooler months (Grimm-Seyfarth et al. 2017). Presence of vegetation around log piles in arid habitats can also increase the abundance and richness of reptiles, probably due to a range of benefits including increased food availability, predator refuge, and options for behavioural thermoregulation (Melville and Schulte Ii 2001; Attum et al. 2006; Munguia-Vega et al. 2013). The effects of cover on predation are mixed: some taxa are more susceptible to predation in habitats with less vegetative cover (Pietrek et al. 2009). Similarly, many species also preferentially forage in areas of vegetation (Moreno et al. 1996; Tchabovsky et al. 2001; Pizzuto et al. 2007). However, other studies show that predation can increase if cover provides perches for ambush predators (Hawlena et al. 2010; Oversby et al. 2018), likely why skinks preferred less vegetation cover at canopy height. Therefore, selection of log piles with overhanging vegetation either benefits both thermoregulatory capacity and refuge from predation by skinks or is a trade-off between the two.

4.6.2 Management implications

Within Australia, many semi-arid and arid-dwelling lizard species are uncommon, with their distribution often correlated with habitat, microhabitat, or diet specificity (Pianka 2014). As inappropriate habitat selection is one of the major reasons that herpetofauna translocations often fail (Germano and Bishop 2009), we predicted that microhabitat structure may limit log pile suitability for skink colonies, contributing to their limited distribution within the landscape. Our results support a degree of microhabitat selectivity by skinks, with occupation linked to log pile length, number of logs, canopy cover, and overhanging vegetation. Log pile length and composition can be easily manipulated when selecting translocation sites or introducing coarse woody debris to restoration sites. However, selection of sites with reduced canopy cover, but overhanging mid-storey, may take longer to influence through management. Biomass and vegetation complexity at the understorey and mid-storey height can be significantly reduced by introduced grazers (Tsiouvaras et al. 1989; Tasker and Bradstock 2006), retaining the canopy layer which they cannot reach. As skink habitat both in our study, and regionally occurs in areas with a long history of pastoralism and landscape degradation from grazing and mining operations (Payne et al. 1998), restoration efforts and establishment of exclusion zones may be required to recover appropriate vegetation structure prior to any translocations into the area. In areas of mining restoration, while coarse woody debris can be introduced into the landscape, growth of surrounding vegetation cover may take time to establish (Lamb 2011; Shoo et al. 2016), leading to a lag-phase in the development of suitable habitat for fauna recolonisation or translocation. Pre-planning is, therefore, critical to ensure that recipient sites have suitable microhabitat characteristics to support skink colonies prior to any translocations taking place.

In addition to pre-planning and active management, we also recommend additional research be undertaken to further improve our knowledge of the ecological requirements of the Western Spiny-tailed Skink. Our observations of occupied colonies were restricted to a short window of time and our contemporary distinction between inhabited and uninhabited log piles may not be reflective of the sites most suitable to support colonies. Habitat degradation through grazing is a major threat for the skink (Pearson 2012), and the study area has a long history of pastoralism (Payne et al. 1998). Remnant skink colonies may, therefore, have been increasingly prevented from dispersing to other suitable uninhabited log piles by habitat degradation and fragmentation arising from grazing and mining infrastructure (Bowler and Benton 2005). Increased predation pressure from introduced pests such as Feral Cats (*Felis catus*) may also have impacted dispersal capability across the landscape, as has been observed for other species of *Egernia* in degraded or disturbed landscapes (Stow et al. 2001; Stow and Sunnucks 2004). Further research is recommended to understand if skinks have limited dispersal capacity within degraded landscapes, and if the non-dispersing older adults are remaining with younger adults and failing to establish new colonies. This research could also help determine if the trends in inhabited log pile characteristics observed in this study become more pronounced with a greater influence of habitat choice.

4.6.3 Conclusions

The degree to which we could obtain highly accurate and finely resolved measurements of inhabited and uninhabited microhabitats was essential to our capacity to differentiate between the two, and the application of LiDAR made this possible in a way that conventional measurements would not have. Overall, such detailed characterisation of microhabitat structure will provide important insight into the management of a cryptic, endangered skink, such as selecting appropriate sites for translocation. Guiding translocations is, however, only one aspect of wildlife management and species recovery to which an understanding of microhabitat preferences is central. We suggest that other applications might include designing restoration landscapes to facilitate skink colony return, increasing the targeted nature of monitoring surveys, highlighting key areas within their broader habitat range for protection, and indicating areas for targeted invasive predator control. The novel application of terrestrial LiDAR for microhabitat characterisation will be a cost-effective, accurate tool with far-reaching applications in the future study of ecological systems around the world, such as the assessment of other complex microhabitat structures (e.g., roosting structures for bat species including the Orange Leaf-nosed Bat (Armstrong 2001) or nesting sites of endangered parrots such as the Swift Parrot (Webb et al. 2012) to better understand sites for protection, translocation, or replication in restoration and other threatened species management.

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4.9 Supplementary material

Table S4.1. Results of the best subset modelling of the log pile structural data showing all well-supported models ($\omega_i > 0.1$) describing the relationship (whether +ve or –ve) between log pile occupancy and structural variables, plus ΔAIC_c, ω_i, and Nagelkerke's R^2 values of that model are shown. Nagelkerke's R^2 values represent the amount of variation explained by the model.

Structural variables	**ΔAIC_c**	**ω_i**	**Nagelkerke's R^2**
Canopy cover (-ve) Length of log system (+ve) Number of logs (-ve) Presence of overhanging vegetation (+ve)	48.43	0.18	0.469

Chapter 5. Diet of the Western Spiny-tailed Skink

(*Egernia stokesii badia*)

Photo: Holly Bradley

5.1 Abstract

A number of colonial lizards display an ontogenetic shift in dietary preferences, commonly from omnivory to herbivory. This pattern, as well as considerable dietary plasticity, has been observed in several *Egernia* species, and this may enable them to maximise the nutritional value of their diet and adapt to the seasonal availability of resources. *Egernia stokesii badia* is an endangered subspecies of *Egernia* whose diet in unknown. Due to the colonial nature of the subspecies and its occurrence in a semiarid environment, it was predicted that *E. s. badia* would exhibit the ontogenetic resource partitioning and opportunistic, highly varied diets observed in other species of *Egernia*. These predictions were tested by visual dissection and DNA metabarcoding of scats. The findings from these methods suggest that *E. s. badia* is a skink that is predominantly herbivorous, with no ontogenetic dietary shift in the proportion of invertebrates consumed; however, subadults likely directly target the consumption of some invertebrates to support rapid growth and development. While the skinks also consume plant matter from a diversity of families, they are not indiscriminate foragers, and preferentially forage on some specific resources.

5.2 Introduction

Ontogenetic shifts in diet have been observed for a number of omnivorous lizard species (Toyama et al. 2018), generally characterised by a shift from a greater proportion of invertebrates in the subadult diet to a greater proportion of plant material in the adult diet (Duffield and Bull 1998; Fialho et al. 2000; Durtsche 2004). The consumption of invertebrates can be particularly important for subadults, as invertebrates provide a more effective source of protein and nutrients than plant material, which is important for the construction of body tissues and to support rapid growth, which may reduce predation risk (Troyer 1984; Durtsche 2000). Once size at maturity has been reached, adults may rely more on herbivory due to the seasonal nature, unreliability, and scarcity of invertebrate prey items, particularly for species in arid environments (Pietczak and Vieira 2017). As most plant material is relatively undigestible and provides low energetic returns, herbivory may also be more effective for adults with a larger body size and bigger gut volume to increase food fermentation and assimilation time, maximising digestive efficiency (Guard 1980; Bjorndal 1985; Mackie et al. 2004). However, such a diet may only be sustainable in warm environments, due to the high body temperature likely required for the microbial fermentation of plant tissues and the release of energy stored in plant cells (Troyer 1987; McNab 2002).

Such a pattern of herbivory linked with body size is apparent in the skink genus *Egernia*, where the proportion of plant matter in the diet increases with body size for nine species (Chapple 2003). Smaller species, such as *E. striata* (105mm snout-vent length (SVL); 4.6% plant diet) and *E. whitii* (90mm SVL; 8.4% plant diet) feed mostly on invertebrates (Chapple 2003). The largest *Egernia* species (>170mm SVL) have the greatest propensity towards herbivory, with some species being almost entirely herbivorous: *E. kingii* (88% plant diet) (Arena and Wooller 2003), *E. cunninghami* (92.8% plant diet) (Brown 1991), and *E. stokesii* (97% plant diet) (Duffield and Bull 1998). These three large *Egernia* species also have ontogenetic shifts in their dietary preferences, changing from omnivores as subadults to predominantly herbivores as adults (Chapple 2003). By exhibiting this degree of dietary plasticity, skinks are likely to maximise the nutritional value of their diet and adapt to the seasonal availability of invertebrates and plants (Chapple 2003).

Egernia stokesii, one of the larger *Egernia* species (average SVL of 170mm; Chapple 2003), has a range across western, central, and eastern Australia (Pearson 2012), and studies have found an ontogenetic dietary shift from a greater proportion of invertebrates as subadults to a largely herbivorous diet as adults (Duffield and Bull 1998). However, these studies were conducted on *E. s. zellingi* in South Australia, and there is little information on the diet of other subspecies from Western Australia (Pearson 2012). There have been conflicting reports, such as Storr (1978) suggesting the skinks are “partly vegetarian” in Western Australia, and anecdotal reports of coastal Western Australian populations having diets of both plant and arthropod material (Pearson 2012). The aim of this investigation was to assess whether one western subspecies, the endangered *E. s. badia,* has the same ontogenetic shift in dietary preference from invertebrates as subadults to a predominantly herbivorous diet as adults, as has been observed for *E. s. zellingi* in South Australia (Duffield and Bull 1998). As *E. s. badia* is a colony-living skink which lives in an arid Western Australian landscape dominated by “boom and bust” cycles of resource availability (Arthington and Balcombe 2011; Pearson 2012), it was predicted that *E. s. badia* would also exhibit ontogenetic resource partitioning, as well as opportunistic, highly varied diets depending on resource availability. To test these predictions, both visual identification and DNA metabarcoding analysis were used to determine the plant and invertebrate content of scats across different colony locations in the Mid West region of Western Australia and compared diet composition between subadult and adult *E. s. badia,* both within and across different colonies.

5.3 Methods

5.3.1 Sample collection

Scats were collected from colony latrine piles (Figure 5.1) located on mining tenements in the semi-arid Mid West region of Western Australia, approximately 450km northeast of Perth (29°10'54"S, 116°32'55"E). Not all scats were removed from each log pile, to ensure olfactory cues persisted at the site, in case they played an important role in territorial marking. A total of 30 scats (14 adults and 16 subadult) was collected from five active colony sites in August 2018 and used for visual inspection of the skinks' diet. The length and width of each scat was recorded prior to dissection for dietary analysis. Subadult scats were smaller than adult scats, being >27mm in length (assessed through distribution of scat dimensions).

A total of 36 scats (18 adult and 18 subadult) was collected from nine different log pile sites in September 2019 for genetic analysis of the lizards' diet. Known active log piles were visited every day to every two days (due to an inability to visit all sites in a single day) for two weeks, and any fresh scats identified within that time were collected. Photographic evidence, colouration, and other cues (such as visible moisture and presence of flies) were used to identify fresh scats. Each scat was placed in a separate 10mL vial, pre-filled with silica beads (1/3 of the container) and separated from the scat with a cotton wool ball. Vials were kept in a dark box to prevent DNA degradation from sunlight. Methods for scat preservation follow the recommendations of Pearson et al. (2015), who suggested drying to be the most appropriate preservation method when sampling in semi-arid and arid locations away from electricity supplies. Each scat was then subsampled (approximately 100mg), and samples were transported to the laboratory where they were stored at -20°C until scheduled for DNA extraction.

Figure 5.1 Example images of **(A)** an adult *E. s. badia* individual, **(B)** varying sizes of *E. s. badia* individuals cohabiting the same log pile, **(C)** a log pile surrounded by flowering annuals, and (d) a colony latrine pile located adjacent to their log pile residence. Images taken by Holly Bradley.

5.3.2 Visual identification of invertebrates

Each scat from August 2018 was placed in a petri dish with water and gently teased apart with forceps. Using a dissecting microscope, each scat was divided into invertebrate and other (mostly plant) material and placed in separate petri dishes to air dry, and dry weights were recorded. Invertebrate taxa were identified to order, the most specific classification possible through visual means. The number of individuals of each different invertebrate order was recorded for each scat sample.

5.3.3 Genetic analysis of plant and invertebrate scat composition

5.3.3.1 DNA extraction

DNA was extracted in September 2021 from each September 2019 scat using a Qiagen PowerFecal Pro kit (Qiagen). First, sub-samples of each scat were taken, with 40-280mg (average 82mg) of material transferred to a sterile PowerBead tube. Four glass beads (3mm) were added to all samples as well as two extraction controls, followed by 800µL CD1 buffer. DNA extraction then proceeded as per the manufacturer's protocol using a QIAcube extraction platform (Qiagen), with ~100uL of the extracted DNA transferred to 1.5mL low bind tubes and stored at -20°C.

5.3.3.2 DNA metabarcoding and sequencing

An initial polymerase chain reaction (PCR) was performed for each sample. PCR master mixes comprised of: 2mM $MgCl_2$ (Applied Biosystems), 10× PCR Gold buffer (Applied Biosystems), 0.25mM dNTPs (Astral Scientific, Australia), 0.4mg/ml bovine serum albumin (Fisher Biotec), 0.4µmol/L forward and reverse primers (Table 5.1), 0.12× SYBR® Green (Life Technologies), 1U AmpliTaq Gold DNA polymerase (Applied Biosystems), and 2µL template DNA. All PCRs had a final volume of 25µL and were performed using StepOne Plus Instruments (Applied Biosystems), with the extraction, positive, and negative controls included in each PCR run at the neat concentration. PCR cycling conditions consisted of denaturation at 95°C for 5 minutes, followed by 50 cycles of: 95°C for 30sec, 46°C (COI) or 52°C (trnL) for 30sec, 72°C for 45sec, finishing with a final extension stage at 72°C for 10 minutes.

Once the appropriate dilution was determined for each sample, fusion tag primers containing a unique 8bp multiplex identifier (MID) tag, the gene-specific primer, and Illumina's sequencing adaptors were used in a PCR reaction with the same conditions as described above. Sequencing libraries were then created from these pools by combining them in approximate equimolar ratios based on the quantification.

Table 5.1 Primers for the COI and trnL assays used in this study.

Assay	Primer name	Primer sequence (5'-3')	Reference
COI	mLCOIintF	GGWACWGGWTGAACWGTWTAYCCYCC	Leray et al. (2013)
	jgHCO2198	TAIACYTCIGGRTGICCRAARAAYCA	Geller et al. (2013)
trnL	trnLg	GGGCAATCCTGAGCCAA	Taberlet et al. (2007)
	trnLh	CCATTGAGTCTCTGCACCTATC	Taberlet et al. (2007)

5.3.3.3 Bioinformatics and taxonomy assignments

The bioinformatic pipeline eDNAFlow (Mousavi-Derazmahalleh et al. 2021) was used to analyse raw sequence data generated from the metabarcoding, with all operations performed on a high-performance computing cluster (Pawsey Supercomputing Centre). The sequencing results were demultiplexed and assigned to their sample of origin based on their unique indices, trimmed using Obitools, and quality filtered with Usearch v11 for sequencing errors (maxee=1) with a minimum length of 150 (COI) or 30 (trnL). Sequences were then dereplicated and unique sequences were transformed into zero radius operational taxonomic units (ZOTUs) to provide sensitive taxonomic resolution (Usearch v11) (Edgar 2018). Generated ZOTUs were queried against the nucleotide database NCBI (GenBank) and the PilbSeq database (trnL data only) (Nevill et al. 2020) and assigned to the species level where possible. Taxonomic assignments were based on an in-house Python script which further filtered the Blast results (evalue ≤1e-5, %identity ≥95, qCov=100, diff=0.5), combined them with the ZOTU table results and produced a table containing the taxonomic information available from Blast taxonomy database (accessed October 2021). Based on the distribution of sequence abundance for each assay, results were further filtered by removing sequence reads ≤5 (COI) or ≤9 (trnL). Samples were examined for potential contamination by checking the extraction and negative controls, and the sequencing success was ratified through the positive controls. Where a species-level taxonomic assignment was made, the sequence similarity was checked; where the match was <97%, the assignment was dropped back to the genus-level. Taxonomic nomenclature was validated using the Global Biodiversity Information Facility (2021), with the final taxa list converted into a presence/absence matrix for each assay.

5.3.4 Data analysis

5.3.4.1 Proportion of invertebrates (combined) within the diet

Analyses were conducted in the *R 4.10* statistical environment (R Core Team 2017). To determine whether the proportion of invertebrates in August 2018 scat samples varied according to age class, a Wilcoxon rank-sum test was conducted, with age class (subadult and adult) as the independent variable and the dry weight proportion of invertebrates as the dependent variable. The same test was used to determine if the proportion of invertebrate OTU detections from September 2019 scats differed according to age class, with age class as the independent variable and the OTU proportion of invertebrates (measured as the proportion of invertebrate OTU detections out of the total OTU detections per scat) as the dependent variable.

To determine whether the proportion of invertebrates in scat samples varied according to site, a Kruskal-Wallis rank sum test was conducted, with site (four sites) as the independent variable, and dry weight proportion of invertebrate as the dependent variable. The same test was used to determine whether the proportion of invertebrate OTU detections differed according to site, with site as the independent variable, and OTU proportion of invertebrates as the dependent variable.

5.3.4.2 Invertebrate families

A 6 x 1 Chi Square test was conducted to determine whether age classes contained an equal number of invertebrates from the seven families detected using DNA metabarcoding. Standardised residuals of chi square values greater than two were considered significant. This test was not repeated to determine whether sites contained an equal number of invertebrates due to the frequency of zero values in the data. To determine if the presence or absence of invertebrate families in scat samples varied according to site or age class, a Euclidean dissimilarity matrix was first constructed on the presence/absence data and then an ANOSIM (analysis of similarity) on that matrix was conducted, with age class and site (sites 1-10) as the independent variables. The number of OTUs per family was not compared as each scat sample did not contain more than one OTU per family. However, invertebrate richness, quantified as the number of invertebrate families present within each scat, was compared between age classes using a Wilcoxon rank-sum test with age class as the independent variable, and invertebrate richness as the dependent variable. To assess whether invertebrate richness varied according to site, a Kruskal-Wallis rank sum test was conducted, with site (sites 1-10) as the independent variable and invertebrate richness as the dependent variable.

5.3.4.3 Invertebrate orders

To determine whether the invertebrate abundance (number of individuals for each invertebrate order per scat) identified visually varied according to age class or site, we constructed an Euclidean dissimilarity matrix on invertebrate abundance data and then conducted an ANOSIM test on that matrix with age class and site (four sites) as the independent variables. To determine if there was a difference between orders in the number of individuals detected per scat (regardless of site or age class), a Kruskal-Wallis rank sum test was conducted, with order as the independent variable and number of individuals as the dependent variable.

5.3.4.4 Plant presence and richness

To determine whether the presence of plant families in scat samples identified from DNA metabarcoding varied according to age class or site, a Euclidean dissimilarity matrix was constructed, then an ANOSIM test was conducted on the matrix, with age class and site (sites 1-10) as the independent variables. To determine if plant OTU richness (number of OTUs observed for each plant family in each scat sample) from DNA metabarcoding varied according to age class or site, we first constructed an Euclidean dissimilarity matrix on the number of OTUs in each family and then ran an ANOSIM test on that matrix with age class and site (sites 1-10) as the independent variables. To determine whether the total families (total number of plant families observed in each scat sample) varied according to site (sites 1-10), a Kruskal-Wallis rank sum test was conducted, with site as the independent variable. A Wilcoxon rank sum test was used to test whether the total families varied according to age class, with age class as the independent variable.

5.3.4.5 Comparison with surrounding vegetation

To compare the proportion of plant species per family detected in surrounding vegetation to the proportion of plant families detected in skink diets, species composition data from vegetation surveys conducted by Woodman Environmental Consulting (2012) was used, at the same mine site in the Mid West region of Western Australia. Woodman Environmental Consulting (2012) recorded the presence of all vascular plant species in 187 quadrats measuring 20m x 20m in size, during different sampling periods from 2008 to 2011. Only two survey quadrats were within 500m of the skink colonies sampled for this study, so the vegetation composition data from only these two quadrat sites were pooled for the analyses. A paired samples Wilcoxon test was used to compare the familial proportions of species in the quadrats (number of species found per family), with the proportion of plant families found in skink scats.

5.4 Results

5.4.1 Proportion of invertebrates (combined) within the diet

There was no significant difference between age classes in either the dry weight proportion of invertebrates (*W* = 82.00, *p* = 0.939; Figure 5.2), with invertebrates comprising approximately 9% of the diet, or the OTU proportion of invertebrates (*W*=132.50, *p* = 0.261). Additionally, there was no significant difference between sites in either the dry weight (χ^2_3 = 4.18, *p* = 0.243) or OTU (χ^2_9 = 4.88; *p* = 0.845) proportions of invertebrates.

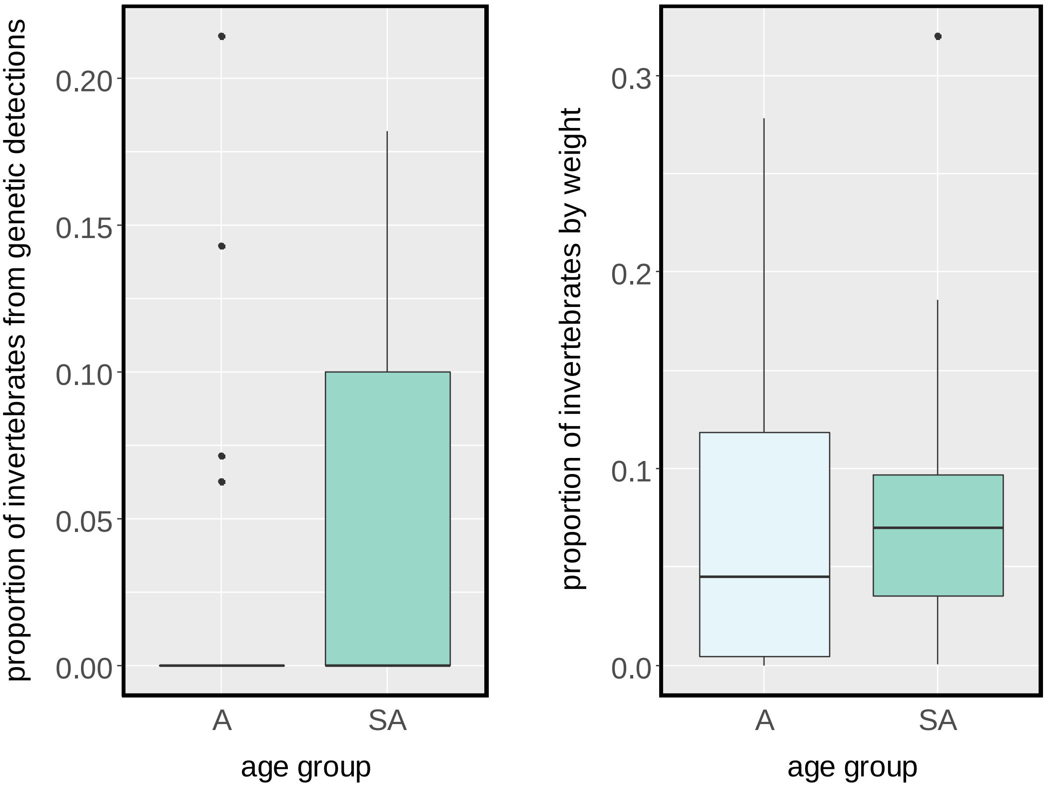

Figure 5.2 The proportion of invertebrates in adult (A) and subadult (SA) scat samples, according to both DNA metabarcoding (left) and dry weights (right). Notably, the genetic techniques failed to identify a high proportion of invertebrate prey in the adult scat (mean = 0+/-0), with the exception of

some outliers, but visual inspection identified a higher proportion of invertebrate prey by weight in both adult and subadult scat. Both techniques, however, showed there were no ontogenetic differences in the presence of invertebrates in the scat samples.

5.4.2 Invertebrate families

DNA metabarcoding analysis revealed a number of invertebrate families present in scats during September (Figure 5.3). Age classes contained a significantly different number of invertebrates from the seven families ($\chi^2_6 = 14.00$, $p = 0.030$), with subadults consuming more cicadellids than other families compared to the relatively equal proportions of invertebrate families discovered in the adult diet (Figure 5.4). The presence or absence of invertebrate families in scat samples did not vary according to site ($R = -0.08$, $p = 0.797$) or age class ($R = -0.03$, $p = 1.00$). Invertebrate richness also did not differ significantly between age classes ($W = 210.00$, $p = 0.062$) or sites ($\chi^2_9 = 6.16$, $p = 0.724$).

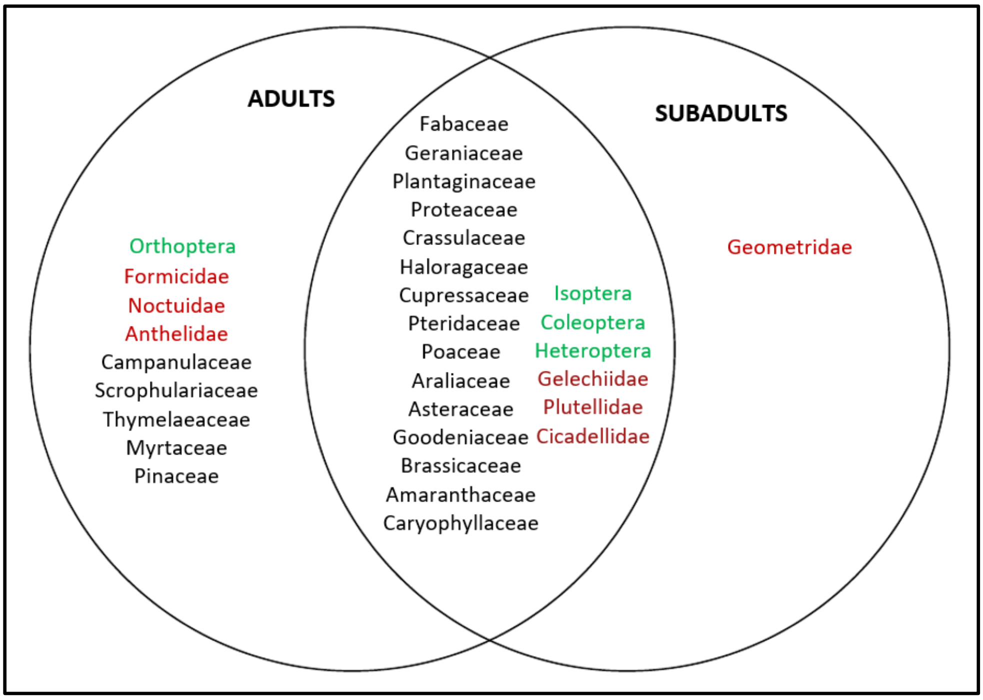

Figure 5.3 Venn diagram of the plant families (black) and invertebrate families (red) identified in the adult and subadult diet of *E. s. badia* using DNA metabarcoding. Invertebrate orders identified using visual identification have also been added in green.

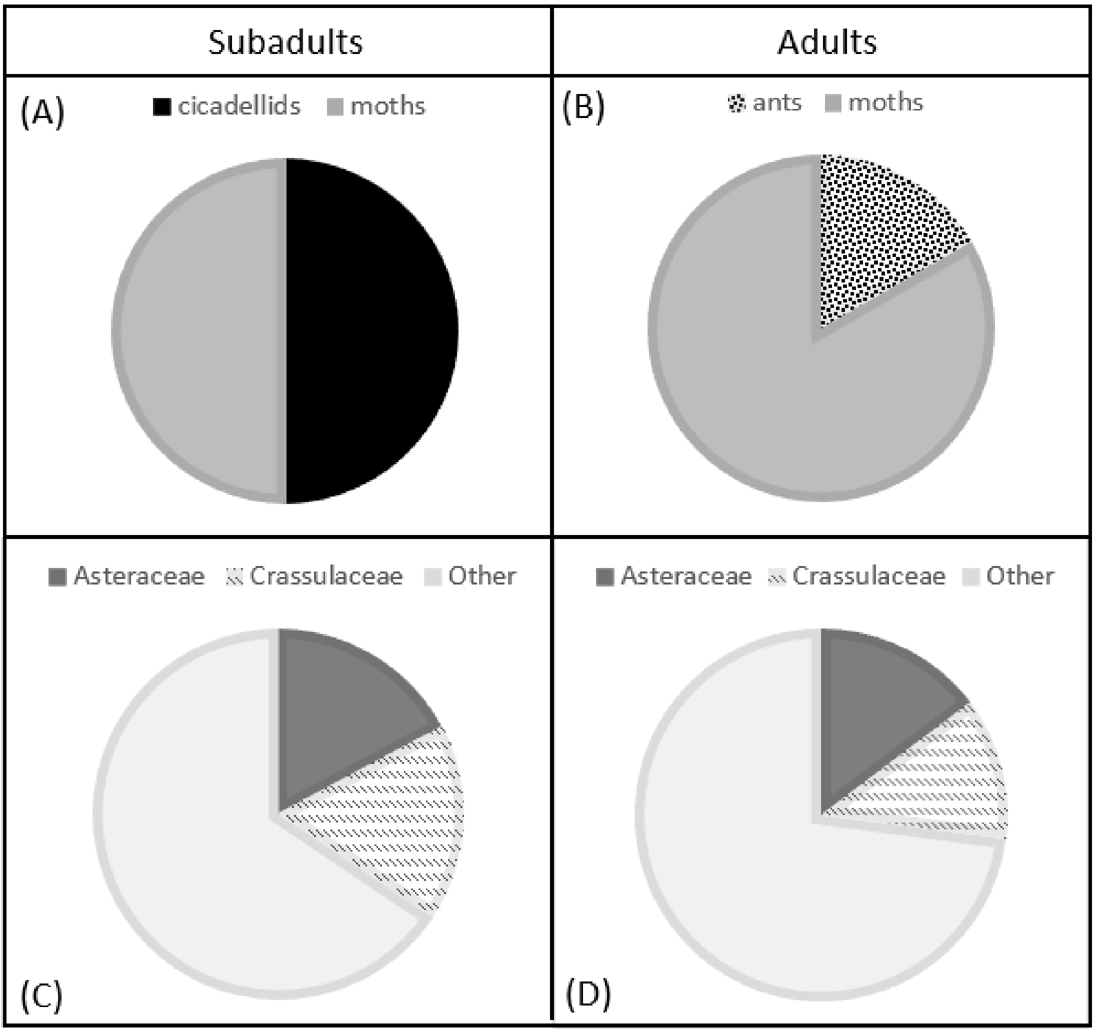

Figure 5.4 Pie charts highlighting the proportion of invertebrate groups in subadult **(A)** and adult **(B)** scat samples, and the most abundant plant families in subadult **(C)** and adult **(D)** scat samples, as identified through DNA metabarcoding.

5.4.3 Invertebrate orders

Based on the visual inspection of scats, the diet was largely comprised of the insect orders Isoptera and Coleoptera (Figure 5.5). There was no significant difference in invertebrate abundance between sites ($Pseudo\text{-}F_{3,19} = 0.50$, $p = 0.652$) or age classes ($Pseudo\text{-}F_{1,19} = 0.39$, $p = 0.503$). However, there was a significant difference in the abundance of orders ($\chi^2_4 = 76.05$, $p = <0.001$; Figure 5.5), with more Isoptera than any other orders, and more Coleoptera than Hemiptera, Araneae, and Orthoptera (Table S5.1).

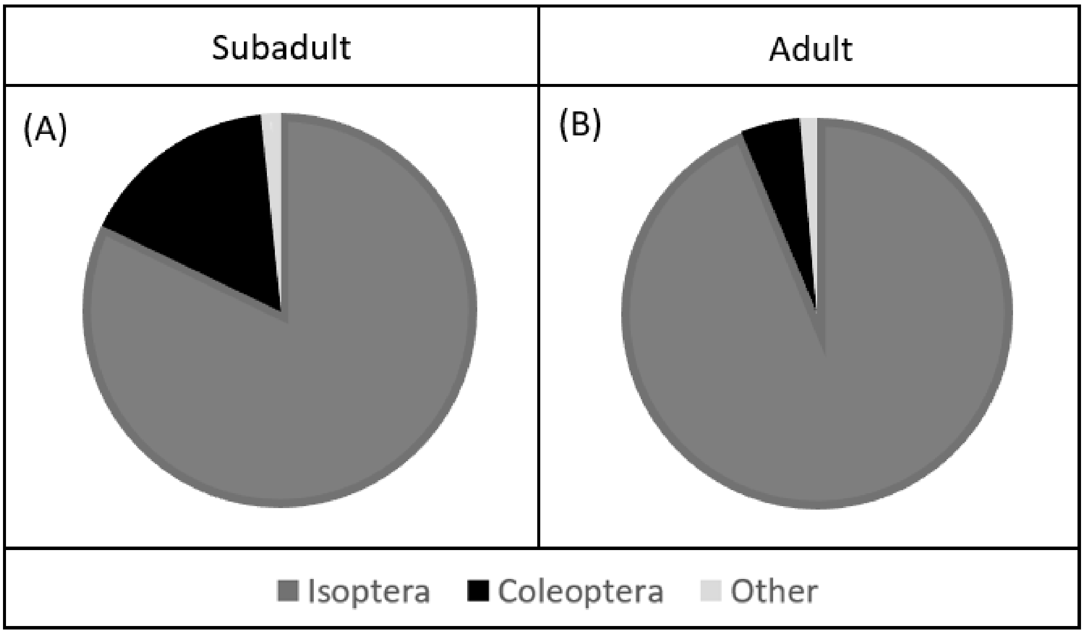

Figure 5.5 The proportion (calculated from the minimum number of individuals) of invertebrates from the different identified orders within **(A)** subadult and **(B)** adult scats collected in August 2018. 'Other' includes invertebrates belonging to Hemiptera, Araneae and Orthoptera.

5.4.4 Plant presence and richness

The presence of plant families in scats did not differ between sites (R = -0.48, p = 0.360) or age classes (R = 0.04, p = 0.426); Figure 5.4). Plant OTU richness per scat also did not differ among sites (R = 0.08, p = 0.254) or age class (R = -0.07, p = 0.667). The total families did not vary according to site (χ^2_9 = 11.60, p = 0.237), or age class (W = 202.5, p = 0.202).

5.4.5 Types of invertebrates and plants consumed

According to the DNA metabarcoding analysis, *E. s. badia* largely fed on moths in September, likely at the larval or pupal stage (Table S5.2). Five moth families containing six OTUs, Formicidae (ants) with one OTU, and Cicadellidae (leafhoppers) with three OTUs were identified according to DNA metabarcoding. The skinks also fed on a diversity of plant material, likely relying on a range of resource types from plants including flowers, buds, leaves and seeds (Table S5.3). Overall, the proportion of plant species per family in scats and survey quadrats did not differ significantly (V = 238.00, p = 0.664, Figure S5.1).

5.5 Discussion

The complementary diet analysis using visual identification and DNA metabarcoding successfully identified both invertebrates and plants in the diet of *E. s. badia*. Generally, 9% of the diet by dry weight was composed of invertebrates, with the rest mostly plant material, and there was no difference in the proportions between adult and subadult skinks. There was general agreement in this finding between both genomic and visual techniques, but the genomic techniques were able to identify dietary items to the family, and sometimes species, level, with seven families of invertebrates and 19 families of plants identified in the diet. This suggests that *E. s. badia* is a generalist herbivore that does not partition its foraging niche ontogenetically. They are not, however, indiscriminate foragers, and preferentially forage on some specific resources.

5.5.1 Invertebrate diet

Previous research on *E. stokesii zellingi* found an ontogenetic shift in diet, with a greater proportion of the subadult diet containing insects (Duffield and Bull 1998). While no significant difference was found in the proportion of invertebrates in the adult and subadult diet (either by OTU detections or weight) of *E. s. badia,* it was found that subadults consumed more Cicadellidae (leafhoppers) than adults. Cicadellids were also consumed in greater proportion than any other invertebrate family in the subadult diet. Subadult skinks also fed on geometrid moths which were not found in the adult diet.

Peak activity and abundance of some *Orosius* spp., the leafhopper genus identified in the diet of subadults, occurs in the austral spring, around mid-September (Trębicki et al. 2010). Geometrid moth larvae are also extremely active after the first significant rains of the year (Common 1990). As these two invertebrate groups were likely both highly abundant and accessible during the time when scats were collected for DNA metabarcoding analysis, and subadult skinks consumed significantly more of them than adult skinks, this supports the prediction that subadult skinks are more likely to directly target invertebrate prey than adults. The direct targeting of invertebrates by subadults is likely to support their fast growth rate (Duffield and Bull 1998).

The majority (71%) of invertebrate families identified using DNA metabarcoding were moths (Table S5.4). For adult skinks, 83% of the invertebrate families identified using

DNA metabarcoding were moths. Due to the concealed nature and small number of moths and ants consumed by adult skinks, these invertebrates present in the adult diet were likely the result of opportunistic or indirect consumption when foraging for vegetative matter. For example, Gelechiidae moth larvae are concealed in fallen *Acacia* seed pods, and ants have an unfavourable predator:prey size ratio, lack of palatability, prevalence of unfavourable defence mechanisms such as chemicals and stings (Abensperg-Traun and Steven 1997), and have a substantial cost in salt load and other toxins (Withers and Dickman 1995). Therefore, while there was no evidence of an ontogenetic shift in the volume of invertebrate matter in the skink diet, there was some evidence of a shift from subadults directly targeting invertebrate prey towards more opportunistic feeding by adults.

Visual identification revealed some invertebrate orders that were not identified using DNA metabarcoding. These were largely hard-bodied or hard-mandibled invertebrate orders, Isoptera (termites) and Coleoptera (beetles), followed by Hemiptera (true bugs), Araneae (spiders) and Orthoptera (e.g. grasshoppers). There was a greater number of Isoptera and Coleoptera in scats than from any other order. Although there was also a greater number of Isoptera than Coleoptera in scats, this likely accounts for the much larger volume a single beetle provides, compared to the smaller-bodied termites. As the skinks occupy log piles, the same microhabitat of termites, this likely explains why termites were prevalent in the skink diet. Subadults had a greater proportion of beetles present in scats compared to adults, potentially indicating that they are more likely to venture away from the log pile to hunt for these larger, high-reward prey items than adults.

Scats were collected in September for DNA metabarcoding analysis and in August for visual identification. The identification of different invertebrate groups supports the prediction that skinks have dietary plasticity and vary their diet according to the seasonal availability of resources, which can greatly fluctuate across arid landscapes (Puckridge et al. 2000; Jenkins et al. 2005; Arthington and Balcombe 2011). Skinks may have to rely on the consumption of termites, living in the same log pile habitat, to supplement their predominantly herbivorous diet until August, when some invertebrate groups, such as moths, are at their peak abundance, due to the dispersal and peak activity of larvae (Common 1990; Farias et al. 2020). Other larger invertebrates such as centipedes, scorpions, and spiders that tend to hit peak activity in the summer (Sutherland and Trinca 1978; Polis et al. 1981; Jelinek 1997; Balit et al. 2004), were also largely absent from the skink diet. These invertebrates may be absent from scat samples due to a lack of activity during the time of collection but may also indicate the propensity for *E. s. badia* to target less dangerous, more accessible prey items that do not require active hunting at night and provide less of an energy cost to consume.

The differences revealed by the two methods of taxonomic identification highlight the importance of using complementary techniques to gain an in-depth understanding of diet. Visual identification was able to detect the harder-bodied invertebrate parts which maintained form post-digestion, whereas softer-bodied invertebrates such as moths, while identified as a critical resource using the DNA metabarcoding analysis, were not identified during visual identification. Overall, the combination of both techniques has allowed us to determine a range of invertebrates present within the *E. s. badia* diet.

5.5.2 Plant diet

There was no significant shift in the volume or richness of plants according to skink age class or site. This suggests that both adult and subadult skinks likely have a largely opportunistic diet, reflecting the commonly available plant resources in the surrounding landscape. An opportunistic diet supports the prediction that to persist in an arid environment characterised by the patchy distribution of resources in both time and space, skinks will require a level of dietary plasticity to meet their energetic requirements (Chapple 2003).

More plant matter than invertebrate matter was consumed by both subadult and adult skinks, following the pattern of a largely herbivorous diet reported for other large species of *Egernia* (Chapple 2003). The highest diversity and greatest representation of a plant family in scat samples was the Asteraceae. Asteraceae species in the study area are predominantly small, annual flowering plants, with soft leaves low to the ground, and fairly large flowers (Western Australian Herbarium 2021). Many of these annuals appear in high abundance and high density in the austral spring in the Mid West open eucalypt woodlands, at the same time the scat samples were collected (Table S5.3). The high abundance of Asteraceae in the diet of both adults and subadults, therefore, likely reflects the easy accessibility and high abundance of the resource at that time.

One exception to what appears to be a largely opportunistic diet, was the abundance of Crassulaceae in the diet, with all material from the genus *Crassula*. In contrast to the highly abundant Asteraceae, *Crassula* spp. have a narrower niche breadth, often occurring in moist, shaded areas (Green 1947; Sheppard et al. 2006). *Crassula* spp. were not recorded in the vegetation survey quadrats near to the occupied logs (Woodman Environmental Consulting 2012), and the presence of Crassulaceae in scats was not linked to colony site location. Therefore, the presence of Crassulaceae in the diet might indicate they are a targeted food resource for both adult and subadult skinks. *Crassula* spp. often have fleshy, succulent leaves (Western Australian Herbarium 2021), and are, therefore, potentially valuable for both their nutrition and water content. *Crassula* spp. have been considered a likely important water source for other reptiles in arid habitats (Loehr 2002), where maintaining water balance and minimising water loss are key physiological issues for fauna to overcome (Ezcurra 2006). A high diversity of other plant families was also consumed by skinks, largely soft leaves and large flowers or flowers aggregated in inflorescences (Table S5.3), as an easily accessible and high reward-for-energy food resource. One exception was *Acacia* where, due to the height and prickly foliage of plants, it is likely that skinks fed on fallen seeds, which are encircled by elaborate arils containing a high lipid content, and a high 'profitability' due to the high proportion of digestible pulp compared to 'cost' in seed bulk (O'Dowd and Gill 1986).

5.5.3 Conclusions

The findings of this investigation suggest that *E. s. badia* is predominantly herbivorous, with consumption of invertebrates to supplement their diet, and some targeted hunting of high-reward energy prey by subadults likely to support rapid growth and development. While the skinks consume plant matter from a diversity of families, they are not indiscriminate foragers, as some plant families present in their scats were not recorded in the nearby vegetation surveys. This suggests that they are preferentially foraging on some specific resources, such as *Crassula,* to meet their nutritional or water requirements. However, caution must be taken to interpret the results of DNA metabarcoding studies, as there is potential contamination from the plant material in the prey diet.

5.6. Acknowledgements

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5.8 Supplementary material

Table S5.1 Significance values highlighting the differences in the number of individuals present per invertebrate order within scats, through visual identification.

Order with a greater minimum number of individuals	Order with a lesser minimum number of individuals	*Z* value	*p* value
Isoptera	Araneae	-6.84	<0.001
Isoptera	Coleoptera	-2.67	0.0075
Isoptera	Hemiptera	-6.37	<0.001
Isoptera	Orthoptera	6.84	<0.001
Coleoptera	Araneae	-4.17	<0.001
Coleoptera	Hemiptera	3.70	<0.001
Coleoptera	Orthoptera	4.17	<0.001

Table S5.2 The likely mode of consumption (indirect, direct, or opportunistic) and life stage (larvae, pupae, or adult) of prey moth species consumed by *E. s. badia*, with supportive reasoning.

Family	**Genus**	**Likely mode of consumption**	**Life stage likely consumed**	**Reasoning**	**References**
Anthelidae	*Anthela*	indirect or opportunistic	larvae	Adults of Anthelidae are almost always nocturnal. Larvae also often feed exposed during the night (often on grasses, *Acacias*, or eucalypts), and may seek shelter during the day. Consumption of larvae by skinks likely occurs when feeding on host plants. Cocoons are often found on the trunk of the food tree and may be less likely to be found by a skink.	Zborowski and Edwards (2007)
Gelechiidae	*Ephysteris*	indirect or opportunistic	larvae	Adults often conceal themselves in the crevices of bark on trees or among leaf litter during the day. Larvae often spend a large portion of their life in shelters formed by joining adjacent leaves with silk. As both life stages are well concealed, skinks likely indirectly feed on the larvae, when consuming the foliage that the larvae are persisting in.	Common (1990)
	Xerometra	indirect or opportunistic	pupae and larvae	Some larvae feed on the seeds in *Acacia* seed pods and can be found within fallen pods in the June-September months. They also pupate within a silk-lined cell in *Acacia* seed pods. Skinks may incidentally consume pupae/larvae when targeting seeds.	Common (1990)
Geometridae	*Amphicrossa*	direct or opportunistic	larvae	Adults are nearly all nocturnal and fly in late summer and autumn, with an annual life cycle. The larvae feed in the cooler months and pupate in the spring. The first-instar larvae of Geometridae are extremely active, usually after the first significant rains, as new growth is stimulated on food plants. Mortality is expected to be high during this dispersal period, due to predation, such as from skinks.	Common (1990)
Noctuidae	*Ectopatria*	indirect or opportunistic	pupae and larvae	Most adults are nocturnal, and the larvae of most noctuids feed on live foliage, flowers, flower buds and fruits of woody herbaceous plants, and can hide among the foliage during the day. Skinks can incidentally find or consume the larvae as they feed on the targeted vegetation.	Common (1990)
Plutellidae	*Plutella*	indirect or opportunistic	pupae and larvae	Peaks of pupae and larvae abundance in the southern hemisphere are from winter to spring. Some *Plutella* also largely target Brassicaceae, a major plant food source for the skinks, and are likely indirectly or opportunistically consumed when skinks feed on Brassicaceae vegetation.	Farias *et al.* (2020)

Table S5.3 The likely mode of consumption (indirect and direct) and part of the plant (leaves, flowers, buds, seedlings, or whole plant) of plant families consumed by *E. s. badia,* with supportive reasoning.

Family	Parts likely consumed	Indirect or direct	Reasoning	References
Asteraceae	flowers, leaves and buds	direct	Many are herbaceous annuals of high abundance in the spring. Flowers and leaves are soft and low to the ground. Fruit and seeds are too small to likely have nutritional value for skinks.	Western Australian Herbarium (2021)
Crassulaceae	whole plant	direct	The Crassulaceae family often have fleshy, succulent leaves, which are likely desirable for nutritional value plus providing water content.	Bender (1971), Hooks and Niu (2019)
Goodeniaceae	leaves and flowers	direct	Leaves can be soft and low to the ground; many *Goodenia* have basal leaves. Flowers are fairly large. Fruits are non-fleshy and unlikely to be attractive to skinks.	Western Australian Herbarium (2021)
Brassicaceae	seedlings, leaves and flowers	direct	Brassicaceae recorded in the area such as *Menkea australis* and the weed *Brassica tournefortii* are annual herbs with leaves at ground level, and have accessible, soft flowers.	Woodman Environmental Consulting (2012), Western Australian Herbarium (2021)
Plantaginaceae	leaves	direct	Mostly herbaceous, some with a basal concentration of leaves. *Plantago* often have rosetted, small flowers that are less likely to be desirable to the skinks. Fruits are non-fleshy.	Western Australian Herbarium (2021)
Amaranthaceae	seedlings, leaves	direct	Only Amaranthaceae recorded within scats was *Ptilotus*, which have erect stems, making flowers unlikely to be accessible. Leaves are accessible from the ground, but the small, softer seedlings may be the most palatable form.	Western Australian Herbarium (2021)
Fabaceae	fruit and seeds	direct	Of the Fabaceae, skinks were found to eat from *Senna* and *Acacia* plants, which include prickly trees that are unlikely to have reachable resources for skinks. However, some *Acacia* in the Mid West such as *A. tetragonophylla* seeds have elaborate, bright yellow arils which wrap tight around the seed and have a high lipid content, a likely attractant for vertebrates, as brightly coloured arils are associated with bird dispersal. Greater aril mass is also associated with bird compared to ant dispersal syndromes in *Acacia*\. Skinks are, therefore, likely to target the seeds and seed pods of *Acacias.*	Hughes et al. (1993), O'Dowd and Gill (1986)

Family	Parts likely consumed	Indirect or direct	Reasoning	References
Geraniaceae	whole plant	direct	*Erodium* are herbaceous with a basal concentration of leaves. Flowers are also aggregated in inflorescences. Fruits are non-fleshy but open up to release seeds.	Western Australian Herbarium (2021)
Poaceae	leaves and seeds	direct	Poaceae plants detected in skink scats including *Pentameris* are annual grasses, with low-lying leaves and accessible seed heads.	Western Australian Herbarium (2021)
Araliaceae	flowers and leaves	direct	Flowers are soft and aggregated in inflorescences, making them easy to access and of concentrated reward for effort. They are also herbs with a basal concentration of leaves, making them accessible to skinks. Fruits are non-fleshy and are unlikely to be targeted.	Western Australian Herbarium (2021)
Caryophyllaceae	whole plant	direct	Mostly herbaceous, with well-developed leaves. Flowers are aggregated in inflorescences, making them a higher reward for effort. Dehiscent fruit are up to 15mm long, with small to minute seeds.	Western Australian Herbarium (2021)
Pteridaceae	new fronds or spores or rhyzomes	direct	Rhizomatous, perennial herbs or ferns, generally less than a metre high, accessible to skinks.	Western Australian Herbarium (2021)
Scrophulariaceae	dropped flowers	direct	Many Scrophulariaceae in the area are *Eremophilas*, which are small trees and shrubs, with foliage less likely to be accessible to skinks. Fruits are usually non-fleshy and are unlikely to attract skinks. Flowers are fairly large and soft and would be accessible to skinks once dropped on the ground.	Western Australian Herbarium (2021)
Haloragaceae	flowers and leaves	direct	Herbaceous, with flowers solitary or aggregated in inflorescences. Fruits are non-fleshy and are unlikely to be attractive to skinks.	Western Australian Herbarium (2021)
Campanulaceae	seedlings, leaves and flowers	direct	Mostly perennial herbs, with a basal concentration of leaves accessible for skinks. Flowers either solitary or aggregated as inflorescences. Fruits are usually non-fleshy with small seeds and are unlikely to be attractive to skinks.	Western Australian Herbarium (2021)
Thymelaeaceae	seedlings, leaves, and flowers	direct	*Pimelea* can be shrubs or annual herbs. Leaves are small- to medium-sized and accessible from the ground. Flowers are solitary or aggregated in inflorescences. Fleshy cotyledons may make seedlings desirable for skinks. Fruits can be fleshy and non-fleshy.	Western Australian Herbarium (2021)
Cupressaceae	seeds	direct	*Callitris* grow as trees or shrubs, meaning flowers and leaves are unlikely accessible to skinks. Fruits are non-fleshy, but skinks may target fallen seeds.	Western Australian Herbarium (2021)
Protaceae	seeds or nothing	contamination /direct	Proteaceous plants such as *Grevillea* and *Hakea* found in the area grow as small trees and shrubs, often with long, thin, and spinose leaves that are unlikely attractive to skinks. The genetic detection may have come from the diet of consumed invertebrates, or from pollen	Western Australian Herbarium (2021), Auld and Denham

Family	Parts likely consumed	Indirect or direct	Reasoning	References
			or leaf litter contamination on the scat. There are records of vertebrate seed predation on Proteaceous plants, so skinks may target fallen seeds.	(1999), Bond and Breytenbach (1985)
Myrtaceae	NA	contamination	As Myrtaceacous plants such as *Eucalyptus* within the area are tree form, leaves and flowers are unlikely to be accessible for skinks. Leaf litter or bark from the log pile are likely to have contributed to the genetic material found during scat analysis.	Western Australian Herbarium (2021)
Pinaceae	NA	contamination	Pinaceous plants are not common within the landscape, and pines are not accessible to skinks for feeding on leaves or flowers. Pollen contamination of scats is the likely reason for the genetic material found during scat analysis.	Woodman Environmental Consulting (2012)

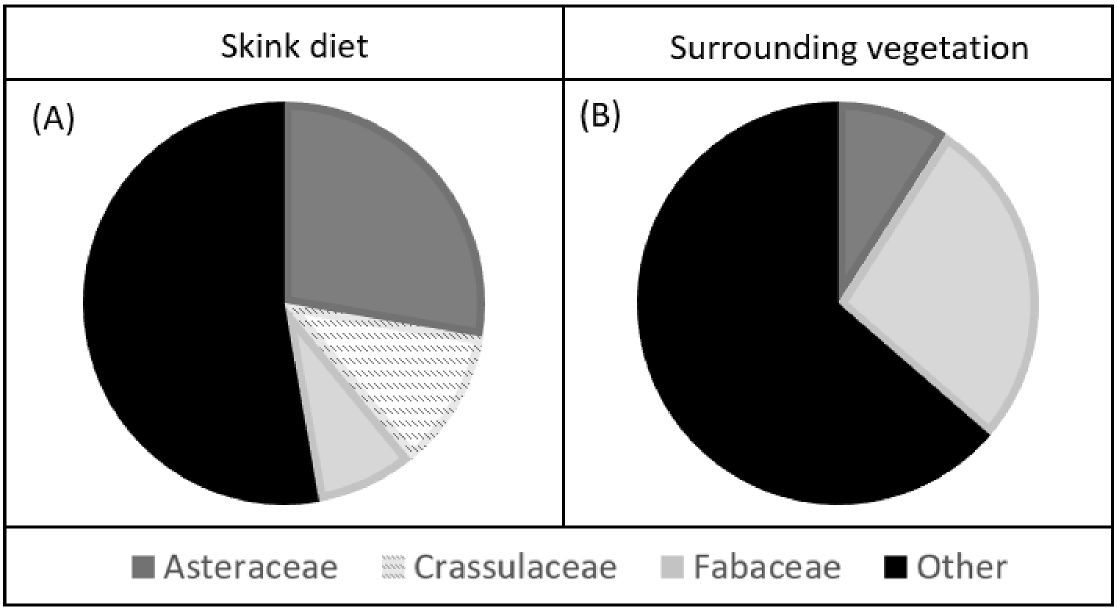


Figure S5.1 Pie charts highlighting the: **(A)** proportion of scat samples containing plant matter from different family groups, and **(B)** proportion of species per plant family observed in adjacent vegetation quadrats.

Chapter 6. General discussion

Photo: Holly Bradley

6.1 Introduction

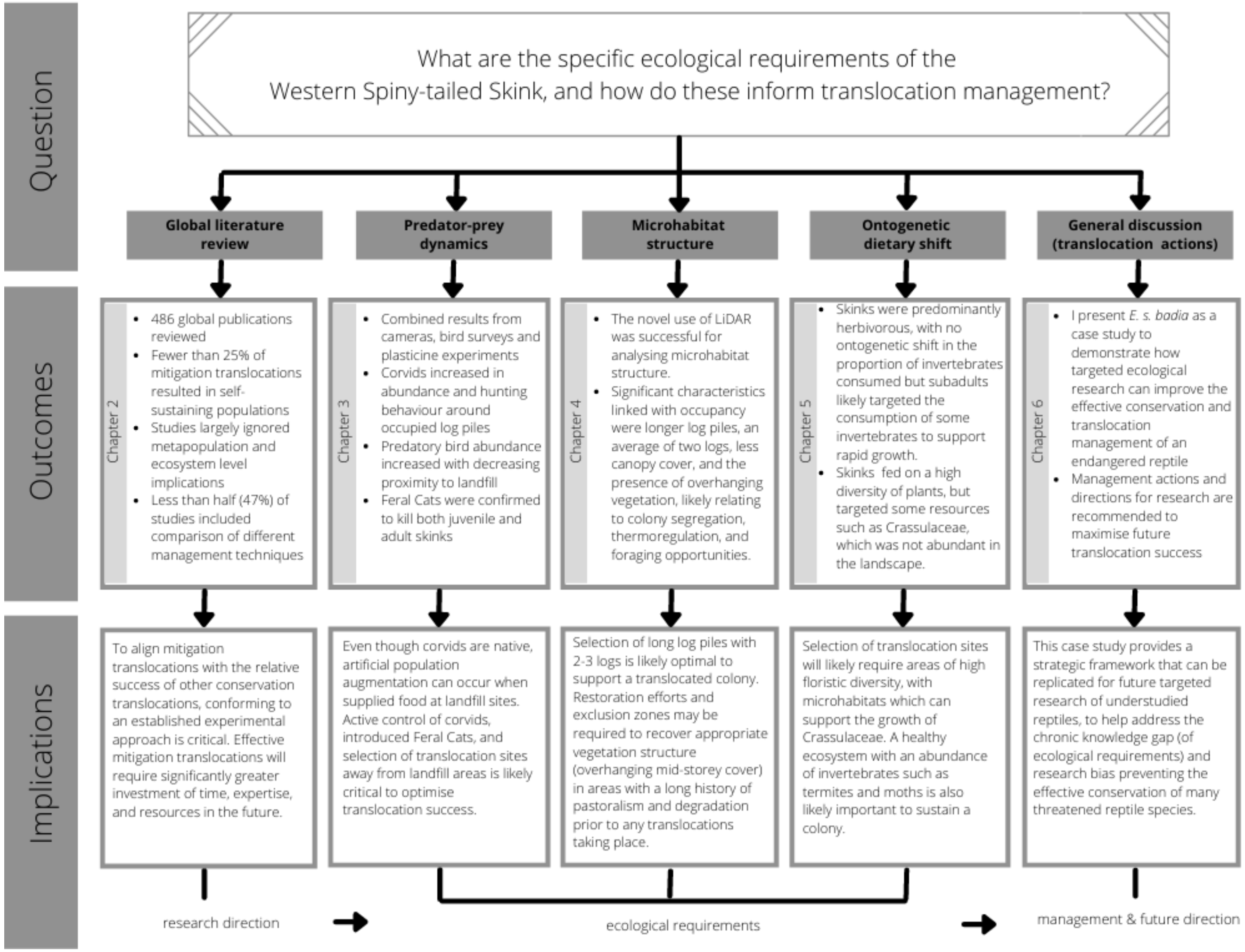
Australia is a global hotspot for reptile diversity, harbouring approximately ten percent of all currently described reptile species globally and more species than any other country in the world (Geyle et al. 2020; Tingley et al. 2019). However, a challenge of having such high diversity, largely concentrated in poorly accessible, sparsely populated arid and semi-arid habitats (Powney et al. 2010; Roll et al. 2017), is a chronic knowledge gap concerning the ecology of the vast majority of these species (Tingley et al. 2019). Only approximately 64% of the world's reptile species known globally have had their conservation status assessed by the IUCN (Tingley et al. 2019), and, of these, species in Australia are some of the most under-represented in conservation status assessments (Meiri and Chapple 2016). As such, reptile species are under-represented in recovery planning in Australia (Walsh et al. 2013), which may deprive them of conservation investment as listed threatened species are more likely to receive management and investment than are non-listed species (Farrier et al. 2007).

Whether fauna are charismatic (prone to popular appeal) or not has been found to outweigh threatened status in garnering public support for conservation (Colléony et al. 2017). As a result (i.e., because they are not considered charismatic), the general public tend to regard reptiles as having a lower conservation priority than other taxa (Czech et al. 1998). Indeed, the stigma and fear associated with reptiles, particularly snakes (da Silva et al. 2021), has resulted in antagonistic behaviour by members of the public, such as intentional road-kills (Paul Ashley et al. 2007; Secco et al. 2014). Charisma, body size, and level of appeal also largely drive the listing and planning processes for threatened species in Australia (Walsh et al. 2013), resulting in bias towards large mammals and birds (Ducarme et al. 2013; Feldhamer et al. 2002; Kellert, 1993). Therefore, despite high diversity, large knowledge gaps, and a need for conservation assessment, reptiles are under-represented in conservation research and planning, both in Australia and globally (Geyle et al. 2020; Tingley et al. 2016).

There is also a clear geographical bias in research on reptile ecology and conservation, with conservation research historically largely concentrated in the northern hemisphere (Lawler et al. 2006; Rozzi et al. 2008). Deserts are the least studied biome in global ecological literature in this context (Martin et al. 2012), and in Australia, ecological research on terrestrial reptiles is largely driven by the accessibility of habitat (Piccolo et al. 2020). This limits research attention to areas of lower reptile biodiversity, given the comparatively poor accessibility of semi-arid and arid regions of northern Australia, which harbour the greatest species richness and the highest levels of Data Deficiency (Doughty et al. 2011; Powney et al. 2010; Roll et al. 2017; Tingley et al. 2019).

The Australian population is one of the most urbanised in the world (Flies et al. 2017). Even in the less populated arid and semi-arid areas, there are other pressures on biodiversity, including agriculture, pastoralism and mining operations in regions harbouring significant reptile diversity, such as in much of Western Australia (Industry Commission 1998; Roche and Mudd 2014). Wildlife relocations from sites to be cleared for urbanisation or mineral extraction are often seen as a mitigation tool to prevent mortality in these areas (Cowan et al. 2020; Milne et al. 2021; Thompson and Thompson 2015). However, most mitigation translocations fail, such as failing to establish self-sustaining populations (Bradley et al. 2020; Germano et al. 2015; Sullivan et al. 2015), due to a lack of detailed information on the establishment, persistence, metapopulation and ecosystem-level requirements of a species prior to translocation (Armstrong and Seddon 2008). There is also often a lack of legal obligation to follow this strategic approach, if the goal of a mitigation translocation is simply to remove wildlife from an area of human-wildlife conflict. Without an informed ecological knowledge base to support mitigation translocation planning, even well-intentioned translocation attempts run the risk of undermining biodiversity conservation efforts (Bradley et al. 2020).

This thesis presents the Western Spiny-tailed Skink (*Egernia stokesii badia*) as a case study of how identifying and addressing knowledge gaps on the ecological requirements of a threatened reptile in Australia is key to informed, effective management. The following discussion recapitulates the contribution of each previous chapter towards this aim (Figure 6.1). In summary, Chapter 2 (**review**) provided a global literature review which investigated the efficacy of mitigation translocation as a conservation tool and the major knowledge gaps inhibiting best practice. Chapter 3 (**predation**) then provided empirical evidence for the species which can be considered predators of *E. s. badia*, plus those of particular threat due to their abundance and behaviour in response to the presence of log piles inhabited or uninhabited by skinks, plus proximity to landfill. Chapter 4 (**microhabitat requriements**) provided discussion on optimal log pile site characteristics, through the novel use of LiDAR as a measurment tool to quantify structural features which promote occupancy by skink colonies. Chapter 5 (**diet**) examined the plant and invertebrate diet of *E. s. badia* and the ontogenetic dietary shift from subadult skinks likely targeting high-reward prey items to adult skinks with a more opportunistic supplementation of their herbivorous diet with invertebrates, through visual identification and DNA metabarcoding. The following discussion provides further details on the significant findings and management implications of this research.



What are the specific ecological requirements of the Western Spiny-tailed Skink, and how do these inform translocation management?

- 486 global publications reviewed
- Fewer than 25% of mitigation translocations resulted in self-sustaining populations
- Studies largely ignored metapopulation and ecosystem level implications
- Less than half (47%) of studies included comparison of different management techniques

- Combined results from cameras, bird surveys and plasticine experiments
- Corvids increased in abundance and hunting behaviour around occupied log piles
- Predatory bird abundance increased with decreasing proximity to landfill
- Feral Cats were confirmed to kill both juvenile and adult skinks

- The novel use of LiDAR was successful for analysing microhabitat structure.
- Significant characteristics linked with occupancy were longer log piles, an average of two logs, less canopy cover, and the presence of overhanging vegetation, likely relating to colony segregation, thermoregulation, and foraging opportunities.

- Skinks were predominantly herbivorous, with no ontogenetic shift in the proportion of invertebrates consumed but subadults likely targeted the consumption of some invertebrates to support rapid growth.
- Skinks fed on a high diversity of plants, but targeted some resources such as Crassulaceae, which was not abundant in the landscape.

- I present *E. s. badia* as a case study to demonstrate how targeted ecological research can improve the effective conservation and translocation management of an endangered reptile
- Management actions and directions for research are recommended to maximise future translocation success

To align mitigation translocations with the relative success of other conservation translocations, conforming to an established experimental approach is critical. Effective mitigation translocations will require significantly greater investment of time, expertise, and resources in the future.

Even though corvids are native, artificial population augmentation can occur when supplied food at landfill sites. Active control of corvids, introduced Feral Cats, and selection of translocation sites away from landfill areas is likely critical to optimise translocation success.

Selection of long log piles with 2-3 logs is likely optimal to support a translocated colony. Restoration efforts and exclusion zones may be required to recover appropriate vegetation structure (overhanging mid-storey cover) in areas with a long history of pastoralism and degradation prior to any translocations taking place.

Selection of translocation sites will likely require areas of high floristic diversity, with microhabitats which can support the growth of Crassulaceae. A healthy ecosystem with an abundance of invertebrates such as termites and moths is also likely important to sustain a colony.

This case study provides a strategic framework that can be replicated for future targeted research of understudied reptiles, to help address the chronic knowledge gap (of ecological requirements) and research bias preventing the effective conservation of many threatened reptile species.

Figure 6.1 Conceptual framework highlighting the overall research question, chapter divisions, summarised outcomes, and wider implications of this thesis.

6.2 Summary of findings

The key questions in reintroduction biology (Table 6.1) are critical for informing the requirements of a translocated organism or group of organisms prior to relocation (Armstrong and Seddon 2008). For species in well-studied ecosystems, many of these questions are answerable from the literature, leaving just the translocation-specific questions to be answered during translocation planning, such as the optimal number of individuals to harvest for both the source and recipient sites (Armstrong and Seddon 2008). However, for less studied regions such as deserts (Martin et al. 2012), including arid central and western Australia, there is a knowledge barrier which prevents both timely and best practice translocation. For many reptiles, including threatened species, even their basic ecological requirements remain unknown (Meiri 2016), such as diet and habitat (e.g. *E. s. badia*; Pearson et al. 2012), and require research prior to translocation. This is in addition to overcoming practical and logistical constraints, such as creating fenced predator exclusion zones if necessary. For many mitigation translocations, the primary goal may also be resolving a human-wildlife conflict, which can clash with any conservation related goals or outcomes. Achieving positive conservation outcomes, therefore, relies on the required interventions being framed within the state or national legislation.

Table 6.1 Armstrong and Seddon's (2008) 10 key questions in reintroduction biology at the population, metapopulation and ecosystem level.

Population		Metapopulation	Ecosystem
Establishment	Persistence		
1. How is establishment probability affected by size and composition of the release group?	3. What habitat conditions are needed for persistence of the reintroduced population?	5. How heavily should source populations be harvested?	8. Are the target species/taxon and its parasites native to the ecosystem?
2. How are post-release survival and dispersal affected by pre- and post-release management?	4. How will genetic makeup affect persistence of the reintroduced population?	6. What is the optimal allocation of translocated individuals among sites?	9. How will the ecosystem be affected by the target species and its parasites?
		7. Should translocation be used to compensate for isolation?	10. How does the order of reintroductions affect the ultimate species composition?

Egernia s. badia is an example of an endangered reptile, endemic to the arid and semi-arid regions of Western Australia (Pearson 2012). The skinks are colonial (Chapple

2003), and groups of closely related individuals live together in long-term log pile residences (Pearson 2012). Beyond this basic information, little was known about the ecological requirements of this subspecies prior to undertaking this thesis research, with inferences largely made from what is known of other *Egernia stokesii* subspecies, despite a different distribution and their use of log piles rather than rock crevices (Pearson 2012). While there are anecdotal reports of unsuccessful translocation attempts having been made in the past, there are no records of successful translocation of this species.

Due to the endangered status of the subspecies, the highest caution must be taken to prevent any further colony losses, and detailed investigation into their complex ecological requirements should be undertaken prior to any future translocations. As much of the range of *E. s. badia* is located on mining tenements in the Mid West region of Western Australia it is likely that future translocation of some populations will be required due to clearing for mineral exploration and extraction activities (Pearson 2012); research into understanding their ecological requirements to inform translocation management is therefore of high priority. As such, I designed a three-year research program to understand the: (i) predators (Chapter 3); (ii) microhabitat requirements (Chapter 4); and (iii) adult and juvenile dietary requirements (Chapter 5) of *E. s. badia*, to directly inform on-ground management and future translocation planning for the skink.

6.2.1 Predation

The major relvant findings for predation management relating to *E. s. badia* translocation pertain to: (i) translocation site selection; and (ii) the targeted management of predators. Firstly, two predators of significant threat for the skink are Feral Cats (*Felis catus*) (Desmond and Chant 2001; Lee-Steere 2008; Pearson 2012) and corvids (*Corvus orru, C. bennetti,* and *C. coronoides*). Globally, native corvid populations can become overabundant in anthropogenically modified landscapes with increased access to resources such as at landfill sites (Coates and Delehanty 2004; Marzluff et al. 2001; Preininger et al. 2019). Therefore, selection of translocation sites for skink colonies needs to be as distant as possible from potential anthropogenic food sources, like landfill sites, which can also supplement and augment populations of generalist invasive predators such as Feral Cats (Mirmovitch 1995; Tennent and

Downs 2008). Translocation site selection away from linear infrastructure such as powerlines, which are correlated with an increase in corvid abundance in the U.S. (Cunningham et al. 2016; Howe et al. 2014), is also important to reduce potential predation pressure of individuals more vulnerable to predation post-translocation.

Targeted management to control the abundance of predators in areas marked for translocation is also necessary. For example, targeted lethal control of invasive Feral Cats, such as baiting (Christensen et al. 2013; Johnston et al. 2011), shooting (Algar and Burrows 2004; Garrard et al. 2017), and trapping (Algar and Burrows 2004; Molsher 2001) near translocation areas is likely to be important in reducing predation risk for translocated skinks. This is because, in addition to being one of the predators of greatest relative activity observed throughout my study, my work also confirmed the capability of Feral Cats to kill adult and subadult skinks. Methods to avoid overabundance of native corvids would also be valuable, through management of anthropogenic food sources. Control of corvids and Feral Cats is particularly important, as both are attracted to novel objects and areas (Bradshaw et al. 2000; Church et al. 1994; Heinrich 1995; Miller et al. 2015; Reina 2010), such as translocation sites. Controlling overabundant and invasive predators is, therefore, a priority, as any site modifications to enhance the structure of coarse woody debris (CWD) may attract these predators, increasing the risk of translocation failure.

6.2.2. Microhabitat requirements

Egernia s. badia have specific microhabitat requirements that must be considered when selecting or modifying an optimal translocation site. Skink occupancy is highest in long log piles with an average of two logs with some overhanging vegetation, preferably at a mid-storey height and reduced at canopy height (Figure 6.2). This is likely to ensure enough space for segregation between members of the skink colony, which vary in size between juveniles, adults, and gravid females (Schwarzkopf et al. 2010). The presence of overhanging vegetation to provide microhabitat variability is also likely to be important in supporting behavioural thermoregulation by providing temperature gradients and thermal buffering (Grimm-Seyfarth et al. 2017). This is particularly vital in an arid landscape with scattered vegetation (Grimm-Seyfarth et al. 2017), where daytime temperatures often far exceed the upper thermal tolerance limits of most reptiles (Bradshaw 1986).

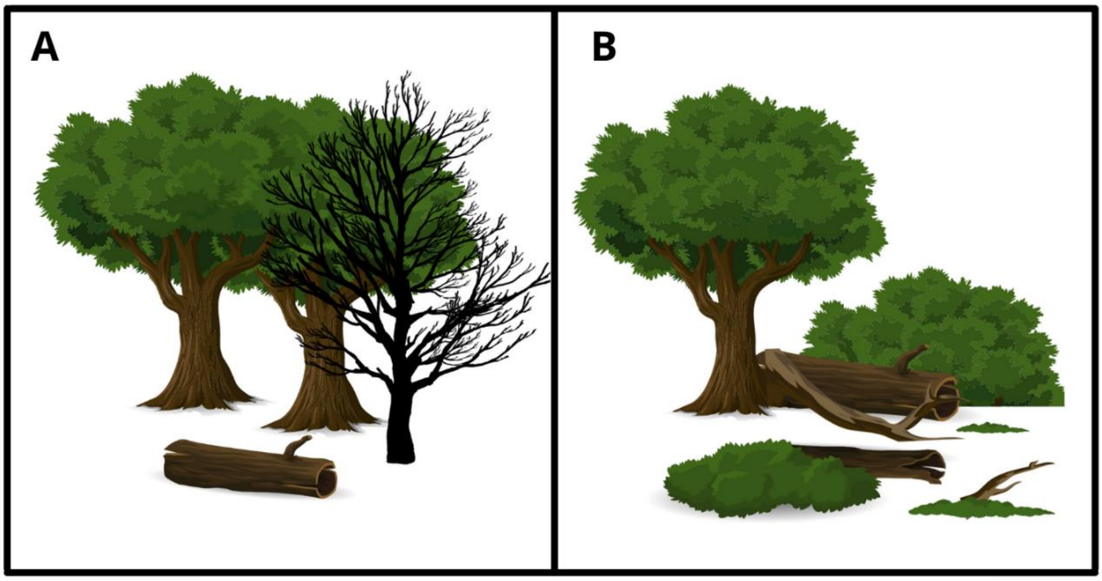


Figure 6.2 Graphic example of relatively subtle differences between unsuitable translocation site microhabitat structure for spiny-tailed skinks **(A)** with a single log, no understorey or mid-storey cover, and expansive canopy cover with multiple perching options for avian predators; and **(B)** a suitable translocation site with a complex, longer log pile with two logs and multiple branches, adjacent and overhanging mid-storey cover and understorey cover, and minimal canopy cover.

In some instances, the presence of canopy cover can provide perches for ambush predators (Hawlena et al. 2010; Oversby et al. 2018). As corvids were found to increase in relative activity and focus their hunting behaviour around CWD inhabited by *E. s. badia*, this potentially explains the trend for log pile selection with reduced canopy cover and perching options for avian predators. Overall, when selecting or modifying translocation sites, microhabitat characteristics which facilitate optimal thermoregulation, predator refuge and social segregation are important considerations to maximise the likelihood of colony establishment and persistence.

6.2.3 Diet

Egernia. s. badia was identified to have a predominantly herbivorous diet (approximately 91%), supplemented by the opportunistic consumption of invertebrates, except in the case of subadults, which appeared to directly target some invertebrate prey items of high nutritional value (cicadellids) likely to facilitate rapid growth and development. The skinks fed from a high diversity of food plants but in particular, the plant family Asteraceae was largely relied upon by both adult and subadult skinks. Asteraceae are largely made up of small, soft annual flowering plants of high abundance in the spring months of the Western Australian Mid West region, and a high abundance in the diet likely reflects the high abundance in the landscape during this time. Another plant family of high abundance in the skink diet was Crassulaceae, generally characterised by plants of fleshy, succulent leaves (Western Australian Herbarium 2021). In contrast to Asteraceae, Crassulaceae have a narrower niche breadth, often occurring in moist, shaded areas (Green 1947; Sheppard et al. 2006), and were, therefore, likely directly targeted by the skinks. Crassulaceae likely offer both a valuable source of nutrients and water, which has also been suggested for other reptiles persisting in an arid habitat (Loehr 2002). In the context of selecting appropriate translocation sites, provision of a high floristic diversity including annual species, plus microhabitat complexity to support the growth of more specialist plants such as *Crassulas,* is likely critical to support the foraging requirements of an *E. s. badia* colony. A healthy ecosystem which supports a diverse invertebrate community is also likely important to support a colony, particularly for the growth and development of younger subadults.

6.3 Significance of the thesis

Australia is a global hotspot for reptile diversity, particularly in areas of high aridity (Powney et al. 2010; Roll et al. 2017; Tingley et al. 2019). However, large knowledge gaps surround the conservation status of reptile species, particularly in Australia (Tingley et al. 2019), and a conservation research and planning bias towards larger, more charismatic vertebrates, particularly birds and mammals (Ducarme et al. 2013), limits ecological knowledge of reptiles at risk of extinction, and the capacity for their conservation management. The research in this thesis is structured around the Western Spiny-tailed Skink (*Egernia stokesii badia*) as a case study to demonstrate how targeted ecological research can improve the effective conservation and translocation management of a largely unstudied, endangered reptile. Through the combination of a number of technologies and experimental techniques (including terrestrial LiDAR, DNA metabarcoding, plasticine replica surveys, unbounded point count surveys, and camera trapping), I have studied and identified (i) the predators, (ii) the microhabitat requirements, and (iii) the adult and subadult dietary requirements of *E. s. badia*. This information substantially increases the ecological knowledgebase for this endangered subspecies, fundamentally increasing the capacity for effective management, and the likelihood of successful translocations in the future.

Translocation design and site selection can now be informed by a knowledge of which predators require population management, the log pile and vegetation structural requirements of a translocation site, and food plants required in proximity to the refuge site. This significantly improves the previous practice basing translocation site selection on a broad understanding of suitable vegetation type and the presence of logs.

This research also presents the use of terrestrial LiDAR as an effective technique for the highly accurate quantification of microhabitat structural requirements of a threatened species. This method has numerous possible applications, such as the assessment of other complex microhabitat structures, including rock piles or cave sites, to better understand sites for protection, translocation, or recreation in restoration to encourage recolonisation by threatened species. Overall, this case study provides a strategic framework, and a novel application of a technology, that can be replicated for future targeted research of understudied species, particularly reptiles, to help address the chronic knowledge gap and research bias preventing the effective conservation of many of our threatened reptile species.

In addition to providing positive on-ground management implications for *E. s. badia,* the research in this thesis is also the first of its kind to exemplify how the efficacy of mitigation translocations as a conservation tool can be improved through following the same scientific rigour and framework as expected for conservation translocations (Armstrong and Seddon 2008). I answered the call by Armstrong and Seddon (2008) for translocation biologists to consider the biological requirements of the target species when understanding ‘habitat’, and not just focus on the easier, rapidly assessable landscape features such as vegetation type. As such, I investigated habitat requirements in the context of predators, microhabitat structure and food for *E. s. badia*. My advancement of the structural framework by Armstrong and Seddon (2008) in application to a mitigation translocation also has broad implications for the protection of threatened species. As mitigation translocations continue to be used as a compensatory measure for the ‘rescue’ of threatened fauna at sites marked for development, the research in this thesis makes it clear that a ‘continue as normal’ approach where the speed and scope of infrastructure development sets the pace for managers to conduct translocations in an *ad hoc* manner without feasibility analysis (Gardner and Howarth 2009), is no longer acceptable (Chapter 2). A significant investment into the planning, design, implementation and monitoring of translocation events is required, for mitigation translocations to be considered an effective management tool (Bradley et al. 2020).

6.4 Looking forward

The targeted research in this thesis has helped to identify specific ecological requirements of *E. s. badia* for consideration during translocation, including target predators requiring management, microhabitat structural requirements of a translocation site, and food species required in proximity to a release site. While each of these factors is key to an ecological understanding of the skink, they only help to answer question three (what habitat conditions are needed for persistence of the reintroduced population?) of the ten questions considered critical to maximise translocation success (Armstrong and Seddon 2008). Therefore, while our knowledge of the translocation requirements of *E. s. badia* has increased, continued research into further understanding the optimal translocation requirements is important to maximise the likelihood of successful management in the future. Here I discuss the potential future directions for research to improve the translocation management of *E. s. badia* and increase the efficacy of mitigation translocation for the conservation of colonies.

6.4.1 Immediate steps to improve management

An important topic for future research is the translocation establishment requirements of *E. s. badia*, with a focus on pre- and post-release management. The most common cause of reptile translocation failure is homing behaviour and substantial movement away from the release site (Germano and Bishop 2009), which suggests that trials of different management solutions are an important starting point in developing a standard operating procedure by which to translocate *E. s. badia*. Soft-release fencing is a commonly used mitigative technique (Cain et al. 2018; Calvete and Estrada 2004; De Milliano et al. 2016; Parker et al. 2019). However, monitoring is important to ensure that translocated individuals are not subject to high vulnerability from ‘fence pacing’ behaviour when inhibited by a fence line (Farnsworth et al. 2015). This is particularly important as corvids, an identified predator that is potentially overabundant close to mining infrastructure (Chapter 3), are known to use anthropogenic vertical structures such as power lines and fence posts, which can enhance predation efficiency (Prather and Messmer 2010). Trials of deterrents such as overhead lines and wires, which have been used in other environments to exclude avian predators (Curtis et al. 1996; Moerbeek et al. 1987), may be appropriate.

Metapopulation dynamics are also of significant importance for optimal translocation design (Armstrong and Seddon 2008). Understanding colony home range size and potential use of satellite log pile structures within their home range is important to inform the optimal spatial distribution of log piles at release sites to prevent intraspecific competition for resources. Understanding home range size is an important component in determining the ecosystem carrying capacity (Baber and Craig 2003; Nagy and Haroldson 1990). Analysis of dispersal capacity can also inform translocation site selection, through understanding the appropriate density of coarse woody debris in the landscape to facilitate successful dispersal and colonisation of other log piles in the future. Radio-tracking is a common, useful tool used to determine movement patterns and habitat use of species (Beier et al. 1995; Krone and Treu 2018; Wassens et al. 2008). However, the spinose scales of *E. s. badia* prevents safe attachment of a radio-tracking or GPS device without protruding to an extent that would risk prevention of re-entering log crevices. The endangered status of *E. s. badia* also means that surgical implantation of tracking devices would be a high-risk solution. Such restrictions call for an innovative approach to future studies. One possible method to examine metapopulation dynamics could be to extract individual skink DNA from scats collected from colony latrine piles found next to inhabited logs (Lanham 2001), as a non-invasive method to estimate genetic structure and diversity within and between colonies (Pearson et al. 2015). Observational or genetic studies of colony size and composition would also assist with understanding the variation in resources and structure of log piles required to support different colony types.

One further possibility for investigation is the role of stress mitigation and management to promote successful survival and establishment. Few studies have investigated the impact of stress to herpetofauna pre- and post-translocation (Germano and Bishop 2009). However, a range of studies have found that translocation alters the stress physiology of wildlife (Bosson et al. 2013; Cabezas et al. 2007; Letty et al. 2000) and chronic stress is likely a major contributor to translocation failure (Dickens et al. 2009). When stress is considered during the translocation process, it is often limited to a single stressor (Kock et al. 1990; Teixeira et al. 2007), such as capture stress, when in reality there are possible stressors at the capture, captivity, veterinary examination, transportation, release, establishment, and monitoring stages of a translocation (Teixeira et al. 2007). Evaluation of behavioural and physiological stress responses at each stage of the translocation process can be important to optimise each step, such as the capture method, duration of a capture event, number of animals in a transportation enclosure, release method, and the transport time to the release site (Teixeira et al. 2007). The skink *Eulamprus heatwolei* for example, suffered more stress (measured by plasma corticosterone levels) from microchip implantation, blood sampling, an unfamiliar enclosure, or exposure to a heterospecific lizard than it did from handling and measuring, toe-clipping for identification or exposure to predator scent, suggesting that it is not always intuitive what processes are more stressful for reptiles (Langkilde and Shine 2006). Overall, there are clear next steps for future research to delineate the best management strategies and translocation site selection criteria moving forward.

6.4.2 Adaptive management

Beyond the initial establishment phase, it is also important to consider the life-history strategy of *E. s. badia* to understand the feasibility of ongoing persistence post-translocation. For example, the reproductive requirements and recruitment rates of this subspecies are currently unknown (Pearson 2012). While known to be viviparous, the litter size and seasonality cues for reproduction are also unknown (Pearson 2012). These life-history traits are important to understand in the context of climate-resilient translocation planning, as species with restricted ranges, long generation times, limited genetic variation, and limited ability to disperse are vulnerable to demographic collapse and extinction in response to climate change (Sinervo et al. 2010; Urban 2015). Declines in these species are also notoriously difficult to detect, as long-lived adults can persist in the face of gradually eroding habitat suitability and reduced recruitment rates, known as the ‘perception of persistence’ (Lovich et al. 2018). Due to a trend of consistent warming and changes to rainfall patterns across Australia (Hughes 2003), areas currently suitable to support *E. s. badia* colonies may become less viable in the near future. The relocation of colonies outside of their historical range to combat extinction risks associated with climate change is a controversial process known as assisted colonisation (Seddon et al. 2009), but may be worth investigating to safeguard the future of the subspecies. Subfossil records or discussion with Indigenous Elders to learn of Oral History relating to *E. s. badia* may also help inform translocation/assisted colonisation site selection.

Although failure rates of reptile translocations are still high, the proportion of successful translocations continues to increase (Germano and Bishop 2009), and there are success stories on which to base future improvements to management. For example, previous translocations of highly threatened reptiles have successfully achieved assisted colonisation based on thermal and hydrological projections (Bouma et al. 2020; Mitchell et al. 2013). However, in the context of a cool Mediterranean climate, hydroperiod length and temperature were considered critical factors in promoting individual growth and persistence for these translocations (Mitchell et al. 2013). Assisted colonisation was also considered the best option as any potential suitable habitat near the original population had been modified or destroyed (Bouma et al. 2020). The possibility of using assisted colonisation of *E. s. badia* to track the progression of their suitable climatic envelope southwards may be less viable, as habitat availability and quality significantly decrease moving from their northern range in the Mid West region, to the southern part of their range in the Wheatbelt region, where there has been widespread clearing for agriculture, such as in the central Wheatbelt where over 93% of the original vegetation, and up to 97% of the native woodland, has been removed (Saunders 1989). However, investigation into the thermoregulatory niche of *E. s. badia* may indicate the need for a southern extension of their range in response to the changing climate, which would require pre-emptive habitat restoration at southern degraded or cleared sites for suitable establishment prior to future translocations. As juveniles are conjectured to take at least five years to disperse from their original colony site (Lanham 2001), modelling may be required to determine if the creation of habitat corridors for self-dispersal or if assisted colonisation is more likely to match the rate of projected climate change.

The capacity for adaptive management may also be considerably limited by the responsive nature of mitigation translocations. As *E. s. badia* translocations in the future are likely to be in direct response to habitat clearing for mineral extraction (Pearson 2012), trials are likely to be at a small scale, over a limited timeframe. Although there is currently no known founder size threshold which definitively increases the success likelihood of reptile translocations (Germano and Bishop 2009), a number of failures or difficulties with reptile translocations have been attributed to the release of an insufficient number of individuals (Ewen et al. 2014). Previous reptile translocation successes which have had the benefit of a large number of translocated individuals have sometimes been facilitated through a captive breeding program (Schubert 2020; Towns et al. 2016). Large founder populations can be important to sufficiently overcome the demographic and stochastic problems of small populations (Moritz 1999; Towns and Ferreira 2001), and a greater number of propagules for translocation is likely to increase the prospect of translocation success (Griffith et al. 1989; Towns et al. 2016). Investigation into the viability of a captive breeding program for *E. s. badia* may be useful to both increase the founder group size and likelihood of translocated population persistence, as well as provide substantial replicates to trial different management techniques, such as stress abatement measures or installation of soft-release fencing. Adherence to an experimental, scientific approach is often considered a criterion for translocation success (Bradley et al. 2020; Taylor et al. 2017; Towns et al. 2016), and is, therefore, a desirable goal for future translocation efforts. Irrespective of the overall population size to be translocated, colonies should be kept together and released at the same recipient log pile site. Maintenance of social structure is likely to be a key element for maximising translocation success, as has been observed for other translocation studies (e.g., Clarke et al. 2002; Sheir 2006).

In addition to climate-resilient planning and maximising the number of translocation propagules, another common theme of successful reptile translocations, or those in the early stages of success, has been to create insurance populations on islands (Baling et al. 2013; Michaelides et al. 2015; Schubert 2020). However, this is a non-viable option for *E. s. badia,* due to a lack of suitable habitat on islands, plus the chance of genetic admixture between mainland individuals and distinct island and coastal populations of *E. stokesii* (Pearson 2012). However, islands are often seen as a desirable recipient site for translocations due to their isolation as a distinct management unit for more effective control or isolation from threatening processes (Algar et al. 2020; Cardoso et al. 2009; Moro 2003). Similar benefits can be recreated on the mainland, through construction of large, fenced enclosures to prevent invasive species incursion once threats have been eliminated within the fenced area (Burns et al. 2012; Legge et al. 2018). A number of successful translocations have opted to release individuals at an enclosure site (Lohr et al. 2021; Pople et al. 2001; Short and Turner 2000). As *E. s. badia* occurs in areas with a long history of pastoralism and habitat degradation from grazing and mining operations (Payne et al. 1998), and invasive predators such as the Feral Cat have been confirmed to hunt the skinks (Chapter 3), establishment of a large enclosure site to focus management efforts may be an effective strategy moving forward in the future trial of translocations.

Finally, perhaps the most effective way to understand priority areas for research is through effective monitoring. Without well-documented translocation examples, there are no model systems to emulate (Germano et al. 2015), and no empirical evidence for best practice techniques without explicitly testing and comparing the effectiveness of different management alternatives (Taylor et al. 2017). Monitoring also needs to be focused and targeted to address questions identified *a priori*, to maximise the effective use of conservation funds (Nichols and Williams, 2006). Therefore, budgeting and determination of *a priori* hypotheses for testing must be incorporated into the translocation planning process (Bradley et al. 2020), to provide critical monitoring data for informed best practice techniques, and to direct future research.

6.5 Thesis conclusion

My research presented in this thesis has found that mitigation action, such as translocation for the protection of reptiles, runs the risk of being an inadequate use of conservation dollars and time if current habitat clearing patterns persist without a significant, long-term investment into understanding the ecological requirements and conservation status of reptiles. This is particularly relevant to reptile translocations in Australia, which is a global hotspot for reptile diversity (Geyle et al. 2020; Tingley et al. 2019), yet suffers a chronic knowledge gap surrounding the conservation status and ecological requirements of its diverse reptilian fauna (Tingley et al. 2019). The overarching question of this thesis was "what are the ecological requirements of *E. s. badia*, and how do they inform translocation management?" Using *E. s. badia* as a case study, this thesis provides a clear strategic framework for the targeted research of meaningful ecological factors which influence on-ground translocation decision making. It was demonstrated that even native predators can become an issue for skinks when populations are artificially augmented around anthropogenic food sources, such as landfill sites on mining tenements. Corvids were shown to both increase in relative activity and hunting activity around inhabited log pile sites, compared to uninhabited log piles and control sites with no logs. Cats were also a confirmed predator of both adult and subadult skinks and, therefore, capable of depleting an entire colony. The novel application of terrestrial LiDAR was also confirmed as an effective tool to quantify the structural microhabitat requirements of the cryptic skink colonies, a method which has extensive applications for the assessment of complex microhabitat types essential to the persistence of other threatened species, such as rock piles and cave structures. Skinks were found to be more likely to occupy longer log piles with an average of two logs, and overhanging vegetation preferably at the mid-storey height. Lastly, the complementary use of visual identification and DNA metabarcoding was also successful at identifying that *E. s. badia* has a largely herbivorous diet, supplemented by the consumption of invertebrates such as moths and termites. The plant families Asteraceae and Crassulaceae were also found to be largely relied upon by both adults and subadults, marking the close proximity of these plants to translocation sites as a likely important contributor to translocation success. The detailed ecological information provided within this thesis substantially increases the ecological knowledgebase for this endangered subspecies, fundamentally increasing the capacity for effective management, and the likelihood of successful translocations in the future.

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To Whom It May Concern, I, Holly Bradley, contributed to the study conceptualization, data collection and analysis, and writing and editing of the following manuscript:

Bradley, H.S., S. Tomlinson, M.D. Craig, A.T. Cross, and P.W. Bateman. 2020. Mitigation translocation as a management tool. Conservation Biology. doi: 10.1111/cobi.13667

I, as a co-author, endorse that this level of contribution by the candidate indicated above is appropriate.

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To Whom It May Concern, I, Holly Bradley, contributed to the study conceptualization, data collection and analysis, and writing and editing of the following manuscript:

Bradley, H.S.; M.D. Craig, S. Tomlinson, A.T. Cross, M. Bamford, and P.W. Bateman. 2021 (Under Review). Predators in a mining landscape: targeting of a behaviourally unique, endangered lizard, Austral Ecology.

I, as a co-author, endorse that this level of contribution by the candidate indicated above is appropriate.

Michael D. Craig ________________________

Sean Tomlinson ________________________

Adam T. Cross ________________________

Mike Bamford ________________________

Philip W. Bateman ________________________

A version of this chapter has now been published on June 9, 2022:

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Austral ECOLOGY A Journal of ecology in the Southern Hemisphere

Austral Ecology (2022) ••, ••–••

Predators in a mining landscape: Threats to a behaviourally unique, endangered lizard

HOLLY S. BRADLEY,[1*] MICHAEL D. CRAIG,[2,3] SEAN TOMLINSON,[4,5,6]
ADAM T. CROSS,[1,7] MICHAEL J. BAMFORD[8] AND PHILIP W. BATEMAN[9]
[1]*ARC Centre for Mine Site Restoration, School of Molecular and Life Sciences, Curtin University, Kent Street, Bentley, Perth, West Australia, 6102, Australia (Email: holly.bradley@postgrad.curtin.edu.au);* [2]*School of Biological Sciences, University of Western Australia, Crawley, West Australia, Australia;* [3]*School of Veterinary and Life Sciences, Murdoch University, Murdoch, West Australia, Australia;* [4]*School of Molecular and Life Sciences, Curtin University, Perth, West Australia, Australia;* [5]*Department of Biodiversity, Conservation and Attractions, Kings Park Science, Perth, West Australia, Australia;* [6]*School of Biological Sciences, University of Adelaide, Adelaide, South Australia, Australia;* [7]*EcoHealth Network: https://ecohealthglobal.org, Brookline, Massachusetts, USA;* [8]*Bamford Consulting Ecologists, Kingsley, West Australia, Australia; and* [9]*Behavioural Ecology Laboratory, School of Molecular and Life Sciences, Curtin University, Perth, West Australia, Australia*

Abstract Patchy resource distribution can cluster predator activity around areas of the highest productivity in ecosystems. For the endangered Western Spiny-tailed Skink (*Egernia stokesii badia*) in Western Australia, the log piles that they permanently inhabit in an otherwise patchy, arid landscape, represent a potentially reliable, high abundance food source for predators. Not only are encounter rates by potential predators of *E. s. badia* likely to be influenced by vegetation structure at the microhabitat scale but also *E. s. badia* occurs in a region where mine sites and associated infrastructure, such as landfill sites, likely concentrate generalist predators (e.g. Feral Cats and corvids). We assessed the influence of the presence of coarse woody debris (CWD) and distance to the landfill on predator behaviour towards *E. s. badia* through plasticine model experiments, unbounded point count bird surveys and camera trapping. We found that CWD inhabited by *E. s. badia* attracted a greater relative activity of corvids compared with uninhabited CWD, or control sites without CWD. The relative activity of corvids and predatory birds combined increased with decreasing distance from the landfill. Preferential hunting by corvids at CWD inhabited by *E. s. badia* compared to both uninhabited CWD and open sites suggests that inhabited CWD may be targeted by generalist predators in the region, and that adaptive management may be required for species conservation around active mining areas.

Key words: *Egernia stokesii*, mitigation translocation, optimal foraging, predation, threatened species management.

Chapter 4: Under review at Scientific Reports. As an author, permission is automatically granted for reuse of the copyright material in my thesis.

To Whom It May Concern, I, Holly Bradley, contributed to the study conceptualization, data collection and analysis, and writing and editing of the following manuscript:

Bradley, H.S.; M.D. Craig, A.T. Cross, S. Tomlinson, M. Bamford, and P.W. Bateman. 2022. Revealing microhabitat requirements of an endangered specialist lizard with LiDAR. Scientific Reports **12**(1), 5293.

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I, as a co-author, endorse that this level of contribution by the candidate indicated above is appropriate.

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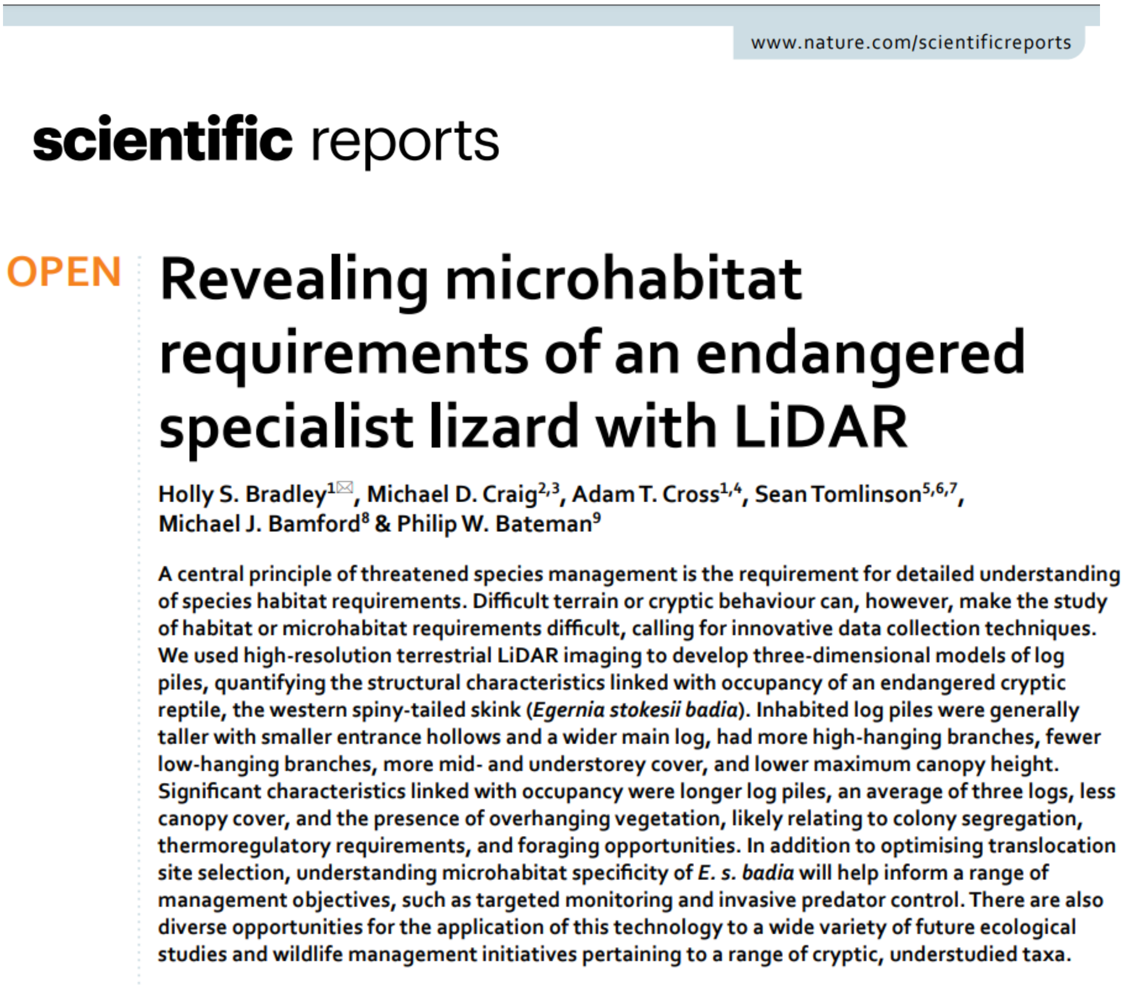

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Revealing microhabitat requirements of an endangered specialist lizard with LiDAR

Holly S. Bradley[1]✉, Michael D. Craig[2,3], Adam T. Cross[1,4], Sean Tomlinson[5,6,7], Michael J. Bamford[8] & Philip W. Bateman[9]

A central principle of threatened species management is the requirement for detailed understanding of species habitat requirements. Difficult terrain or cryptic behaviour can, however, make the study of habitat or microhabitat requirements difficult, calling for innovative data collection techniques. We used high-resolution terrestrial LiDAR imaging to develop three-dimensional models of log piles, quantifying the structural characteristics linked with occupancy of an endangered cryptic reptile, the western spiny-tailed skink (*Egernia stokesii badia*). Inhabited log piles were generally taller with smaller entrance hollows and a wider main log, had more high-hanging branches, fewer low-hanging branches, more mid- and understorey cover, and lower maximum canopy height. Significant characteristics linked with occupancy were longer log piles, an average of three logs, less canopy cover, and the presence of overhanging vegetation, likely relating to colony segregation, thermoregulatory requirements, and foraging opportunities. In addition to optimising translocation site selection, understanding microhabitat specificity of *E. s. badia* will help inform a range of management objectives, such as targeted monitoring and invasive predator control. There are also diverse opportunities for the application of this technology to a wide variety of future ecological studies and wildlife management initiatives pertaining to a range of cryptic, understudied taxa.

Chapter 5: In preparation for publication

To Whom It May Concern, I, Holly Bradley, contributed to the study conceptualization, data collection and analysis, and writing and editing of the following manuscript:

Bradley, H.S.; A.T. Cross, M.D. Craig, S. Tomlinson, M. Bamford, and P.W. Bateman. 2021. Diet plasticity of the Western Spiny-tailed Skink. In preparation.

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Adam T. Cross __________________________

Michael D. Craig __________________________

Sean Tomlinson __________________________

Mike Bamford __________________________

Philip W. Bateman __________________________

Appendix II. Additional Publications

There was one additional article which I was involved with publishing during the PhD that was not included within the main body of the thesis. As an author, I am automatically awarded permission to reproduce the content in this thesis. The article is listed below:

Cross, S.L., **H.S. Bradley**, E.P. Tudor, M.D. Craig, S. Tomlinson, M.J. Bamford, P.W. Bateman, and A.T. Cross. 2021. A life-of-mine approach to fauna monitoring is critical for recovering functional ecosystems to restored landscapes. Restoration Ecology. doi: 10.1111/rec.13540

A life-of-mine approach to fauna monitoring is critical for recovering functional ecosystems to restored landscapes

Running Title: Fauna monitoring for ecosystem recovery

Sophie L. Cross[1,*], Holly S. Bradley[1], Emily P. Tudor[2], Michael D. Craig[3,4], Sean Tomlinson[1,5], Michael J. Bamford[3,6], Philip W. Bateman[2] , Adam T. Cross[1,7]

[1]ARC Centre for Mine Site Restoration, School of Molecular and Life Sciences, Curtin University, Kent Street, Bentley, WA 6102, Australia.

[2]School of Molecular and Life Sciences, Curtin University, Kent Street, Bentley, WA 6102, Australia.

[3]School of Biological Sciences, University of Western Australia, Stirling Highway, Nedlands, WA 6009, Australia.

[4]Environmental and Conservation Sciences, Murdoch University, South Street, Murdoch, WA 6150, Australia.

[5]School of Biological Sciences, University of Adelaide, North Terrace, Adelaide, SA 5000, Australia

[6]Bamford Consulting Ecologists, 23 Plover Way, Kingsley, WA 6026, Australia

[7]EcoHealth Network, 1330 Beacon St, Suite 355a, Brookline, MA 02446, United States. https://ecohealthglobal.org.

*Corresponding author: Sophie.L.Cross@outlook.com

Author contributions: SLC, PWB, ATC, HSB conceived and designed the research; SLC led the writing of the manuscript; SLC, HSB, EPT, MDC, ST, MJB, PWB, ATC contributed to the writing and revisions of the manuscript.

Abstract

Mineral extraction activities are intensely disruptive to ecosystems and their associated fauna. Few countries globally have comprehensive legislation surrounding mine site restoration, but within Australia, restoration of discontinued mine sites is a legislative requirement. However, substantial ambiguity regarding the optimal techniques for restoring biodiverse and functional fauna assemblages remains, and monitoring activities typically focus on vegetation communities despite functioning ecosystems being reliant on key trophic interactions involving fauna. When fauna are considered, monitoring efforts typically yield baseline surveys of species richness and the presence or absence of conservation significant taxa. Even where complete ecosystem recovery is not the goal of post-mining ecological recovery, we argue that there is a critical need for a life-of-mine approach to fauna monitoring underpinned by greater dialogue between researchers, environmental regulators, and the mining industry. Environmental Impact Assessments should include requirements for the consideration of all potential impacts of mining on the structure, behaviour and ecological roles of fauna communities, restoration practices must facilitate the return of functional, resilient, and biodiverse fauna communities to restored post-mining landscapes, and the scope of monitoring practices should be broadened to a holistic examination of fauna communities. Recognising, quantifying, and monitoring the impacts of mining activities and subsequent rehabilitation or restoration on fauna is vital to understanding how anthropogenic disturbances affect natural ecosystems, and in assisting in the successful recovery of ecosystem functionality to areas that have been damaged, degraded, or destroyed.

Key words: Mine; Impact Assessment; Monitoring; Management; Restoration

Implications for practice

- Complete ecosystem recovery relies on key trophic interactions involving fauna
- The assumption of fauna return to restored landscapes may be inadequate for returning fully functional ecosystems
- Restoration must consider the requirements of fauna groups that promote long-term, functional, biodiverse fauna assemblages
- Researchers, environmental regulators, and the mining industry must take a life-of-mine approach to fauna monitoring to assist in the successful recovery of ecosystem functionality to landscapes that have been degraded or destroyed

Background

Australia is one of a handful of countries with comprehensive legislation outlining expectations of mine site rehabilitation and restoration following the discontinuation of mining activities (Clark and Clark 2005). However, there are still few examples of landscapes being successfully returned to functional native ecosystems following mining (Cross et al. 2020a; Carlucci et al. 2020). Notably, despite ecosystem recovery being reliant upon the reassembly of trophic interactions between animals and other components of the ecosystem (Ruiz-Jaen and Mitchell Aide 2005; Fraser et al. 2015; McAlpine et al. 2016), fauna remain broadly overlooked in restoration planning and monitoring (Cross et al. 2019). Our understanding of how fauna communities reassemble and behave in rehabilitated or restored areas in comparison with reference ecosystems remains limited for many taxonomic groups (Cross et al. 2019, 2020a). We argue that there is a critical need for collaboration and discussion between scientists, environmental regulators, industry, and associated restoration practitioners, towards a life-of-mine (i.e., before, during, and after active operations) approach to considering fauna. This approach should be three-pronged; aiming to a) appropriately assess and quantify all potential impacts of mining on fauna during Environmental Impact Assessments, b) mitigate these impacts, where possible, and improve restoration practices to facilitate the return of functional, resilient, and biodiverse fauna communities to restored post-mining landscapes, and c) improve and broaden the scope of monitoring practices to provide more holistic information about community assembly and the behaviour of fauna communities along restoration trajectories. Although we present an Australian focus, we stress that these concepts are globally portable (e.g., Gann et al. 2019).

Improving Environmental Impact Assessments

Although the importance of Environmental Impact Assessments (EIAs) in conservation and management planning is well recognised, concerns have arisen regarding their effectiveness in guiding environmental management best practice (Morgan 2012; Bigard et al. 2017). Current guidelines focus upon baseline surveys of fauna population assemblages, with an emphasis on presence and abundance of conservation significant taxa (e.g., DMIRS 2020). However, EIAs should also provide baseline reference for structure and ecological function of fauna communities within proposed impact zones; assessments currently rarely acknowledge ecological processes and the roles of functional groups in providing critical ecological services (Slootweg and Kolhoff 2003; Frick et al. 2014). Given the critical role of fauna in ecosystems (e.g., Majer 1989; Godínez-Álvarez 2004; Jouquet et al. 2011), a lack of specific consideration of the diverse impacts of mining activities on fauna represents a major weakness in the foundations of mine site restoration right from the planning stage. Regulatory agencies should also include requirements for the assessment of structure and functioning (e.g., degree to which fauna perform integral ecosystem roles; Akçakaya et al 2020) in fauna assemblages, determination of the requirements of species critical to ecosystem functioning, and identification of the ecological processes sustaining viable fauna populations (Slootweg and Kolhoff 2003; Wale and Yalew 2010). Knowledge of the requirements for population support (e.g., baseline resources, key habitats), particularly for specialist species whose ecological requirements may not be met in the restored environment, is crucial to ensuring targeted and adaptive management strategies can be developed and implemented from the outset of mining operations. Pre-approval surveys provide a critical opportunity to lay solid foundations for effective and successful mine site rehabilitation and restoration.

Restoring biodiverse and functional fauna communities

Current regulatory guidance requires the monitoring of specific environmental outcomes and 'performance criteria' relating to fauna throughout active mining operations, for example 'no death of conservation significant native fauna through entrapment in mine facilities' (DMIRS 2020). However, guidance and policy apposite to returning abundant, diverse, and functional fauna communities to mined lands other than these monitoring requirements are lacking (Thompson and Thompson 2020). A significant body of literature exists which could inform the development of guidance and policy in this This article is protected by copyright. All rights reserved. area, and such guidance would greatly assist industry in setting appropriate goals for post-mining restoration and establishing completion criteria (Block et al. 2001; Lindell 2008; Majer 2009; McAlpine et al. 2016). For example, movement ecology of many species is predicated upon landscape-level changes in habitat (e.g., Allen and Singh 2016; Tarszisz et al. 2018; Cross et al. 2020b,c), yet movement ecology and other knowledge of fauna behaviour are rarely considered in restoration planning despite their vital role in facilitating the functional place of animals in ecosystems (Lindell 2008; Cross et al. 2020a). Although behavioural studies are sometimes overlooked due to being perceived as an ineffective financial investment (Blumstein and Berger-Tal 2015; Burger-Tal et al. 2016), such studies are increasingly becoming more financially viable both for monitoring individuals (e.g., Fisher et al. 2018) and population trends (e.g., Wildermuth et al. 2013).

Even where mining industry is under legislative obligation to monitor fauna populations (e.g., conservation significant species), impacts other than habitat loss, such as the effectiveness of prescribed exclusion zones in adequately protecting fauna from the deleterious effects of disturbances (e.g., dust, noise, vibration, and light), are rarely addressed (Raiter et al. 2014; Cross et al. 2021). These factors can create significant ecological, behavioural, and physiological barriers to community reassembly and ecosystem functioning (e.g., Baker and Richardson 2006; Kight and Swaddle 2011;), potentially constraining the success of rehabilitation and restoration efforts. While studies have demonstrated the value of incorporating high-resolution spatial data to guide the habitat requirements of key fauna groups in their restoration and translocation (Tomlinson et al. 2018; Tomlinson et al. 2020; Saleeba et al. 2020), this approach is also yet to be widely incorporated into EIA processes, restoration and closure planning, or other mitigation measures (Bradley et al. 2020). We encourage more dialogue between academics, policy makers and industry to facilitate three-way knowledge sharing in this space - about current scientific theory and best practice relating to fauna ecology, how this can inform policy development that simultaneously provides improved guidance for industry and ensures adequate environmental protection, and how practical, effective and cost-efficient restoration solutions can be developed to assist industry in meeting and exceeding these requirements.

Broadening the scope of fauna monitoring

Assessments of fauna in post-mining restoration typically still favour presence-absence surveys of select taxonomic groups (Cross et al. 2019, 2020a), which provide limited insight into ecosystem functionality, resource availability, and whether restored landscapes might support functional and self-sustaining fauna populations (Aldridge and Boyce 2007; Lindell 2008; Cross et al. 2020a). More holistic assessments of community composition, coupled with study of the physiological and behavioural responses of fauna to changing environmental conditions, provide greater insight into the integrity and resilience of fauna assemblages than presence-absence data alone (Sutherland 1998; Jones et al. 2016; Hale and Swearer 2017; Hale et al. 2020). For example, while restored landscapes may be inhabited by fauna, they often lack key resources necessary for population support (e.g., refuges such as coarse woody debris: Craig et al. 2014; Cross et al. 2020b), which can constrain the self-sustainability of fauna communities by increasing the energetic costs or predation risks associated with the restored landscape (Tomlinson et al. 2014; Cross et al. 2020c). Understanding how animals interact with their environment and respond to environmental change is vital to ensuring the effective return of functional ecosystems representative of pre-disturbance landscapes (Tomlinson et al. 2014). Better understanding these requirements will also assist in reintegrating restored areas into the surrounding landscape, rather than leaving a legacy of discrete, isolated patches of incompletely restored habitat.

Conclusions

Even where complete ecosystem recovery is not the ultimate goal of post-mining ecological recovery initiatives (Gann et al. 2019), efforts to collect, interpret and synthesise ecological data to better inform how fauna are considered in restoration are crucial. A life-of-mine approach to returning fauna to mined landscapes is urgently required, underpinned by a much stronger focus on the composition, structure, and behaviour of fauna assemblages and how these factors are impacted by potential mining disturbances. It is crucial that restoration activities are planned and executed with this information in mind to ensure developing ecosystems support the requirements of all fauna groups, with monitoring assessing whether habitats undergoing restoration are promoting the long-term return of functional, biodiverse fauna assemblages (Lindell 2008; Majer 2009, Cross et al. 2020a). Achieving this aspirational goal will require meaningful engagement and long-term partnership between academia, industry, and regulators, but will yield generational dividends by avoiding a legacy of revegetated landscapes devoid of animal life and supporting the social license of industry to mine.

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