Faculty of Science and Engineering
School of Molecular and Life Sciences

Behavioural Responses of Varanids to Mine Site Restoration

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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 acknowledgment has been made.

This thesis contains no material which has been accepted for the award of any other degree or diploma in any university.

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 ARE2016-2 (Chapters 3-6). Additional approvals were provided by the Western Australian Department Parks and Wildlife, Approval numbers 01-000141-2 and 08-000250-2 (Chapters 3-6).

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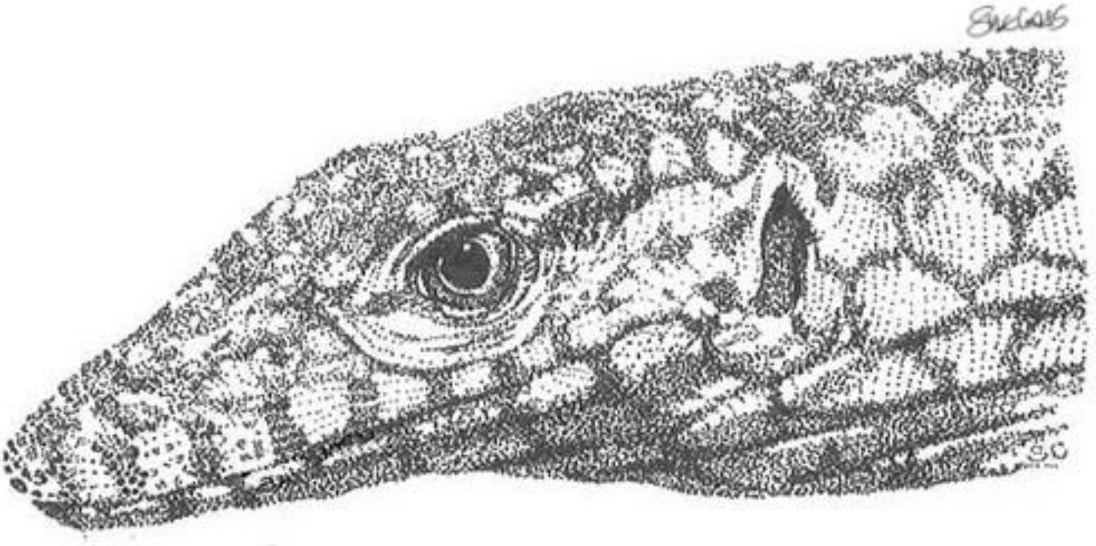

Varanus giganteus

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Statement of Contribution by Others

Chapters 2 to 6 have been prepared as manuscripts for peer-reviewed publication in the scientific literature. These chapters are reproductions of submitted and published manuscripts, with the exception of formatting consistent with the thesis. Signed author statements can be found in Appendix 1. I have obtained permission from the copyright owners to use any third-party copyright material reproduced in this thesis, and to use any of my own published work in which the copyright is held by another party. Permission for paper reproductions in this thesis can be found in Appendix 5.

The study presented in Chapter 2 was published within the peer-reviewed journal, '*Pacific Conservation Biology'* on January 25th, 2019:

> **Cross, S.L.**, Tomlinson, S., Craig, M.D., Dixon, K.W. and Bateman, P.W. (2019). Overlooked and undervalued: the neglected role of fauna and a global bias in ecological restoration assessments. *Pacific Conservation Biology* **25**, 331-341. doi: 10.1071/PC18079.

> All authors conceived the ideas and designed the methodology; I collected and analysed the data; I wrote the manuscript; all authors contributed to the revisions of the manuscript.

The study presented in Chapter 3 was published within the peer-reviewed journal, '*Journal of Zoology'* on November 29th, 2019:

> **Cross, S.L.**, Craig, M.D., Tomlinson, S. and Bateman, P.W. (2019). I don't like crickets, I love them: invertebrates are an important prey source for varanid lizards. *Journal of Zoology*. Online Early. doi: 10.1111/jzo.12750

> All authors conceived the ideas and designed the methodology; I collected the data; I analysed the data with guidance from ST, MDC, and PWB; I wrote the manuscript; all authors contributed to revisions of the manuscript.

The study presented in Chapter 4 is in preparation for submission for publication within the peer-reviewed literature:

Cross, S.L., Craig, M.D., Tomlinson, S. and Bateman, P.W. Build it and some may come: restoration of discontinued mine site initially favours herbivores. *In preparation for submission*.

All authors conceived the ideas and designed methodology; I collected the data; I analysed the data with guidance from MDC, ST, and PWB; I wrote the manuscript; all authors contributed to the revisions of the manuscript.

The study presented in Chapter 5 was accepted for publication in the peer-reviewed journal '*Austral Ecology*' on the 9th of April 2020:

Cross, S.L., Craig, M.D., Tomlinson, S., Dixon, K.W. and Bateman, P.W. Using monitors to monitor ecological restoration: presence may not indicate persistence. ***In Press***.

All authors conceived the ideas, SLC, MDC, ST, and PWB designed the methodology; I collected the data; I analysed the data with guidance from MDC, ST, and PWB**;** I wrote the manuscript; all authors contributed to the revisions of the manuscript.

The study presented in Chapter 6 was published within the peer-reviewed journal, '*Australian Journal of Zoology*' on February 4th, 2020:

Cross, S.L., Tomlinson, S., Craig, M.D. and Bateman P.W. (2020). The Time Local Convex Hull (T-LoCoH) method as a tool for assessing the responses of fauna to habitat change: a case study using the perentie (*Varanus giganteus*: Reptilia: Varanidae). *Australian Journal of Zoology*. Online Early. doi: 10.1071/ZO19040.

All authors conceived the ideas and designed methodology; I collected the data; I analysed the data with guidance from ST, MDC, and PWB; I wrote the manuscript; all authors contributed to the revisions of the manuscript.

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Dedicated to Adam, for his unwavering support and ability to make any situation seem manageable

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Fieldwork was not only physically tough (with a grand total of 1270km bushwalked over the 6 months – the equivalent of walking just over the entire length of Italy), but often mentally gruelling. So, I'll end with some words which can only be described as the perfect PhD mantra and helped spur me on through the long weeks spent living on a remote mine site:

> *"I began to realise how important it was to be an enthusiast in life. If you're interested in something, no matter what it is, go at it full speed ahead. Embrace it with both arms, hug it, love it, and above all become passionate about it. Luke warm is no good. Hot is no good either. White hot and passionate is the only thing to be."*

- Roald Dahl, in *My Uncle Oswald*

‘Petra the Perentie’

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General Abstract

Globally increasing rates of mine site discontinuations require need for immediate implementation of effective conservation and management strategies. Approximately 60,000 mine sites across Australia are discontinued, but the majority have not been restored and closed. Animals play critical roles in ecosystems, for example as ecological engineers, pollinators, and nutrient cyclers; however, they are often overlooked in assessments of restoration success in favour of standardised vegetation surveys. In Australia, varanid lizards provide a critical role as high-order carnivores with distinctly different ecological requirements and capabilities to mammalian carnivores, yet they are very rarely represented in restoration planning or assessment. Among existing studies of animal responses to restoration, there is a strong focus towards studies of species presence, absence, or abundance of select taxa in habitats. While these metrics are useful tools in monitoring animal populations and in identifying key habitats, such studies are restricted in their ability to show whether restoration facilitates long-term return to viable, self-sustaining fauna populations. Understanding the behavioural responses and movement ecology of animals within landscapes undergoing restoration is key to their conservation in the face of increasing rates of habitat destruction.

This thesis presents an ecological study of varanids in the Mid West region of Western Australia, and a behavioural ecology study of varanid responses to the restoration of a mine site in the Mid West region of Western Australia. I used remote sensing camera traps to assess animal communities within restored and reference vegetation, and mapped habitat usage and used GPS/VHF tracking to assess the behavioural and ecological responses of varanids to habitat change and restoration. Restoration of discontinued mine sites appears to facilitate use by animals; however, the community structure of fauna populations and the behaviour and ecology of varanids in these habitats differs to that in the reference, unmined bushland. Early stage habitat restoration appears to be particularly effective for herbivore species, and although habitats undergoing restoration are used by varanids, these areas are used with increased selectivity and altered behaviour. Restoration vegetation may lack some key resources, for example microhabitats and refuges, that are necessary to support

complex, functional animal communities. Providing increased refuges, such as hollow logs, to areas undergoing restoration may aid in facilitating the return of fauna populations, particularly during the early stages of vegetation establishment. Future conservation and management strategies should consider the behaviour and ecology of a wide range of fauna in assessments of restoration progress, to ensure restored habitats are effectively returning self-sustaining fauna populations and functional ecosystems.

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List of Abbreviations

BBMM: Brownian Bridge Movement Model
BHN: Blue Hills North
D: Simpsons Index
DOR: Dead on road
EVB: Exploration Village Bushland
FD: Foot distance
FO: Frequency of occurrence
GPS: Global Positioning System
IVG: Inter-visit gap
KDE: Kernel density estimates
KML: Karara Mining Ltd.
MCP: Minimum Convex Polygon
nMDS: non-metric multidimensional scaling
MNLV: Mean number of locations in the hull per visit
NSV: Number of separate visits
O_{ab}: Dietary overlap
SE: Standard error
SID: Simpsons Index of Diversity
SL: Stride length
SRI: Simpsons Reciprocal Index
SVL: Snout-vent length
T-LoCoH: Time Local Convex Hull
TL: Total length
TP: Terapod
TW: Tail width
VHF: Very High Frequency
WAM: Western Australian Museum

Glossary

Mine discontinuation or abandonment: Termination of active mining, ownership of land is retained but site is inactive.

Mine closure: “A whole-of-mine-life process, which typically culminates in tenement relinquishment. It includes decommissioning and restoration” (DMP and EPA, 2015).

Restoration: “The process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed’ (Clewell *et al.*, 2004; McDonald *et al.*, 2017).

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Chapter 1. General Introduction

Perentie (*Varanus giganteus*)

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1.1 Introduction

Habitat loss, largely driven by anthropogenic impacts such as mining, urbanisation, and agriculture, is a leading cause of biodiversity loss and species extinctions globally (Fahrig, 1997; Lande, 1998; Cristescu *et al.*, 2012). Although mining activities typically have a reduced physical environmental footprint in comparison to other industries, a high proportion of active mining activities in Australia operate on lands considered to be of high conservation value (Miranda *et al.*, 2003; Bridge, 2004; Cross *et al.*, 2019). Mining activities can fundamentally alter ecosystems and often present serious environmental pollution issues (Salomons, 1995; Bian *et al.*, 2009). Over 60,000 mine sites across Australia have been identified as discontinued, yet despite restoration following the discontinuation of mining activities being a legislative requirement (Gilbert, 2000), very few sites have been confirmed as restored as officially closed (Campbell, 2017).

Assessments of restoration success following the cessation of mining activities have historically placed a heavy emphasis on surveying vegetation structure and communities (Ruiz-Jaen and Mitchell Aide, 2005; Koch *et al.*, 2010). Fauna are integral to ecosystem functionality, for example termites and earthworms aid in nutrient cycling, organic decomposition and soil building (Jouquet *et al.*, 2011; Blouin *et al.*, 2013), insects, birds, and even some reptile species assist in pollination and seed dispersal (Valido and Olsen, 2007; Carlo and Morales, 2016; Wenny *et al.*, 2016), and apex and mesopredator species are vital to predator/prey dynamics (Cortéz-Gomez *et al.*, 2015). Despite numerous calls in recent years for an increased focus upon assessments of fauna return to restored landscapes, animals are often assumed to return unassisted to restored landscapes following the re-establishment of vegetation (Palmer, 1997). Furthermore, among the existing literature assessing faunal return to restored landscapes, there is a strong focus towards assessments of species diversity or abundance of a restricted range of taxa (Cross *et al.*, 2019a). Such studies can provide vital information for ecosystem health and habitat quality; however, they are limited in their ability to understand key plant-animal ecological interactions and the long term-functionality of ecosystems (Lindell, 2008; Cross *et al.*, 2019a, 2020). Understanding how a diversity of animals behaviourally respond to, use, and move through restored landscapes is key to determining whether these areas are supporting the return of self-sustaining and functional fauna populations.

Reptile species are threatened globally, largely through habitat loss and anthropogenic impacts (Böhm *et al.*, 2013). Reptiles occur at particularly high diversity and abundance within the arid regions of Australia and are often the most abundant fauna in these environments (Pianka, 1969; Morton and James, 1988; Roll *et al.*, 2017). Despite their prevalence and importance in Australian ecosystems, reptiles are infrequently considered in assessments of mine site restoration success and few studies have assessed their return to such landscapes (Munro *et al.*, 2007; Todd *et al.*, 2010; Cross *et al.*, 2019a). Within arid Australia, varanids (monitor lizards) often fill apex predator niches in the broad absence of large mammalian carnivores (Read and Scoleri, 2015; Cross *et al.*, 2019b). *Varanus* spp. occupy a wide range of habitat niches, and this diversification has resulted in the largest range of body sizes within a single genus of any vertebrate taxa (King and Green, 1993). The diverse range of body sizes, and therefore home ranges, presents varanids as an ideal group to monitor the impacts of habitat change and restoration over relatively large spatial scales.

1.2 Thesis overview

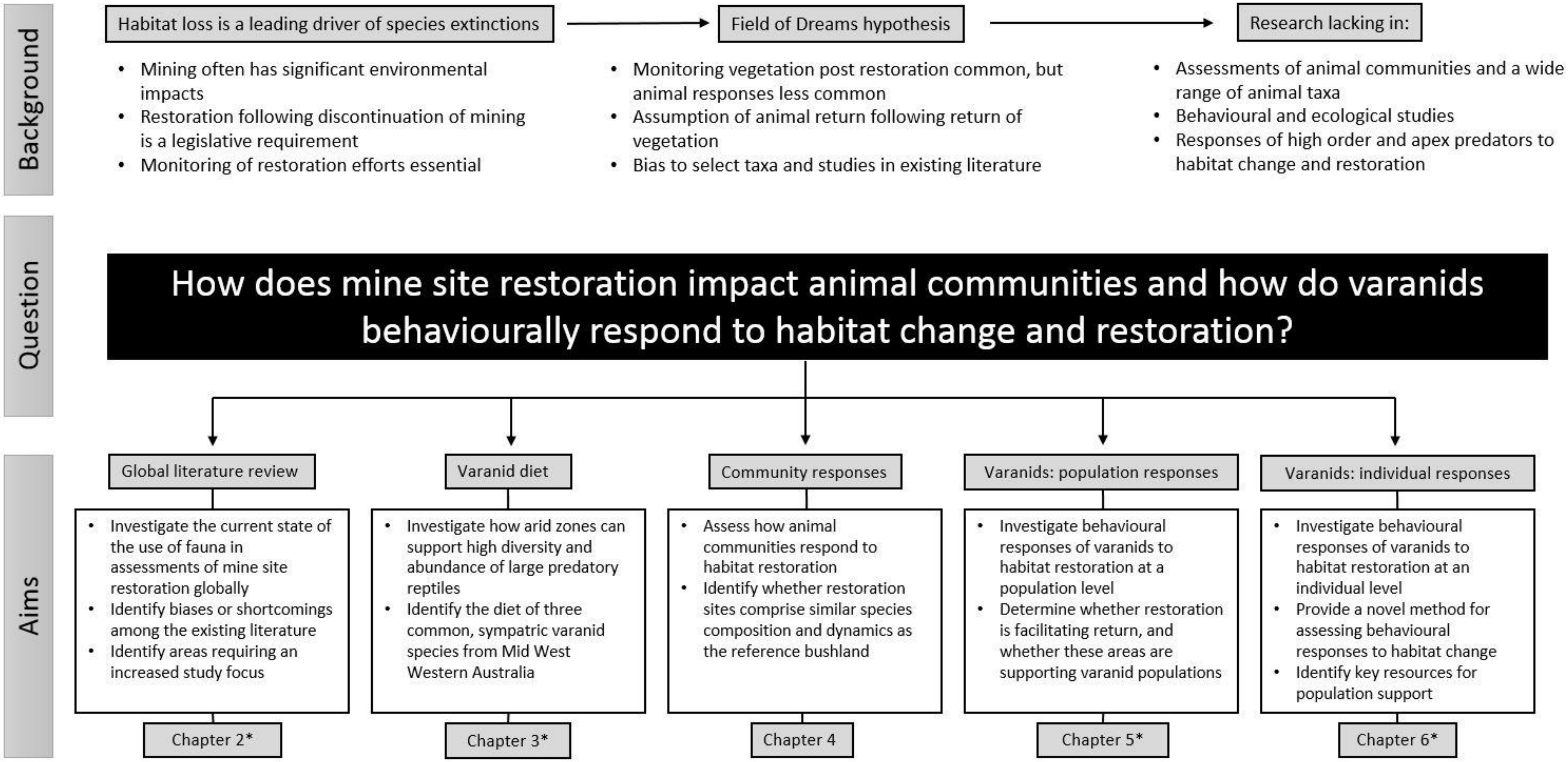
This thesis aims to assess how fauna, particularly large, predatory reptiles, respond to habitat change and restoration following the cessation of mining activities (Fig 1.1). In this thesis, I begin by investigating any potential biases, shortcomings, or knowledge gaps in the existing literature relating to assessments of animal responses to mine site restoration globally (Chapter 2). The literature review presented in Chapter 2 forms the introduction to this thesis. In Chapter 3 I assess the diet of three common *Varanus* species occurring in the arid Mid West region of Western Australia; the black-headed monitor (*Varanus tristis*), Gould's goanna, or racehorse goanna (*Varanus gouldii*), and the yellow-spotted monitor (*Varanus panoptes*), to determine how apex reptilian predators can thrive in low productivity and resource poor habitats, and to examine whether niche partitioning is key to this. Diet is a fundamental component of ecology and understanding an organism's prey items is key to determining how we may be able to facilitate their return to landscapes undergoing restoration, especially in the event of subtle differences between otherwise similar taxa.

In Chapter 4 I begin to investigate the impact of habitat restoration following the discontinuation of mining activities on animal communities. This chapter aims to assess whether restoration sites contain similar species composition and diversity to that within reference habitats. Finally, in Chapters 5 and 6, I assess the movement and behavioural responses of varanids to habitat restoration. Chapter 5 assesses population responses of varanids through indirect assessments of habitat usage, and Chapter 6 assesses individual responses through VHF/GPS tracking. This chapter presents a novel method of assessing the impacts of habitat change and restoration on animals. Finally, in the general discussion (Chapter 7), I synthesise and discuss the conclusions and implications of the research presented in this thesis.

This thesis explores five primary research aims:

1) Identify any shortcomings or biases among the existing published literature relating to assessments of animal responses to mine site restoration, and identify areas requiring further research;
2) Explore the diet of three sympatric, co-existing varanid species in the Mid West region of Western Australia, and determine how populations can thrive in low productivity and challenging environments;
3) Assess how animal communities and foraging guilds differ between reference and restoration vegetation, and the impact of proximity to active mining on animal detection likelihood;
4) Determine how varanid populations respond to habitat restoration and whether habitat use (foraging, movement, or burrowing activity) differs between reference and restoration vegetation; and
5) Investigate how movement and home range data of animals can be used to provide an in-depth analysis of their behavioural responses to habitat change and restoration.

Chapters 2, 3, 5, and 6 have been published within the peer-reviewed literature and Chapter 4 is in preparation for submission to peer review. An additional commentary paper relating to the published literature review presented in Chapter 2 has been published within the journal '*Ecological Management and Restoration*' and has been appended to this thesis.



Habitat loss is a leading driver of species extinctions

- Mining often has significant environmental impacts
- Restoration following discontinuation of mining is a legislative requirement
- Monitoring of restoration efforts essential

Field of Dreams hypothesis

- Monitoring vegetation post restoration common, but animal responses less common
- Assumption of animal return following return of vegetation
- Bias to select taxa and studies in existing literature

Research lacking in:

- Assessments of animal communities and a wide range of animal taxa
- Behavioural and ecological studies
- Responses of high order and apex predators to habitat change and restoration

How does mine site restoration impact animal communities and how do varanids behaviourally respond to habitat change and restoration?

Global literature review

- Investigate the current state of the use of fauna in assessments of mine site restoration globally
- Identify biases or shortcomings among the existing literature
- Identify areas requiring an increased study focus

Chapter 2*

Varanid diet

- Investigate how arid zones can support high diversity and abundance of large predatory reptiles
- Identify the diet of three common, sympatric varanid species from Mid West Western Australia

Chapter 3*

Community responses

- Assess how animal communities respond to habitat restoration
- Identify whether restoration sites comprise similar species composition and dynamics as the reference bushland

Chapter 4

Varanids: population responses

- Investigate behavioural responses of varanids to habitat restoration at a population level
- Determine whether restoration is facilitating return, and whether these areas are supporting varanid populations

Chapter 5*

Varanids: individual responses

- Investigate behavioural responses of varanids to habitat restoration at an individual level
- Provide a novel method for assessing behavioural responses to habitat change
- Identify key resources for population support

Chapter 6*

Fig. 1.1: Conceptual framework of the research question and aims of this thesis. *Published papers

1.3 Study area and species

The research presented in this thesis was conducted at Karara Mining Ltd. in the Mid West region of Western Australia, approximately 415km northeast of Perth (29°11'31"S, 116°45'36"E). The study region experiences an arid climate with an average yearly rainfall of ~300mm (Bureau of Meteorology, http://www.bom.gov.au/climate/data/). We selected three sites of varying proximity to the active mining operation within the Karara region: a reference vegetation site located 3km from the active mine pit (Exploration Village Bushland), and two restoration sites with adjacent reference vegetation located 8km (Blue Hills North) and 12km (Terapod) from the active mining operation (Fig. 1.2). Restoration sites were characterised by a restored waste rock dump (~800 x 500m) surrounded by reference (unmined) vegetation (Fig. 1.3). Vegetation within the region largely comprised *Acacia* shrublands and open *Eucalyptus* woodlands, with restoration sites comprising species present in the reference habitat but at varying stages of establishment. Although I did not specifically collect microhabitat data (e.g., log pile and refuge densities), restoration sites were characterised by increased spatial homogeneity and a reduction in refuge areas to the reference bushland. However, refuge piles composed of piles of sand and woody debris were incorporated into sites at the base of the restored waste rock dump along the edge of the restoration footprint (~400m length, 3m width, 3m height).

Five sympatric *Varanus* species of a range of body sizes co-exist within the Mid West region of Western Australia (Fig. 1.4): the stripe-tailed monitor (*Varanus caudolineatus*, arboreal, total length [TL] 32cm), black-headed monitor (*Varanus tristis*, primarily arboreal, TL 76cm), Gould’s monitor (*Varanus gouldii*, primarily terrestrial, TL 1.2m), yellow-spotted monitor (*Varanus panoptes*, terrestrial, TL 1.4m), and the perentie (*Varanus giganteus*, terrestrial, 2.5m) (Wilson and Swan, 2003; Pianka *et al.*, 2004).

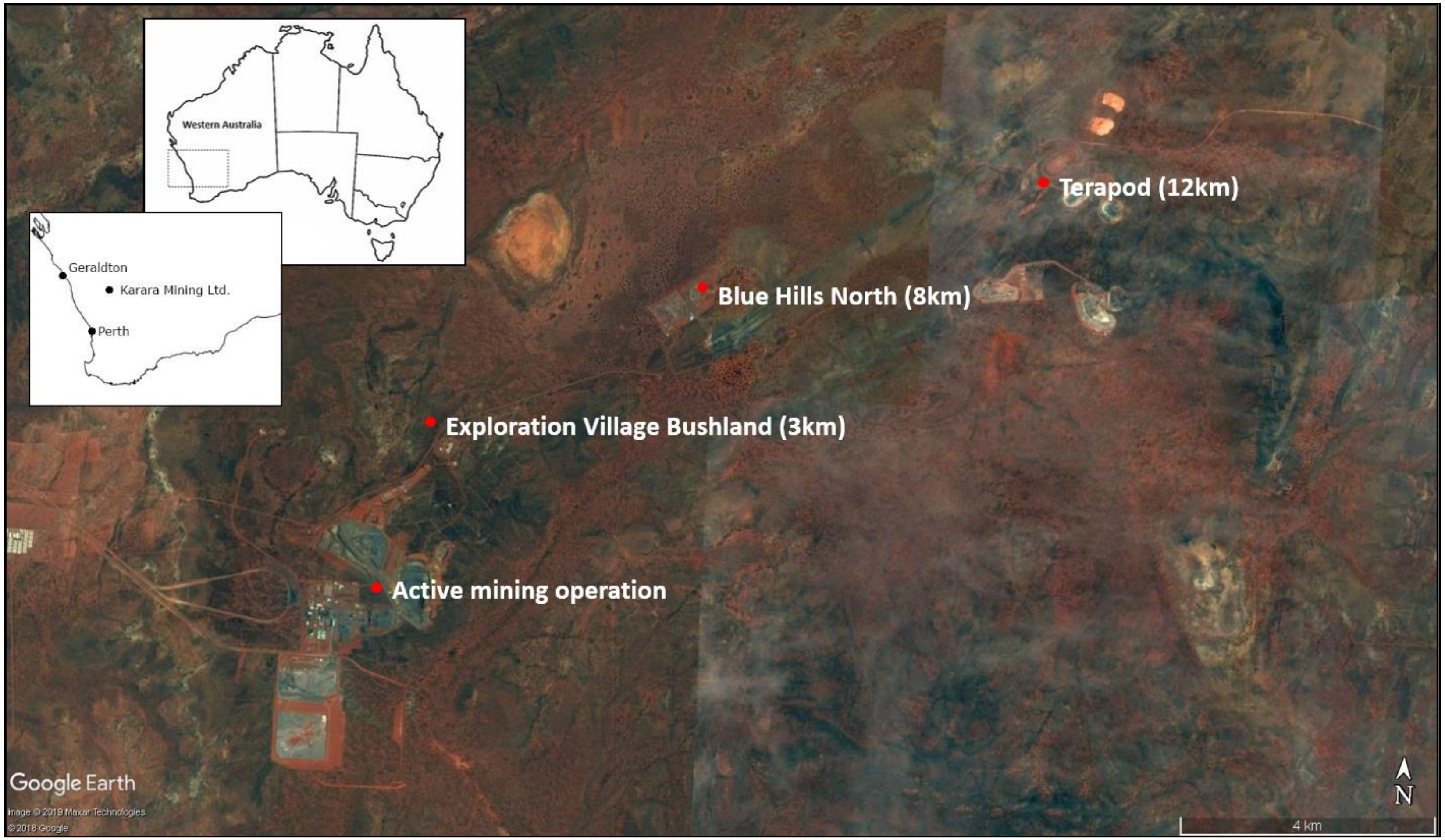


Fig. 1.2: Location of study sites with reference to the active mining operation within the mining tenement of Karara Mining Ltd. in the Mid West region of Western Australia.

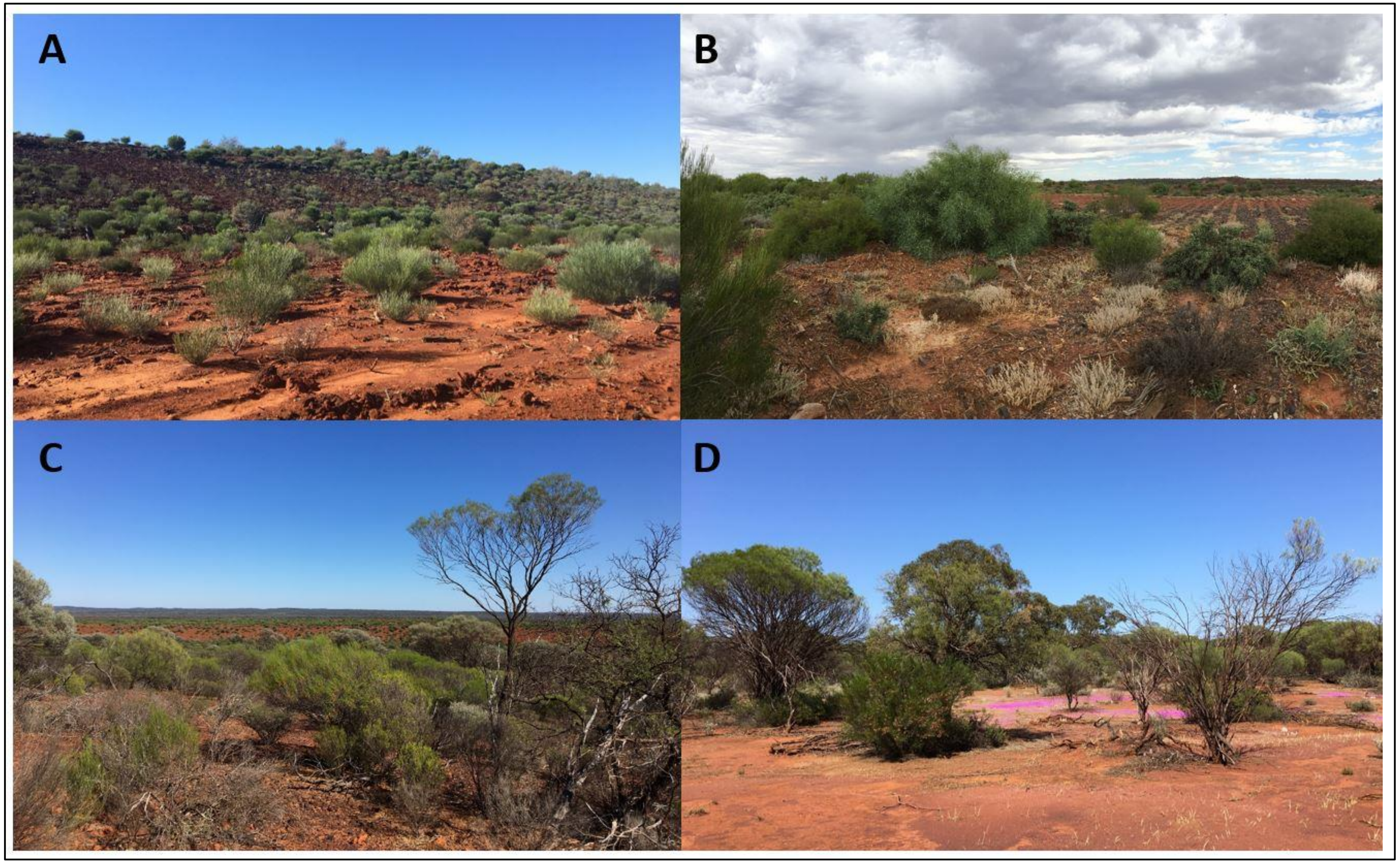


Fig. 1.3: Typical vegetation structure in study sites: A) across the base and slopes of the restored waste rock dump, B) over the top of the restored waste rock dump, C) reference shrubland communities, and D) open woodland communities in the reference vegetation.

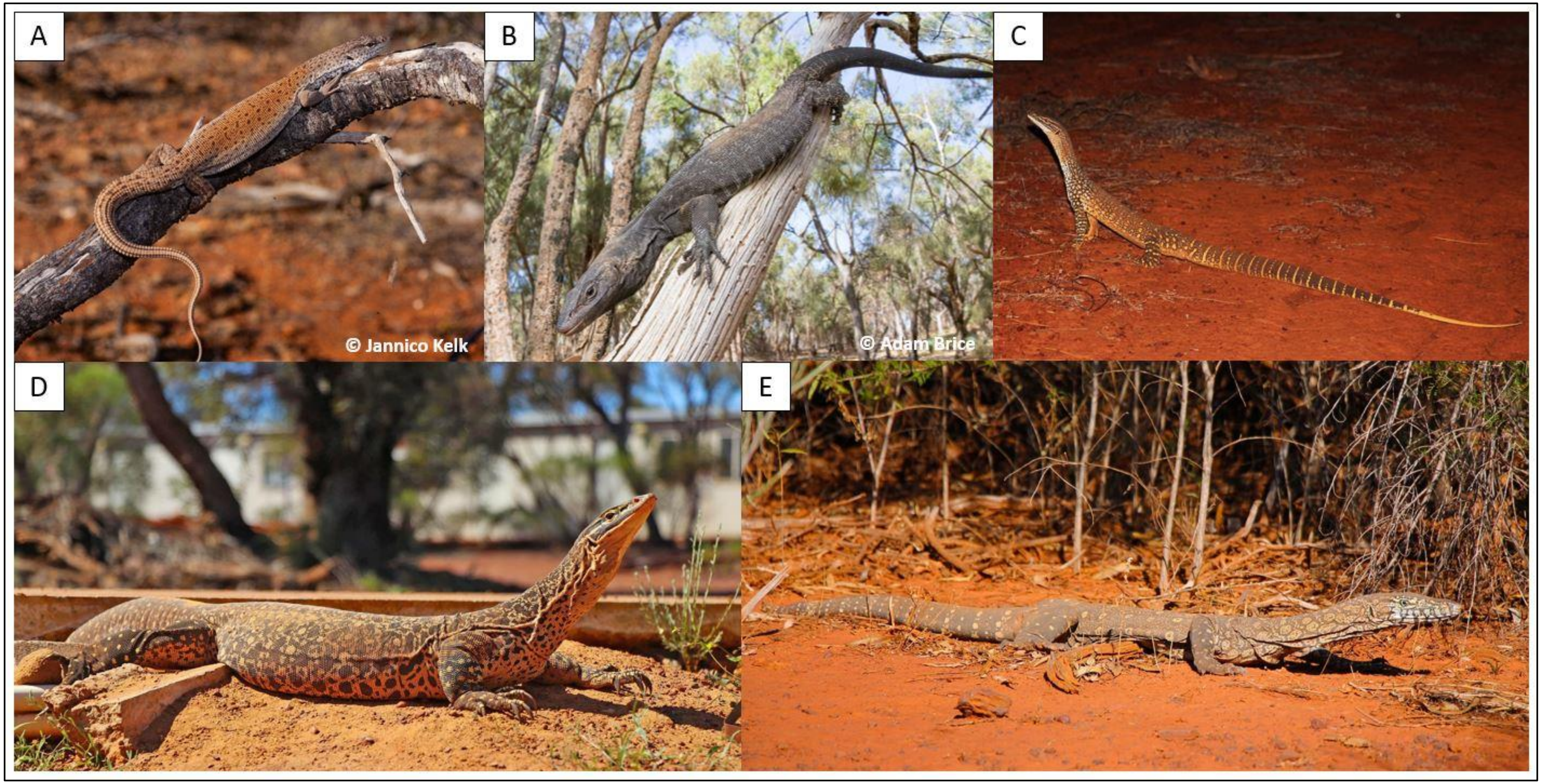


Fig. 1.4: *Varanus* species occurring within the Karara area: A) *Varanus caudolineatus*, B) *V. tristis*, C) *V. gouldii*, D) *V. panoptes*, and E) *V. giganteus.* Image credits: (A) Jannico Kelk; (B) Adam Brice; (C-E) Sophie Cross.

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Every reasonable effort has been made to acknowledge the owners of the copyright material. I would be pleased to hear from any copyright owner who has been omitted or incorrectly acknowledged.

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Chapter 2. Overlooked and undervalued: the neglected role of fauna and a global bias in ecological restoration assessments

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Pygmy spiny-tailed skink (*Egernia depressa*)

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2.1 Abstract

Globally increasing rates of mine site discontinuations are resulting in the need for immediate implementation of effective conservation and management strategies. Surveying vegetation structure is a common method of assessing restoration success; however, responses of fauna to mine site restoration remain largely overlooked and understudied despite their importance within ecosystems as ecological engineers, pollinators, and restoration facilitators. Here we review the current state of the use of fauna in assessments of mine site restoration success globally, and address biases or shortcomings that indicate the assessment approach may undershoot closure and restoration success. We identified just 101 peer-reviewed publications or book chapters over a 49-year period that assess responses of fauna to mine site restoration globally. Most studies originate in Australia, with an emphasis on just one company. Assessments favour general species diversity and richness, with a particular focus on invertebrate responses to mine site restoration. Noteworthy issues included biases towards origin of study, study type, and target taxa. Further searches of the grey literature relating to fauna monitoring in mine site restoration, which was far more difficult to access, yielded six monitoring/guidance documents, three conference proceedings, two book chapters without empirical data, and a bulletin. As with peer-reviewed publications, grey literature focussed on invertebrate responses to restoration, or mentioned fauna only at the most basic level. We emphasise the need for global re-evaluation of regulatory standards to address these major limitations in assessing the capacity of the mining industry to comprehensively and representatively restore faunal communities after mining.

2.2 Introduction

Habitat destruction and fragmentation are primary drivers of biodiversity loss and extinctions worldwide, and the effects of these are being increasingly exacerbated through human activities such as mining, agriculture, forestry and urbanisation (Fahrig, 1997; Lande, 1998; Tilman *et al.*, 2001; Cristescu *et al.*, 2012). While the physical environmental footprint of mining operations is <1% of terrestrial landscape areas, and relatively concentrated in comparison to other industries, e.g. agriculture and urbanisation, which account for 70% and 3% of global land disturbances, respectively (Hodges, 1995; Bridge, 2004; McKinney, 2006), mining often has a substantial local, and often regional, environmental impact (Salomons, 1995; Rybicka, 1996). Activities from mining can fundamentally alter relatively intact and undisturbed habitats into inhospitable land matrices, and can create serious environmental pollution issues such as tailings leakage, dust, and hydrological change (Salomons, 1995; Bian *et al.*, 2009). Though mining activities impact a small terrestrial footprint, 75% of active sites are situated on land considered to be of high conservation value (Miranda *et al.*, 2003; Bridge, 2004). Hence, although environmental impacts of mineral extraction may be restricted in spatial extent, they are intensely disruptive to ecosystems that are often uncommon and fragile. The resultant alteration and degradation from mining activities present some of the most difficult landscapes to restore. As such, lessons learned from the restoration of mine sites may be transferrable to land restoration practices in other areas of high conservation value that have suffered other forms of degrading processes.

Many different environmental components (e.g., soil, plants, microorganisms, and fauna) require study in assessments of ecosystem health and functionality (Duffy, 2003); yet restoration monitoring is typically restricted to plant communities and vegetation structure, which remain a key priority in assessing postmining restoration success (Ruiz-Jaen and Mitchell Aide, 2005; Koch *et al.*, 2010). Majer (1989) highlights this issue; however, the disparity between fauna and plant studies remains a key issue. This is despite fauna being essential to restoration success, and playing critical roles in the provision of numerous essential ecosystem functions, such as seed dispersal, pollination, nutrient cycling, and soil formation (Majer, 1989; Lavelle *et al.*, 2006; Mace *et al.*, 2012). Importantly, fauna, due to their mobility, often rely on spatial scales far greater than plants, and hence are often dependant on habitats and resources that occur both within and outside the restoration patch. However, responses of fauna are often overlooked in favour of standardised vegetation surveys, which typically can be achieved rapidly and follow established principles (Ruiz-Jaen and Mitchell Aide, 2005). Fauna are often assumed to return to pre-disturbance diversity and abundances following the return of vegetation (Block *et al.*, 2001; Cristescu *et al.*, 2012) through what is commonly referred to as the ‘Field of Dreams’ Hypothesis (‘build it and they will come’: Palmer *et al.*, 1997). In practice, recovering animal biodiversity and community structure are some of the most difficult components to understand, achieve, and assess following the restoration of degraded sites (Cristescu *et al.*, 2012; Perring *et al.*, 2015).

Faunal responses to mine site restoration require study across a wide range of habitats and climatic regions to maximise biodiversity outcomes. Biases to certain regions or mineral extraction types limit our ability to inform on best practices for restoring ecosystem function by preconditioning our expectations to outcomes that may be unique to some places or disturbance patterns. Surface (e.g., strip mining, open pit, and quarry) and subsurface (underground) mining have varying levels of physical environmental impact (Dudka and Adriano, 1997). Underground mining can have significant impacts on subsurface hydrology and soil structure (Altun *et al.*, 2010); however, the above-ground impact (other than infrastructure and tailings or waste rock dumps) of underground mining is of a lower magnitude by comparison to the often very large terrestrial footprints of surface mining (Lin *et al.*, 2005). Hence, conclusions drawn from sites of only one extraction type may not be best suited to inform restoration practices for other mining techniques.

Faunal responses to mine site restoration also require studies across varying climatic regions. Many of the world’s 35 global biodiversity hotspots are situated within the tropics (Mittermeier *et al.*, 2011). These regions contain higher proportions of endemic species than areas outside the hotspots (Myers *et al.*, 2000). Endemic species, by virtue of occupying one or few specialised habitats, are likely to be affected more severely by habitat fragmentation and loss than generalist species, increasing the difficulty associated with restoring biodiversity values and potentially ecosystem functioning (Ewers and Didham, 2006). Furthermore, while iron ore extraction from ultramafic soils takes place in biodiverse landscapes in, for example, Brazil, New Caledonia and Australia, it seems unlikely that the best practices of ecological restoration developed in Australia, with its unique flora and fauna and ancient, arid landscapes (Hopper and Gioia, 2004; Hopper, 2009), would translate well to the different tropical ecosystems of an island in the Pacific, or the rainforests of South America to improve restoration practices and biodiversity conservation.

Although a higher focus is being placed on fauna assessments in restoration in recent years (Majer, 2009), of the limited studies that assess animal responses to restoration (particularly in relation to mine site restoration), there is a strong emphasis evident towards the use of certain taxa as biological indicators (bioindicators); for example, ants and birds, both of which typically can be easily surveyed with minimal time and financial investments (Majer, 1983; Andersen *et al.*, 2003; Nichols and Nichols, 2003; Gould and Mackey, 2015). The use of bioindicators has remained a favoured method of assessing environmental health, since the introduction of the concept by Hall and Grinnell (1919). While invertebrates are highly important in ecosystems, and can provide essential information in assessments of environmental health (Majer *et al.*, 2007), basing restoration practices on responses of only ants and common bioindicators may under-represent other groups or negatively affect overall ecosystem development. For restoration efforts to be effective for all faunal groups, assessments for restoration success must be derived from a wider range of fauna, and from their role in the ecosystem, rather than ease of survey effort.

Studies assessing faunal responses to restoration typically favour assessments of species richness and abundance, likely due to reliability and ease of implementation. However, species diversity assessments have several limitations, namely that there is a high probability of missing rare, cryptic, migratory, or seasonally active species, and in the potential for species diversity to be altered through the detection of invasive or cosmopolitan species (Hejda *et al.*, 2009; Chiarucci *et al.*, 2011). Fauna that are capable of dispersing large distances may present a false representation of utilisation of restoration areas, as these areas may only be used opportunistically or transiently and incapable of supporting resident fauna communities in the long term. Isolated assessments of species presence or absence, or diversity, may therefore provide relatively little information as to the functional success of restoration. Studies based primarily on presence or absence do not allow for evaluation of resource use and use of wider restoration landscapes, and hence provide an inaccurate assessment of restoration trajectory and success. Integrative ecological and behavioural studies remain an emerging branch of conservation biology, and might provide an increased understanding of what constitutes a return to a fully restored site. Globally, little is known of how human disturbances alter the behaviour and ecology of fauna that persist in disturbed landscapes, such as postmining environments. Ecological and behavioural studies require significant time investment, and often have higher associated risks and costs than more general species diversity assessments, in terms of the ease of data collection. However, studies of ecology are essential, as behavioural characteristics are the most flexible of faunal adaptations to their environment, and have differing responses to environmental changes (Wolf and Weissing, 2012).

This review assesses the current state of knowledge of the use of fauna in assessments of mine site restoration success. While Cristescu *et al.* (2012) published a review on the use of fauna in assessments of mining restoration success (termed rehabilitation), they primarily assessed the empirical data on faunal recolonisation of mine sites within Australia, whereas we identify and address any potential biases or patterns within literature assessing faunal responses to mine site restoration on a global scale. Specifically, we assess patterns in origin and year of study, targeted taxa, study type (i.e. presence or absence, or species diversity and abundances), and terminology use. We also seek to extend a similar interrogation to the grey literature surrounding faunal monitoring in mine site restoration. Understanding and addressing the current knowledge gaps in mine site restoration literature allows for the identification of areas requiring an increased study focus, and is integral to implementing the 'International Standards for the Practice of Ecological Restoration' (McDonald *et al.*, 2016).

2.3 Methods

We compiled a comprehensive database of peer-reviewed literature composed of studies relating to any use of fauna (invertebrate or vertebrate) in assessments of mining restoration success. Studies were not limited to those using the terminology 'restoration', but included those describing attempted return of vegetation (unassisted natural regeneration or otherwise) following cessation of mining. Mining restoration literature encompasses a wide range of terminologies for describing various restoration practices (Kaźmierczak *et al.*, 2017; Cross *et al.*, 2018). For the purposes of this review, we use ‘restoration’ (adopted terminology in McDonald *et al.*, 2016), which we define as ‘the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed’ (Clewell *et al.*, 2004; McDonald *et al.*, 2016). Literature assessing faunal responses to mining without reference to any form of restoration were discounted. We used three databases to interrogate the literature: Google Scholar, Web of Science (all databases, 1950 – 2018), and Scopus (all documents including secondary documents, all years; last searched November 2018). Additional sources were gleaned from bibliographies in the published literature.

Search terms comprised any combination of ‘Australasia’, ‘Africa’, ‘North America’, ‘South America’, ‘Asia’, or ‘Europe’, AND/OR ‘animal’, ‘fauna’, ‘bird’, ‘reptile’, ‘mammal’, ‘vertebrate’, or ‘invertebrate’ AND ‘response’, or ‘behaviour’ AND ‘mine’, or ‘mining’ AND ‘restoration’, ‘rehabilitation’, ‘reclamation’, ‘recultivation’, ‘afforestation’, or ‘regeneration’. Publications were compiled into a database and sorted based on date of publication, country of origin, target taxa, type of mineral mined, terminology used, and key search terms. The literature comprised 101 publications. As postmining recovery may not be fully represented in the primary literature, we extracted the grey literature from searches and compiled these into a separate database. Grey literature included unpublished data, articles without empirical data, governmental reports, conference proceedings, and bulletins (summarised in Tables A2.1, A2.2 in Appendix 2). Analyses were designed to assess the current state of research in assessments of faunal responses to mining restoration, and identify potential knowledge gaps or biases. Although our aim was to interrogate the grey literature in a similar fashion to the peer-reviewed work, our analyses were insupportable due to the paucity of accessible or relevant data.

First, we identified the number of studies from each individual mine site, allowing for the detection of any potential overlaps or biases to particular sites and type of mineral mined. We then grouped studies based on country of origin and year of publication. Third, we identified the main terminology (the primary term used if multiple terms were present) to assess whether there was a standardised approach to terminology. Lastly, we investigated correlations between location, date of publication, and type of study, with use of particular taxa and type of mineral operation. We identified the following seven variables: (1) mineral type: coal (including publications listing the term ‘lignite’), bauxite, sand, bentonite, gold, iron ore, limestone, tin, uranium, peat, multiple (polymetallic mines, or mines where two or more mineral types were listed), and not stated; (2) taxon group: vertebrate, invertebrate, or both; (3) target clade: Mammalia, Aves, Reptilia, Amphibia, Insecta, Clitellata (a taxon of annelid worm), or multiple targets; (4) main terminology; (5) date of publication; (6) country of origin; and (7) study type: ecology (pollination, density/ biomass, predation), presence/absence, or population abundance of fauna species, and translocations.

Pearson’s Chi-square tests were undertaken to compare differences between all categorical variables. All statistical analyses were conducted in the R 3.4.4 statistical environment (R Core Team 2016), implemented using RStudio (RStudio Inc., Boston, United States, 2018). The results from literature searches have been visualised in a PRISMA 2009 flow diagram (Fig. A2.1, Appendix 2).

2.4 Results

Searches of peer-reviewed, published literature yielded a total of 101 publications from 10 different mineral type operations. Grey literature searches yielded just 12 readily accessible documents, eight of which made direct reference to fauna or fauna monitoring in restoration landscapes. Of the published literature, six studies were based at mines extracting multiple minerals, and five studies did not state the mineral type. Studies predominantly focused on bauxite (*n* = 34), coal (*n* = 26), and mineral sand mines (*n* =19). Two studies each were from limestone, uranium, gold, and peat mines/quarries, and one each from bentonite, iron ore, and tin mines. Many of these minerals are typically extracted through surface mining, with the exception of coal and gold (both surface and subsurface mining), and uranium (subsurface mining). Terminology varied considerably between publications, with a total of seven different terms used: ‘rehabilitation’, ‘restoration’, ‘regeneration’, ‘reclamation’, ‘recultivation’, ‘revegetation’, and ‘afforestation’. Of the 101 publications, 73 used a single terminology to describe restoration activity and 28 mixed terms within the same publication. The countries of origin comprised 14 countries (Australia, United States, Germany, Brazil, Hungary, Spain, South Africa, New Zealand, Czech Republic, United Kingdom, Canada, Colombia, Indonesia, and Italy), two of which are listed in the top five mineral-producing (by metric ton) countries (Fig. A2.2a, b, Appendix 2). Indonesia, Colombia, Brazil, and Australia are listed in the top five megadiverse countries, ranked 1 to 4, respectively (Fig. A2.2c).

Invertebrate responses to mining restoration were assessed in 60 publications; 39 publications assessed vertebrate responses, and two papers assessed both invertebrate and vertebrate responses. Invertebrate studies favoured assessments for insects (90%), and vertebrate studies typically favoured assessments of birds (46%). Studies were significantly more likely to involve assessments for species diversity and abundance (75%, $\chi 2 = 309.5$, $P < 0.001$) compared with those including ecology (including pollination, density/biomass, and predation studies; 18%), presence, absence or population abundance of individual species (6%), or translocations (1%).

2.4.1 Terminology

‘Rehabilitation’ was the most commonly used main term (primary terminology used within the publication; *n* = 47), followed by ‘restoration’ (*n* = 21), ‘regeneration’ (*n* = 10), ‘reclamation’ (*n* = 8), ‘recultivation’ (*n* = 7), ‘revegetation’ (*n* = 4), and ‘afforestation’ (*n* = 3). The main terminology of one study (either ‘restoration’ or ‘reclamation’) could not be ascertained with certainty (Table A2.3, Appendix 2). Use of terminology appeared to be, in part, associated with publication date. While ‘rehabilitation’ had been in consistent use across the range of publication dates (1978 to 2017), ‘restoration’ appeared to be the favoured term within the last decade. Other terminologies do not appear to be in widespread use. European studies had the widest range of terminology (all terminologies apart from ‘regeneration’: Table 2.1). The use of ‘afforestation’ and ‘recultivation’ were exclusively restricted to European studies, and ‘reclamation’ was limited primarily to European and North American studies, with one use in an Australasian study.

Table 2.1: Use of terminology across literature by region.

Region	Terminology	Number of uses
Africa	Rehabilitation	7
	Regeneration	1
Asia	Restoration	1
Australasia	Rehabilitation	34
	Restoration	13
	Regeneration	8
	Revegetation	2
	Reclamation	1
Europe	Recultivation	7
	Restoration	3
	Reclamation	3
	Afforestation	2
	Revegetation	1
	Rehabilitation	1
North America	Reclamation	5
	Restoration	2
	Revegetation	1
	Regeneration	1
South America	Rehabilitation	3
	Restoration	2

2.4.2 Origin and date of study

Studies of fauna in mining restoration were significantly more likely to originate within Australasia than any other region (59%, $\chi2 = 293.41$, $P < 0.001$). While there is a major Australian bias in the literature, 28 of the 60 Australian studies arise from a single organisation: Alcoa of Australia (hereafter Alcoa), which has extensively reported the role of fauna in the restoration of its bauxite operations in the jarrah forests of southwest Australia. These reports account for 82% of studies of bauxite mines globally (n = 28 of 34), and this pattern is the global norm: many studies within mineral categories result from a single mine site. All eight studies within South Africa are from the same locality (Richards Bay), with similar trends among other countries including Germany (n = 2 of 7, Berzdorf lignite mining district, eastern Germany), Czech Republic (n = 3 of 3, north/northwest Bohmia), Hungary (n = 3, Pécs, southern Hungary), and New Zealand (n = 2 of 2, Wangaloa coal mine, Otago). Publication output increased over time; however, study focus appeared to shift from invertebrate to vertebrate species within the last decade (Fig. 2.1). It is noteworthy that output between any given time bracket is not high within this research area, with a peak rate of less than two papers published annually in the years between 2001 and 2010 (Fig. 2.1).

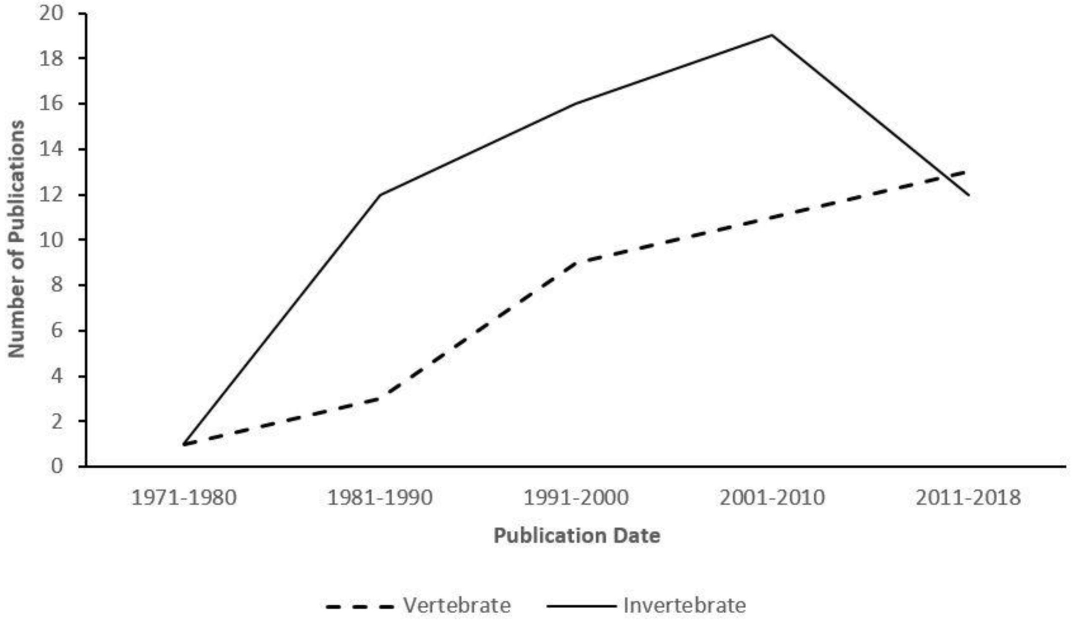

Fig. 2.1: Publication output for vertebrate and invertebrate responses to mine site restoration studies

2.4.3 Invertebrate responses

Invertebrate responses to mine site restoration were reported in 60 publications (comprising over half (59%) of the literature). Invertebrate studies included species from three phyla (Arthropoda, Annelida, and Mollusca), with a particular focus on the Arthropoda (Insecta; *n* = 54 of 60). Excluding those assessing multiple groups, studies primarily assessed responses of the Formicidae (ants; *n* = 19), followed by the Coleoptera (beetles; *n* = 7), Collembola (springtails; *n* = 4), Araneae (spiders; *n* = 3), Diplopoda (millipedes; *n* = 2), Lepidoptera (butterflies; *n* = 2), Oligochaeta (earthworms; *n* = 2), and Hemiptera (true bugs; *n* = 1). Twenty studies did not have a focal group and assessed general species diversity and richness for multiple groups. Studies within Australasia and Europe had the widest range of targeted taxa (Table 2.2). Excluding assessments for multiple invertebrate groups, ants were the most commonly assessed group across almost all mineral types ($\chi2 = 49.6$, $P < 0.001$). Of the eight stated mineral operation types (excluding sites listed as ‘multiple minerals', or 'not stated'), only three had studies examining more than one invertebrate class (bauxite, coal, and sand mines).

Table 2.2: Summary of target class by mineral type and region for invertebrate studies.

Region	Target	Mineral Type
Africa	Multiple invertebrates (2*)	Multiple minerals
	Coleoptera (2)	Sand (1), not stated (1)
	Diplopoda (2)	Multiple minerals
Australasia	Formicidae (14)	Bauxite (5), coal (2), sand (4), uranium (2), iron ore (1)
	Multiple invertebrates (6)	Bauxite (3), coal (1), sand (2)
	Coleoptera (2)	Peat (1), coal (1)
	Araneae (2)	Bauxite
	Collembola (2)	Bauxite
	Hemiptera (1)	Bauxite
Europe	Multiple invertebrates (8)	Coal (6), limestone (2)
	Coleoptera (3)	Coal
	Formicidae (2)	Coal (1), not stated (1)
	Collembola (1)	Coal
	Oligochaeta (1)	Coal
North America	Multiple Invertebrates (3)	Bentonite (1), peat (1), coal (1)
	Lepidoptera (2)	Coal
	Oligochaeta (1)	Bauxite
South America	Formicidae (3)	Bauxite (1), coal (1), gold (1)
	Collembola (1)	Sand

*Denotes number of studies for each target or mineral type.

2.4.4 Vertebrate responses

Studies of vertebrate responses to mining restoration comprised less than half of the total number of publications (*n* = 39 of 101, discounting two studies that assessed both invertebrate and vertebrate responses). Studies significantly favoured the use of birds (45%, $\chi 2 = 19.846$, *P* <0.001) followed by reptiles (18%, *n* = 7), mammals (18%, *n* = 7), and amphibians (3%, *n* = 1). Seven studies assessed responses of multiple groups. Of the 39 vertebrate studies only 12 had specific target species, with the other 27 assessing general species diversity and richness. Vertebrate studies primarily originated from Australasia (*n* = 30), with just three based in each of Europe and North America, and one each in South America, Africa, and Asia. Studies originating outside of Australasia almost exclusively assessed responses of birds, with the exception of three studies (one each in North America, Europe, and Africa) that targeted a combination of mammal, reptile, and amphibian species (Table 2.3). The type of mineral extracted at sites assessing vertebrate responses to mine site restoration appears to be associated with the region of study. Studies of vertebrate responses at bauxite and sand mines occur exclusively within Australasia, whereas those at coal mines are based either in North America or Europe (Table 2.3).

Table 2.3: Summary of target taxa by mineral type and continent for vertebrate studies.

Region	Class	Target	Mineral Type
Australasia	Aves	Cockatoo (*Calyptorhynchus sp.*; 2*)	Multiple minerals (1), bauxite (1)
		Multiple targets (10)	Bauxite (8), sand (2)
	Mammalia	Swamp wallaby (*Wallabia bicolor*; 1)	Sand
		Koala (*Phascolarctos cinereus*; 1)	Sand
		Mouse (*Mus sp.*; 1)	Sand
		Multiple targets (2)	Sand
		Bat (*Chiroptera sp.*; 1)	Bauxite
	Reptilia	South-Western crevice skink (*Egernia napoleonis*; 1)	Bauxite
		Bearded dragon (*Pogona minor*; 2)	Bauxite (1), not stated (1)
		Multiple targets (4)	Bauxite (1), sand (3)
	Amphibia	Multiple targets (1)	Sand
	Mammalia, Reptilia	Multiple targets (2)	Bauxite (1), gold (1)
	Mammalia, Reptilia, Amphibia	Multiple targets (1)	Bauxite
	Reptilia, Amphibia	Multiple targets (1)	Bauxite

North America	Aves	Greater sage-grouse (*Centrocercus urophasianus*; 1)	Coal
		Multiple targets (1)	Coal
	Amphibia, Reptilia	Multiple targets (1)	Coal
South America	Aves	Multiple targets (1)	Not stated
Europe	Aves	Ring-necked pheasant (*Phasianus colchicus)*, European nightjar (*Caprimulgus europaeus*), and Yellowhammer (*Emberiza citronella*; 1)	Coal
		Common quail (*Coturnix coturnix*; 1)	Coal
	Amphibia, Reptilia	Multiple targets (1)	Coal
Asia	Aves	Multiple targets (1)	Tin
Africa	Mammalia	Multiple targets (1)	Not stated

*Denotes number of studies for each target or mineral type

2.5 Discussion

Studies of faunal responses to mine site restoration are lacking globally, and we found over a 49-year period just 101 peer-reviewed publications reporting on fauna as part of mining restoration activities, with over half from Australia. We interpret this number as ‘lacking’ because 46 of the 101 studies originated from either the same mining site, or the same locality within a country. Furthermore, as a very rough guide, as of October 2018, Google Scholar reports ~24 000 papers reporting on ‘vegetation’ AND ‘ecological restoration’ AND ‘mining’ in the same period since 1971. Studies of faunal responses to mine site restoration favoured assessments for general species diversity and abundances of invertebrate species. There is a noticeable lack of studies that assess the behaviour and ecology of fauna, particularly of vertebrate species.

2.5.1 Study origin

Australia is at the forefront of mining restoration initiatives, as one of the few countries with widespread legislation (complemented by non-compliance penalties) aimed at mine closure (Gilbert, 2000; Clark and Clark, 2005; Cristescu *et al.*, 2012). This is reflected in the number of studies reporting faunal responses to mine site restoration originating within Australia. Australia's high activity within the mining restoration field likely results from the increased availability of funding that mineral extraction companies are required to provide for ecological restoration following mine site discontinuation, in order to obtain closure (Clark and Clark, 2005). While a leader in restoration research, a recent report identified ~60 000 mine sites across Australia as abandoned (Campbell *et al.*, 2017), of which the number confirmed as restored and officially closed could be as low as 21 (Western Australia: unknown; South Australia: 18 sites; New South Wales, Victoria, and Tasmania: one site each; Queensland and Northern Territory: no confirmed sites: Campbell *et al.*, 2017). It is apparent that restoration research focused on reinstatement of fauna after mining is still lacking within Australia. Outside Australia, global mine abandonment numbers are largely either unknown or under-reported. Among countries with (soundly estimated) abandonment figures, high numbers are common, with at least 5000 mine sites in South Africa and 10 000 in Canada identified as abandoned (Cowan *et al.*, 2010; Milaras *et al.*, 2014), many unlikely to have any substantial ecological management effort that would achieve restoration as defined by McDonald *et al.* (2016).

Rates of mine site cessations and abandonments are cumulatively growing worldwide; however, legislation relating to mine site closure is lacking in most countries (Clark and Clark, 2005). Within developed nations, only four countries have widespread legislation relating to mine abandonment (Australia, Japan, Ireland and the United Kingdom), and two have legislation in select states (Canada and the United States: Clark and Clark, 2005). Even fewer have legislation for bonding procedures (monetary bond to ensure sites are appropriately restored: Clark and Clark, 2005). Just 11 developing countries have complete legislation relating to mine site closure (Clark and Clark, 2005), none of which appear in our search results. Globally, Australia appears to be one of the leaders in this space, largely due to comprehensive legislation, although this clearly is not the only motivator as, of the three other developed regions with widespread legislated restoration requirements, we found just one publication relating to faunal responses to mine site restoration (from the United Kingdom). While closure legislation is an essential component in the regulation of mining activities, legislated financial support of restoration activities and research is equally critical.

While much of the literature originates from Australia, almost half of these are from a single organisation: Alcoa's bauxite mining operations in south-west Australia. Not only does this organisation account for a significant proportion of Australian studies, but almost all studies from bauxite mines globally – a mining practice with large surface impacts. These studies originate in a unique ecological region, and a biodiversity hotspot that has been isolated from the rest of the world for a substantial period (Hopper and Gioia, 2004). It is highly likely that patterns seen from these studies in the southwest Australian biodiversity hotspot may not provide an accurate representation of faunal responses to mine site restoration in other understudied regions. While it is unlikely that a single, standardised approach to fauna restoration in mining could be implemented globally, due to the ecological diversity of habitats, until legislative requirements and funding increase globally, the diversity of responses by faunal communities to mine site restoration will remain obscure.

2.5.2 Invertebrate responses

Invertebrate species are most commonly studied in assessments of faunal responses to mine site restoration success, and have been studied across a wide diversity of mineral extraction operations. Invertebrates are exceptionally diverse and abundant and typically respond rapidly and with high sensitivity to habitat disturbance, providing an ideal study group for monitoring environmental change and habitat health (Waltz and Covington, 2004; Gerlach *et al.*, 2013). Among the mining restoration literature involving studies of particular invertebrate groups, there is a strong focus on assessing diversity and abundances of ant species. Ants have been used extensively as bioindicators in a range of studies, across many habitat types and land uses (Hoffmann and Andersen, 2003), including savannahs (Majer, 1984; Andersen, 1991; Cross *et al.*, 2016b), coastal environments (Majer and Brown, 1986; Cross *et al.*, 2016b), woodlands and forests (Andersen, 1991; Vanderwoude *et al.*, 1997), including rainforest (King *et al.*, 1998). Ants are an obvious study group of choice, occurring in exceptional abundances in all but three regions (Iceland, Greenland, and Antarctica: Hölldobler and Wilson, 1990).

Ant community dynamics and responses to disturbances are well studied, and sampling can be performed with ease, rapidity, and at comparatively low cost (Majer, 1983; Andersen, 1986). One of the few drawbacks in their use stems from difficulties in taxonomy, with many species yet to be described and named (Gerlach *et al.*, 2013). Their widespread use across the mining literature is therefore unsurprising. While ants are the most commonly targeted group, general species diversity assessments for multiple groups (no specific targets) are equally common. General diversity assessments may present further issues, in that they do not account for varying ecologies of species, and identification tends to be broader (Chiarucci *et al.*, 2011). Species diversity and richness assessments are one of the most straightforward and reliable forms of data collection, especially when targeting fauna present in large numbers (Gerlach *et al.*, 2013), likely accounting for the significant bias towards this form of assessment over all other study types.

2.5.3 Vertebrate responses

Vertebrates are less frequently studied in assessments of mine site restoration, and are generally considered to be less effective for use as bioindicators of habitat health than invertebrates (Landres *et al.*, 1988; Bisevac and Majer, 1999; Gerlach *et al.*, 2013). Unlike invertebrates, many species of which occur in high numbers across many habitats, vertebrates can be cryptic, often present in far fewer numbers, and move over greater spatial scales, considerably increasing detection difficulty (Oliver *et al.*, 2009). Few studies assess behavioural and ecological responses of vertebrate fauna, particularly apex predators, to mine site restoration. Behavioural studies can be particularly costly (especially in the initial set-up stage); however, they can also provide extremely successful measures for assessments of the interactions of fauna with their surrounding habitat (Silveira *et al.*, 2003).

Assessments of vertebrate responses to mine site restoration favour avian fauna. This is particularly evident in studies originating outside of Australasia, two-thirds of which assess responses of birds. Birds are relatively easy to detect and identify, have a stabilised taxonomy, often can be common and widespread, and their environmental interactions are well studied, providing an excellent faunal group for use in studies of ecosystem health (Jordano, 1982). However, birds may not accurately represent restoration use, as their great mobility may allow for easier recolonisation than other fauna groups. Second to birds, there are relatively substantial numbers of mammal-focused studies, particularly of charismatic mammals and those that have threatened conservation status. Australia is a land of lizards, and has extremely high rates of endemism (93% endemism: Chapman, 2009), yet despite being one of the few countries to assess responses of non-avian taxa, there are surprisingly few reptilian studies. Reptiles are experiencing global declines (Böhm *et al.*, 2013), yet they are often overlooked, with few studies examining their response to habitat restoration (Munro *et al.*, 2007; Todd *et al.*, 2010). Reptiles can provide information on thermal environments (e.g. whether restoration areas have higher associated thermal costs than reference habitats), which other groups, such as birds, may not. Hence, extrapolating responses of birds to poikilothermic fauna is potentially problematic.

2.5.4 Ecosystem function

Research is lacking into ecosystem functionality in terms of assessing interactions of fauna with mine site restoration areas. In many ecosystems, functionality is in some way related to faunal interactions, and loss of biodiversity can greatly impact on ecosystem services (Naeem *et al.*, 1994), yet 81% of studies identified in this review of mine site restoration measure species diversity, abundance, presence, or infer absence. While providing important ecological data, these studies have several drawbacks, and may not provide data on whole ecosystem functionality or be appropriate measures for determining whether a site has been effectively restored. By performing only these assessments, there is a significant chance of missing rare and cryptic species, or in incidental captures of animals moving through the site but not inhabiting the area. This may be particularly problematic in terms of achieving outcomes for mining restoration, as it may provide a false community representation and appear as though a habitat is restored when, in fact, that system may only be in use opportunistically, or not even in use at all.

Moreover, only so much may be learnt from assessing faunal biodiversity. Key ecosystem functions can result or fail as a result of altered animal behaviour and movement patterns (Fahrig, 2007; Tarszisz *et al.*, 2018), ecological energetics (Tomlinson *et al.*, 2014), or nutritional physiology (Birnie-Gauvin *et al.*, 2017). This can result in cryptic disruptions to key services such as insect pollination (e.g., Tomlinson *et al.*, 2018) that are not apparent from other studies of pollinator communities such as birds (e.g., Frick *et al.*, 2014). Although there is some evidence that successful mine site restoration is constrained by limited natural recruitment (Koch, 2007; James *et al.*, 2011), the role of fauna-mediated pollination and seed dispersal is understudied. Herbivory is a critical plant/animal interaction that has long hampered the restoration of discontinued mining areas, yet has been rarely studied (Keesing and Wratten, 1998; Koch *et al.*, 2004; Parsons *et al.*, 2007). These dynamic interactions are important to restoration research, yet fauna are studied only in the context of ecological restoration at a restricted level.

2.5.5 Grey literature and issues with its use

While it is possible that information and data surrounding faunal responses to mine site restoration exist within the grey literature, we found little empirical data or relevant information within the few that were readily accessible. Accessible grey literature largely comprises pre-mining surveys for fauna species within and around potential new mine sites, conservation and management strategies for rare and threatened species during the life of the mining operation, conference proceedings, or book chapters without empirical data. There is a noticeable dearth of grey literature directly referencing either short- or long-term monitoring of fauna in restoration landscapes, or methods for assessing faunal responses. However, as with published literature, the marginal volume of grey literature to which we could gain access did not discuss fauna in detail, and did not discuss whole animal community return, or return of fully functioning ecosystems.

We found eight articles directly referencing fauna in restoration landscapes: three conference proceedings or presentations, three book chapters, and two monitoring plans or guidance documents. Grey literature comprised discussion of the role or return of fauna in mine site restoration (Nawrot and Klimstra, 1989; Majer, 1997, 1998; Moloney *et al.*, 1998), a monitoring plan for the conservation of rare and threatened fauna (Nickel and Claremont, 2015), an assessment of nest translocations for bird species in restoration (termed reclamation) sites (McKee, 2007), a guidance document describing techniques for promoting fauna return to rehabilitating sites (Brennan *et al.*, 2005), and a book chapter referencing published studies of vertebrate colonisation of rehabilitating sites at Alcoa (Tibbett, 2015). Other resources do recognise the effects of mining on fauna, but this is limited to simple statements on the need for returning habitat components that promote faunal recolonisation; for example, habitat corridors (McLaughlin, 2012), monitoring plans for threatened species or management of feral species (without reference to restoration) (Guinea, 2007; Weipa, 2015; Knuckey, 2018) or simply recognition that fauna play important roles in ecosystems and are often overlooked in restoration monitoring (Glenn *et al.*, 2014).

Our biggest challenge in extending our analyses to the grey literature was that resources tend to be largely inaccessible, and often unreliable (Farace and Schöpfel, 2010; Corlett, 2011). Information and data in unpublished reports and documents are often accessible only within governmental departments and specific regions or countries, and not by the scientific community (Corlett, 2011). This has likely resulted in a significant proportion of information within grey literature being overlooked during the development of new conservation and management plans, restoration strategies, and mine site closure policies. It also allows for large, multinational companies to apply different standards in different countries depending on the local legislative and regulatory structures and departments. In order to advance the field of mine site restoration and develop targeted and effective fauna conservation and management strategies, data from these grey literature sources must be peer-reviewed, published, and accessible.

2.5.6 Conclusions and future research

The most obvious pattern that has emerged from our review of the literature on responses of fauna to mine site restoration is the overwhelming number of Australian studies contrasted by the surprising dearth of literature for the remainder of the world. This has likely resulted from Australia having both the legislative structure, and financial incentives and capacity for research. To gain an increased understanding of how restoration is impacting ecosystem functioning across a wide range of ecosystems, research must be expanded to a more global level, and encompass a wide range of habitats with varying types of mineral extraction. Not only will this help to account for differences between habitats and ecosystems, but also for the likelihood of varying environmental impact resulting from different mining techniques. Another major limitation is the restricted focus on assessments of behaviour and ecological interactions and functional capacity. Studies of species richness rarely offer insight into the critical ecosystem functions provided by animals. An increased focus must be placed on assessments for ecology and behavioural responses of animals to habitat change and restoration, with an increased emphasis on vertebrate animals within these systems. However, there needs to be a global realisation that mining regulatory systems need to place an emphasis on assessing fauna at multiple taxonomic and functional levels, to ensure that restoration after mining returns an ecosystem to a level of ecological resilience and capacity that matches the local reference ecosystem.

2.6 References

Every reasonable effort has been made to acknowledge the owners of the copyright material. I would be pleased to hear from any copyright owner who has been omitted or incorrectly acknowledged.

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Chapter 3. I don't like crickets, I love them: invertebrates are an important prey source for varanid lizards

> "*There are only two types of lizards: goannas, and goanna food* "
>
> – Dennis King

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Yellow spotted monitor (*Varanus panoptes*)

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3.1 Abstract

Minimal annual rainfall in arid environments results in low productivity ecosystems with fluctuating food availability. Large mammalian predators that require frequent consumption of vertebrate prey tend to be less abundant in desert environments; however, such environments often support numerous large-bodied carnivorous reptiles. Diet is a fundamental component of an animal's ecology, and we explore the diets of three co-existing, sympatric *Varanus* species occurring in arid Australia: *V. tristis, V. gouldii,* and *V. panoptes*. We hypothesised that the diet of varanids living in arid environments would primarily consist of relatively abundant invertebrate prey, and that vertebrate prey items would largely be limited to opportunistically consumed mammalian carrion and small reptilian species. All three *Varanus* species had high dietary overlap and broad, generalist diets. Invertebrate prey, particularly Orthoptera, were key to the diets of all three species. Vertebrate prey was infrequently consumed by all three *Varanus* species; however, when consumed, tended to comprise small reptilian species and mammalian carrion. Unlike large mammalian predators, varanids can survive on invertebrate prey and infrequent feeds, and can aestivate when conditions become unfavourable, contributing to their success in arid environments.

3.2 Introduction

Arid zones are characterised by low and unpredictable rainfall, extreme temperatures, and nutrient deficient soils with poor water retention (Nagy, 1994; Ayal, 2007). Rainfall is a primary driver of food availability and arid areas therefore largely consist of low productivity ecosystems (Hadley and Szarek, 1981). Limited resources and unfavourable conditions in arid environments strongly impact faunal communities and population dynamics in these habitats (McNeely, 2003; Schwinning and Sala, 2004). Even so, arid areas can support high diversities of mammalian and reptilian species; however, mammals tend to occur at far lower abundance than do reptiles and mostly comprise small species of lower trophic rank (James *et al.*, 1995; Geiser, 2004; Ayal, 2007). Low food availability and fluctuating environmental conditions in desert ecosystems are particularly limiting for apex and mesopredator species, and arid zones favour predators with low energy requirements, such as reptiles (Fisher and Dickman, 1993; James *et al.*, 1995).

Diet is a fundamental component of an animal's ecology and can provide important information on habitat requirements for population persistence, resource competition and partitioning, ontogenetic diet variation, and interactions between predator and prey species (Sih and Christensen, 2001; Rocha-Mendes *et al.*, 2010; Dalhuijsen *et al.*, 2014). Population dynamics and the extent to which sympatric species can persist within habitats are influenced by several factors, including partitioning in diet, microhabitat, and activity periods, and the carrying capacity of habitats is directly influenced by food productivity (Schoener, 1974, Simon and Middendorf, 1976; Kozlowski *et al.*, 2008; Derycke *et al.*, 2016). Food chains in arid environments are often heavily structured around invertebrates, which can be abundant and are staple food sources for many desert predators (Stafford Smith and Morton, 1990; Catling, 1988; Ayal, 2007). Although populations of large mammalian predators are constrained by restricted availability of vertebrate food sources, large reptilian predators can thrive in arid habitats (Stafford Smith and Morton, 1990; Read and Scoleri, 2015) because they are particularly well adapted to fluctuating prey and environmental conditions, being able to aestivate and prey upon invertebrates in addition to vertebrate species (Fisher and Dickman, 1993; Christian *et al.*, 1999). By comparison with reptiles, mammalian mesopredator and apex predators tend to require frequent consumption of vertebrate prey and are often less abundant in desert environments, which often have a highly dynamic 'boom and bust' ecology (Paltridge, 2002; Letnic and Dickman, 2010; Arthinton and Balcombe, 2011).

Approximately 70% of Australia's landmass comprises arid environments (Stafford Smith and Morton, 1990; Morton and James, 1988). Although reptiles are common in low productivity environments globally, the arid regions of Australia support amongst the highest richness of reptiles in the world, with reptiles tending to be the most dominant, diverse, and abundant vertebrate fauna in these habitats (Pianka, 1969; Morton and James, 1988; Roll *et al.*, 2017). In arid Australia, varanids fill high order predator roles in the broad absence of apex mammalian species (Read and Scoleri, 2015). Being the high order predators in these ecosystems, varanids exert top-down control of prey (Pianka *et al.*, 2004; Read and Scoleri, 2015), with some acting as keystone species (e.g., *V. gouldii*; Bird *et al.*, 2013). Australian varanids are highly diverse, both ecologically and in size, occupying a range of arboreal and terrestrial niches and encompassing almost the entire size range of the genus (~20cm to 2.5m - *V. sparnus* to *V. giganteus*; Losos and Greene, 1988; King and Green, 1993; Pianka *et al.*, 2004). Varanids forage over large areas and have diverse, cosmopolitan diets (King and Green, 1993). Having a broad diet is likely an adaptation to living in food-limiting environments, and sympatric species of varanids, particularly those occupying the same habitat niche (e.g., terrestrial or arboreal), are likely to share a broad dietary niche overlap (Dalhuijsen *et al.*, 2014). The arid areas of Australia support the highest species richness of varanids (Pianka, 1995) with up to six Varanus species co-existing sympatrically within arid regions (Pianka, 1994), raising the question of how so many large ectothermic species can co-exist.

Here, we analysed the stomach contents of three varanid species (*V. tristis, V. gouldii,* and *V. panoptes*) collected within the arid Mid West region of Western Australia. All three *Varanus* occur sympatrically across much of the Mid West region, with *V. tristis* being primarily arboreal, while *V. gouldii* and *V. panoptes* are primarily terrestrial (Cogger, 2014). We hypothesised that i) due to occupying an arboreal niche, *V. tristis* would have less dietary overlap with the other species, while the two terrestrial species would exhibit significant overlap; ii) a high proportion of the dietary contents of all three *Varanus* species would comprise relatively abundant invertebrate prey, allowing for populations to thrive in arid environments; and iii) consumption of vertebrate prey would largely be restricted to mammalian carrion or small reptilian species.

3.3 Methods

3.3.1 Study site and species

The three *Varanus* species chosen demonstrate a range of body sizes and occupy a range of niches within the Mid West region of Western Australia (Fig. 3.1a,b): *V. tristis* (arboreal, total length 0.76m), *V. gouldii* (terrestrial, 1.2m), and *V. panoptes* (terrestrial, 1.4m) (Wilson and Swan, 2003). Each of the species occurs in abundance across a wide geographic range, and habitats overlap extensively within arid Australia

(Pianka *et al.*, 2004). *Varanus gouldii* and *V. panoptes* share similar habitat preferences; however, *V. gouldii* are predominantly found in habitats with sandy soils (Thompson, 2004; Christian, 2004). *Varanus tristis* occur across most habitat types and are common in arid areas where microhabitats such as tree-hollows and logs are in abundance (Pianka, 2004). Specimens chosen for dissection were restricted to those collected from within the Mid West region, obtained from the Western Australian Museum (WAM; all available Mid West specimens dissected), or collected opportunistically when found dead on roads (DOR).

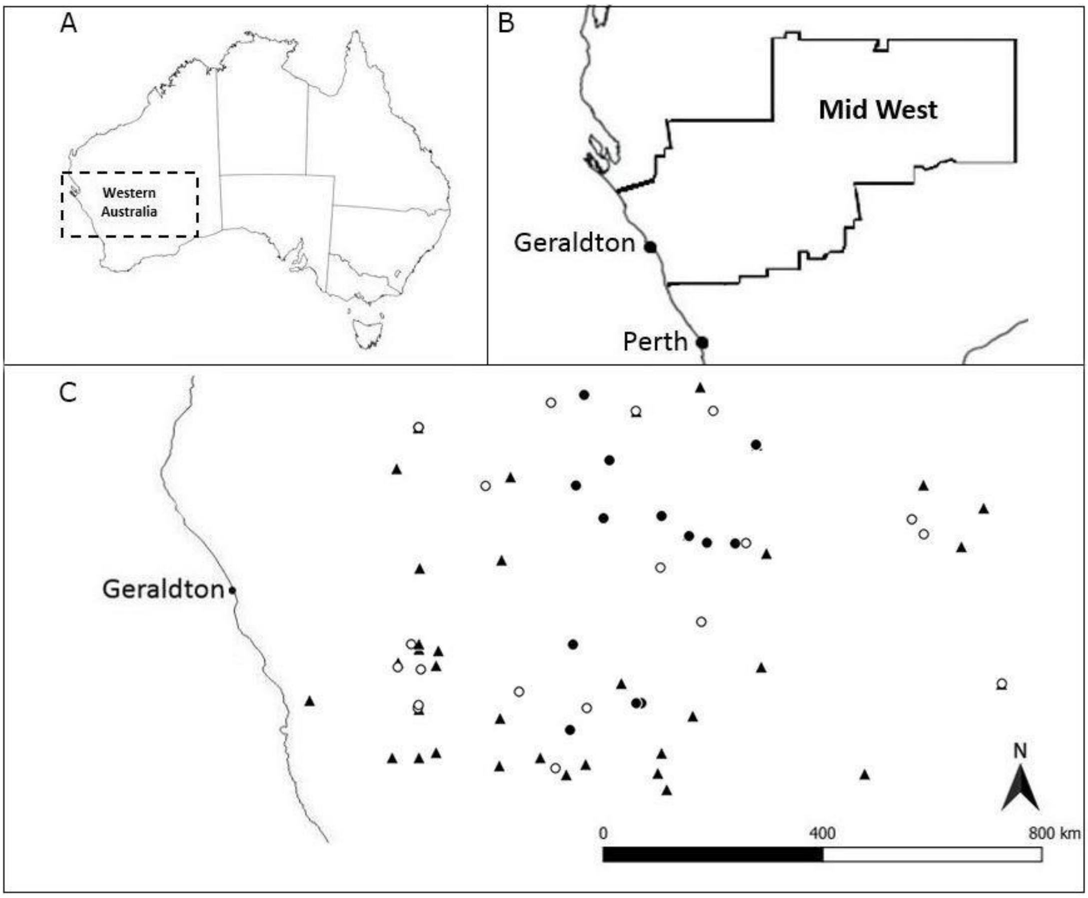

Fig. 3.1: Collection location of *Varanus* specimens: a) the study location within Western Australia, b) the Mid West region of Western Australia, and c) specimen collection locations within the Mid West region for each species, *V. tristis* (○), *V. gouldii* (▲), and *V. panoptes* (•).

We were limited to dissecting a total of 78 specimens (22 *V. tristis*, 40 *V. gouldii*, and 16 *V. panoptes*). Sample sizes of this magnitude are a common constraint amongst studies of reptile diets, including those assessing varanid diet (James *et al.*, 1992; Strahan *et al.*, 1998; Guarino, 2001; López and Giraudo, 2004; Dalhuijsen *et al.*, 2014) and our sample sizes are comparable to those of other studies (James *et al.*, 1992;

Losos and Greene, 1988; Dalhuijsen *et al.*, 2014). We assessed the adequacy of sample sizes for each species using species accumulation curves in PAST (Version 3.25, Hammer *et al.*, 2001). Collection locations for each specimen dissected in our study were evenly spread across the collecting region (Fig. 3.1c), and collection years ranged from 1958 to 2017 (*V. tristis*: 1958 – 2004, *V. gouldii*: 1960 – 2017, and *V. panoptes*: 1979 – 2017; Fig. 3.2). To assess whether there were any biases in collection dates, we used a two-way chi-squared test comparing collection dates (grouped by decade, n = 7) between species (Wolfe *et al.*, 2017).

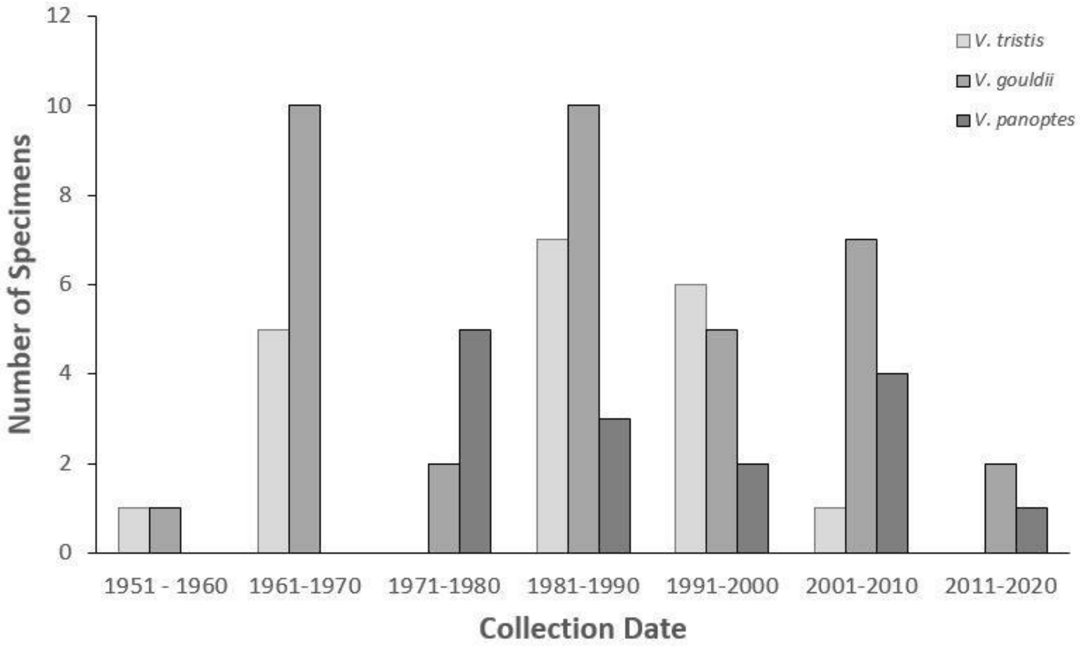

Fig. 3.2: Collection years for *Varanus tristis*, *V. gouldii*, and *V. panoptes* specimens from the Mid West region of Western Australia.

3.3.2 Dissections

Prior to dissection we measured snout-vent length (SVL) for all 78 specimens. Specimens were then opened via a ventral abdominal incision and sexed. Prey items were then removed via an incision along the length of the stomach. Upon removal, stomach contents were drained and blotted to remove excess preserving liquid, weighed (discounting incidental detritus ingested e.g. sand, rocks, sticks), and prey items were identified to taxonomic order, or species where possible. Thirty-two specimens had empty stomachs, with specimens of the smaller species, *V. tristis*, most commonly without prey items (50%), and proportions of specimens without prey items similar between the larger species (40% *V. gouldii*, 32% *V. panoptes*). Reptile specimens often lack stomach contents and a high proportion of empty stomachs is a common issue in studies of reptile diet (Patchell and Shine, 1986: Losos and Greene, 1988: Holycross and Mackessy, 2002). Forty-seven specimens contained identifiable prey items, and the contents of two stomachs could not be identified with certainty and were excluded from further analyses. We recorded the total number of each prey type in the stomach contents of each of the *Varanus* species; however, as many prey items were partially digested, we recorded the total dry weight of all invertebrate and vertebrate prey items, and we controlled for variation across stomach samples by calculating the average proportion (P_{ia}, P_{ib}) of invertebrate and vertebrate species consumed by each species.

3.3.3 Prey items and species niche overlap

Following Dalhuijsen *et al.* (2014) and Waite *et al.* (2011), we calculated the frequency of occurrence (*FO*) of each prey type to assess their prevalence in the diets of *V. tristis*, *V. gouldii*, and *V. panoptes*. As many prey items were partially digested and total weight of each recorded prey type in specimens was minimal, *FO* was analysed using the total number of individuals of each varanid species containing each prey type, where:

$$FO = \left(\frac{Number\ of\ varanids\ of\ species\ a\ containing\ prey\ type\ i}{Total\ number\ of\ varanid\ specimens\ of\ species\ a}\right) \times 100$$

Dietary overlap (O_{ab}) of invertebrate and vertebrate prey items between each species pair (*V. tristis*, *V. gouldii*, and *V. panoptes*) was calculated using Pianka's niche overlap index (Pianka, 1973), where P_{ia} and P_{ib} represent the proportion of the i^{th} prey type in each varanid species (a, b):

$$\mathrm{O}_{ab} = \frac{\sum P_{ia}P_{ib}}{\sqrt{\sum(P_{ia})^2 \sum(P_{ib})^2}}$$

O_{ab} is indicative of the extent to which two species overlap in diet resources and ranges between 0 (no overlap, species differ in diet and do not share any prey items), and 1 (complete overlap, diet between species does not differ), with dietary overlap between species considered high at $O_{ab} > 0.6$ (Wallace, 1981; Waite *et al.*, 2011; Dalhuijsen *et* *al.*, 2014). We assessed whether the proportion of individuals containing invertebrate and vertebrate prey differed between species, using a contingency table with the number of individuals of each species containing invertebrates and vertebrates. As this analysis assessed diet at a coarse taxonomic level, we further assessed dietary differences between species at the ordinal level (unidentified eggs, pupae, and fur were treated as 'orders') by recording the presence or absence of each prey order for each individual. We then constructed a between-individual similarity matrix using a Euclidean similarity measure and used this matrix to visually represent the data using a non-metric multi-dimensional scaling (nMDS) and compare differences between species using analysis of similarity (ANOSIM) in Primer 6.0 (Primer-E, 2006).

3.3.4 Ontogeny and sex

Varanus tristis were considered adult at SVL > 200mm, *V. gouldii* at SVL > 250mm (Pianka, 1994), and *V. panoptes* at SVL > 300mm (Shine, 1986). We had seven adults and four juvenile *V. tristis* specimens; 13 adult and 12 juvenile *V. gouldii*, and eight adult and three juvenile *V. panoptes* containing identifiable prey items. These included one male and five female *V. tristis*, 10 male and 8 female *V. gouldii*, and two male and seven female *V. panoptes*. Twenty-three varanids could not be sexed with confidence, either due to damage or age (hatchlings could not be sexed with certainty). As sample sizes for male and female, and juvenile and adult *V. tristis* and *V. panoptes* were limited, we excluded these species from analyses and restricted analyses between sexes and ages to *V. gouldii*; however, the rigour and power of our analyses are limited by the restricted availability of specimens (Dalhuijsen *et al.*, 2014). As with interspecific diet analyses, we assessed whether there were significant differences in the proportion of invertebrate and vertebrate prey items consumed between juvenile or adult, and male or female *V. gouldii* individuals using contingency tables for each analysis. We also analysed ontogenetic and sexual differences in diet at the ordinal level by constructing a between-individual similarity matrix, using the same methods as for specific differences above, visually representing diets using nMDS and comparing differences between sex/age using ANOSIM. Analyses for age and sex were conducted separately.

3.4 Results

3.4.1 Sample size

Species accumulation for *V. gouldii* stomach contents showed a distinct plateau in accumulation between 22 and 23 specimens, where very few further prey species were likely to be found in the gut contents (Fig. 3.3a). The accumulation curves for our other two species had not yet approached their asymptotes within the number of specimens that we analysed, but when projected further, they extrapolated a plateau in species accumulation between 14-16 specimens of *V. panoptes* and 12-14 specimens of *V. tristis*. (Fig. 3.3b,c). While the power of our analyses may be impacted by sample size, accumulation curves for each species appear to be reaching an asymptote, and sample sizes are unlikely to be a significant constraint to analyses.

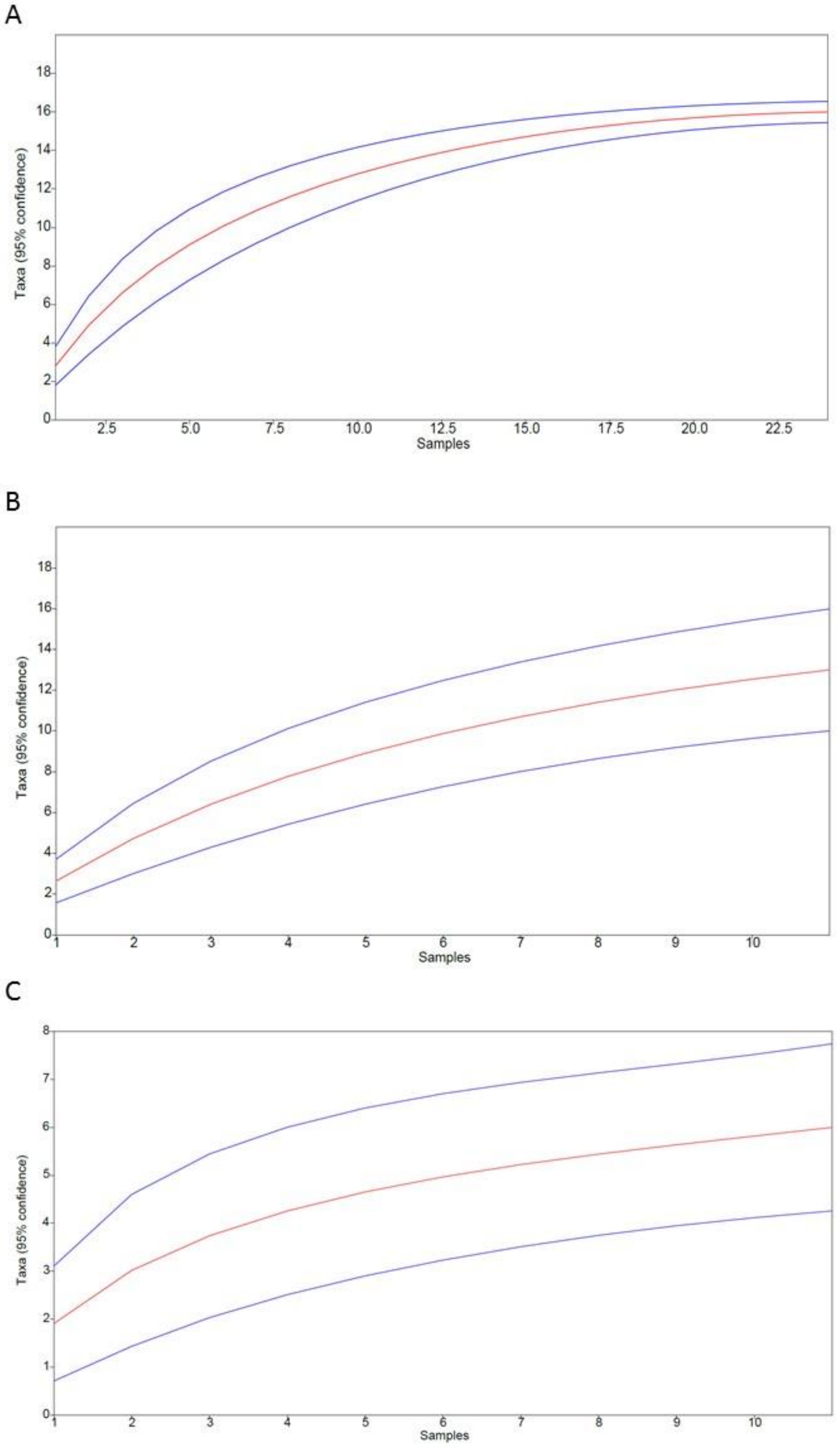

Fig. 3.3: Sample accumulation curves for: a) *Varanus gouldii*, b) *V. panoptes*, and c) *V. tristis*, showing the rate at which new prey species are identified as the number of specimens dissected increases. Our data concerning *V. gouldii* suggest that very few additional prey items were likely to be identified with increasing numbers of specimens. Our extrapolation of species accumulation in the gut content of *V. panoptes* and *V. tristis* suggest that a further two to four specimens might be necessary to gain as complete an understanding of their dietary breadth.

3.4.2. Prey composition

We did not detect any significant temporal differences for collection years of each species (χ^2 = 10.122, d.f. = 12, *P* = 0.605). Prey items in each species were diverse; however, invertebrates were most frequently recorded (Table 3.1). Orthoptera were common and appeared to be a key prey item for all three *Varanus* species. In particular, specimens contained Acrididae, and 'raspy crickets' – large, robust, nocturnal members of the Gryllacrididae > 5 cm long (Rentz and John, 1989). *Varanus gouldii* had the most diverse diet with 58.3% (*n* = 14) of our specimens containing both invertebrate and vertebrate items, 33.3% (*n* = 8) containing only invertebrates, and 8.3% (*n* = 2) containing only vertebrate prey items. *Varanus tristis* showed similar prey composition to *V. gouldii*; however, vertebrate prey was less common in *V. panoptes* specimens (45%, *n* = 5/11), than *V. gouldii* (67%, *n* = 16/24), or *V. tristis* (82%, *n* = 9/11).

Invertebrate prey across all varanid species comprised Blattodea (roaches), Coleoptera (beetles), Collembola (springtails; however due to their small size and low frequency of occurrence, Collembola were likely to be incidentally ingested), Diplopoda (millipedes), Hymenoptera (ants), Lepidoptera (larvae), Orthoptera (crickets, grasshoppers), Scolopendridae (centipedes), and invertebrate cocoons. Vertebrate prey comprised skinks (*Ctenotus* spp., *C. helenae*, *Eremiascincus richardsonii*, *Lerista* spp., *Morethia* spp., unknown skink spp.), dragons (*Ctenophorus* spp., *C. ornatus*, and *C. reticulatus* or *C. nuchalis*; femoral pores unable to be checked for verification), varanids (*Varanus* spp., *V. acanthurus*, and *V. caudolineatus*), and Mammalia (Macropodidae spp., Muridae spp., *Oryctolagus cuniculus*, unknown hair/fur remains), and vertebrate egg sacs. Abundance of prey items recorded in each the three species is summarised in Table 3.1.

Table 3.1: The total number of each prey type recorded in the stomach contents of *Varanus tristis*, *V. gouldii*, and *V. panoptes*, and the frequency of occurrence (*FO*; recorded as a percentage) of specimens of each species containing each prey type. Individual specimens appear multiple times across *FO* categories, where they had a varied gut content.

	Varanus tristis		***Varanus gouldii***		***Varanus panoptes***	
Prey type	**No. prey**	***FO***	**No. prey**	***FO***	**No. prey**	***FO***
Invertebrates						
Arachnida	3	27	14	41	2	18
Blattodea	0	0	0	8	6	27
Coleoptera	0	0	0	25	12	45
Collembola[1]	0	0	0	8	4	27
Diplopoda	0	0	0	0	2	9
Hymenoptera	0	0	62	12	1	9
Lepidoptera	0	0	0	12	13	9
Orthoptera	19	72	45	50	8	45
Scolopendridae	2	9	17	25	0	0
Invertebrate pupae	0	0	0	4	0	0
Reptilia						
Scincidae						
Ctenotus spp.	4	36	7	29	0	0
Ctenotus helenae	1	9	0	0	0	0
Eremiascincus richardsonii	1	9	0	0	0	0
Lerista spp.	0	0	0	0	1	9
Morethia spp.	0	0	1	4	0	0
Unknown spp.	0	0	0	0	1	9
Agamidae						
Ctenophorus spp.	1	9	1	4	1	9
Ctenophorus ornatus	0	0	1	4	0	0
Ctenophorus reticulatus/nuchalis	0	0	2	8	0	0
Varanidae						
Varanus acanthurus	0	0	1	4	0	0
Varanus caudolineatus	0	0	1	4	0	0
Varanus spp.	0	0	1	4	0	0
Mammalia						
Macropod spp.	0	0	0	0	2*	18
Muridae spp.	1	9	2	8	0	0
Oryctolagus cuniculus	0	0	2*	8	2*	18
Unknown hair/fur remains	0	0	2*	8	0	0
Vertebrate eggs	0	0	31	12	11	18

[1] Likely incidentally ingested

*Mammalian prey likely to be consumed as carrion for which accurate prey abundance could not be determined.

3.4.3 Dietary overlap

All three *Varanus* species had strong dietary overlap, with minimal differences in diet between the species. Pianka's niche overlap index was high between all species pairs, with almost complete overlap between *V. tristis* and *V. gouldii* ($O_{ab} = 0.99$), and high overlap between *V. tristis* and *V. panoptes* ($O_{ab} = 0.89$), and *V. gouldii* and *V. panoptes* ($O_{ab} = 0.91$). The proportion of invertebrate or vertebrate prey consumed did not differ significantly between species ($\chi^2 = 4.15$, d.f. = 2, $P = 0.13$), male and female *V. gouldii* ($\chi^2 = 0.76$, d.f. = 1, $P = 0.38$), or juvenile and adult *V. gouldii* ($\chi^2 = 0.67$, d.f. = 1, $P = 0.41$), with high dietary overlap between both juveniles and adults, and males and females ($O_{ab} > 0.9$). There were no significant differences in diet at the ordinal level between species ($r = -0.049$, $P = 0.774$; Fig. 3.4a), or either age ($r = -0.034$, $P = 0.702$; Fig. 3.4b) or sex for *V. gouldii* ($r = 0.031$, $P = 0.312$; Fig. 3.4c).

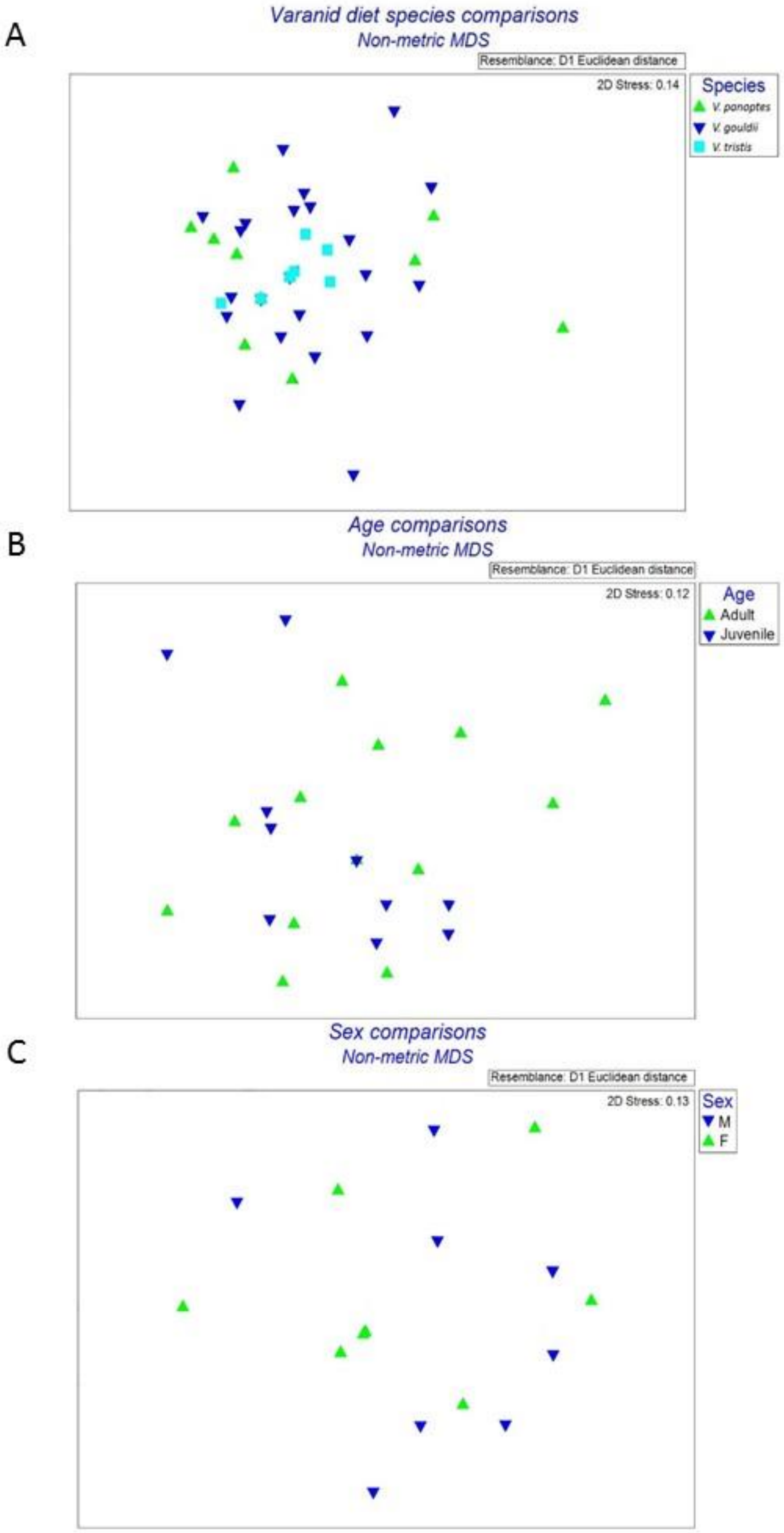

Fig. 3.4: Non-metric multidimensional scaling (nMDS) plots for overlap of diet at an ordinal level between: a) each species; *V. tristis*, *V. gouldii*, and *V. panoptes*, b) juvenile and adult *V. gouldii* specimens, and c) male and female *V. gouldii* specimens.

3.5 Discussion

Our data supported two of our predictions; that *Varanus* species from the arid Mid West region of Western Australia feed primarily on invertebrate food sources, and vertebrate prey consumption was largely restricted to mammalian carrion and small reptilian species. Invertebrate prey items were common in the stomach contents of all three of our species, and particularly Orthoptera and Coleoptera appear to be essential prey groups. Vertebrate prey items were rarely consumed, and when present largely comprised Macropodidae or *O. cuniculus* (rabbit) remains, or species of Scincidae and Agamidae. *Varanus tristis*, *V. gouldii*, and *V. panoptes* specimens had high dietary overlap despite *V. tristis* occupying arboreal niches. Limited sample sizes are a common issue amongst studies of varanid diet, particularly those using museum specimens (Losos and Greene, 1988; Guarino, 2001; Dalhuijsen *et al.*, 2014), however interrogating the species accumulation data extracted from our gut content records suggest that sample sizes are unlikely to significantly impact conclusions drawn from our data. Furthermore, it is possible that our results may be influenced by temporal effects resulting from collecting specimens across a range of decades. However, we did not note any significant differences in collection years between species and, as such, we concluded that temporal impacts were unlikely to have influenced our data.

We did not find any evidence of ontogenetic dietary differences amongst *V. gouldii* specimens, and our data indicated that this species had the most diverse diet. While this broad diet may reflect the larger available sample size in comparison to *V. tristis* and *V. panoptes*, *V. gouldii* have been previously reported as being a highly cosmopolitan and generalist species (Losos and Greene, 1988). Our results reflect those of other studies on similar sized sympatric *Varanus* species which report a high importance of invertebrate prey items (e.g., King and Green, 1979; Losos and Greene, 1988; Weavers, 1989; Bennett, 2002), and high dietary overlap both intra- and interspecific, despite occupying the same habitat niches (e.g., Sutherland, 2011; Dalhuijsen *et al.*, 2014).

3.5.1 Survival in low productivity habitats

Varanids tend to have opportunistic and cosmopolitan diets, feeding on abundant or readily available prey such as invertebrates, reptiles, and carrion (Pianka *et al.*, 2004; Sutherland, 2011; Dalhuijsen *et al.*, 2014). Our results reflect those of other studies which report *V. tristis*, *V. gouldii*, and *V. panoptes* to have diverse and generalist diets, with consumption of invertebrate and small reptilian prey common, but infrequent consumption of mammalian prey (Losos and Greene, 1988). Food availability in low productivity and arid ecosystems is a primary driver of population persistence (Hódar *et al.*, 2006). Vertebrate prey items are energetically important but are often scarce in arid and hot environments and the metabolic costs associated with their capture can be high (Losos and Greene, 1988). Vertebrate prey recorded in each of our species tended to comprise small reptilian species, which are present at high diversity and abundance in arid Australia (Pianka, 1969; Morton and James, 1988; Roll *et al.*, 2017).

Mammalian prey were rarely recorded in *V. tristis*, *V. gouldii*, or *V. panoptes*, consistent with studies of their diet across their Australian geographic range (Losos and Greene, 1988), and when present largely comprised *O. cuniculus* and Macropodidae remains. As varanids scavenge prey (Losos and Greene, 1988; Bennett, 2002; Blamires, 2004) we concluded mammalian prey (excluding *Mus* spp.) were probably carrion and consumed opportunistically. The increased costs associated with the capture of vertebrate prey may reflect their lower frequency in the stomach contents of *V. tristis*, *V. gouldii*, and *V. panoptes* specimens, and we note a high dependency on invertebrate prey items. A restricted availability of vertebrate prey items limits the carrying capacity of mammalian predators in arid habitats (Morton and James, 1988), however high order reptilian predators can thrive, often co-existing at high densities (Pianka, 1981).

Invertebrates are a staple food item for many desert animals, particularly during periods of low rainfall (Losos and Greene, 1988; Paltridge, 1997; Paltridge, 2002; Sutherland, 2011). While invertebrates can be abundant in desert systems, due to their small size they provide a lower energy food source per individual than vertebrate prey, and large mammalian predators, which have high energetic requirements, tend to be constrained in Australian arid environments (Carbone *et al.*, 1999). Varanids can capture larger prey through prolonged high-speed movement and an ability to sustain high metabolic rates (Losos and Greene, 1988; Clemente *et al.*, 2009). Vertebrate prey are often scarce and only seasonally available, and many of the *Varanus* species are primarily insectivorous (Losos and Greene, 1988; Stafford Smith and Morton, 1990). Varanids are well adapted to resource limiting environments with scarce food resources and are thought to forage primarily in areas with high densities of invertebrate prey, often digging for prey items, with capture of vertebrate prey items largely opportunistic (Shine, 1986; Losos and Greene, 1988). In addition to an ability to aestivate during unfavourable or limiting conditions (e.g., Christian *et al.*, 1995; Christian *et al.*, 1999; Doody *et al.*, 2014), varanids can survive on infrequent feeds and invertebrate prey (Morton and James, 1988; Secor, 2001). The success of the *Varanus* spp. in Mid West Western Australia is likely attributable to their ability to prey upon a diverse range of invertebrate prey items.

3.5.2 Intraspecific dietary patterns of V. gouldii

Ontogenetic changes in diet are common amongst many reptilian species (e.g., Duffield and Bull, 1998; Fialho *et al.*, 2000; Durtsche, 2000), but few studies of varanids report on the influence of ontogeny (Losos and Greene, 1988; Dalhuijsen *et al.*, 2014) or sex on diet. While ontogenetic analyses were confined to *V. gouldii*, we did not detect any significant dietary differences between age groups for *V. gouldii*. *Varanus gouldii* is a generalist species with a cosmopolitan diet (Losos and Greene, 1988), and we note this species comprised the most diverse range of prey items in stomach contents of the three *Varanus* species that we studied. Select studies have reported some ontogenetic dietary differences amongst varanid species, for example *V. bengalensis* and *V. komodoensis* (Losos and Greene, 1988), and ontogenetic differences may be present in the other species, particularly *V. tristis*, which occupies a more restricted niche than terrestrial species. Varanids share a common morphology and although males tend to be larger within most species, do not exhibit sexual dimorphism in body shape (Hnízdo *et al.*, 2011). Sex does not appear to influence diet, and a lack of sexual dimorphism in all three species has likely resulted in similar diets between sexes. However, conclusions drawn from our data are limited by a scarcity of specimens.

3.5.3 Limitations

There are some limitations to dietary studies using museum specimens and opportunistic collections. We dissected all available *V. tristis*, *V. gouldii*, and *V. panoptes* specimens collected from the Mid West region, however our sample sizes for each species were limited and may have impacted the power of our analyses for some species. Specimen scarcity is a common issue amongst studies of varanids, even amongst those assessing specimens across large collection regions (Losos and Greene, 1988; James *et al.*, 1992; Guarino, 2001; Dalhuijsen *et al.*, 2014). This issue is not restricted to varanids but is common across the literature assessing reptile diet (Strahan *et al.*, 1998; Holycross and Mackessy, 2002; López and Giraudo, 2004). In addition to low sample sizes, larger-bodied reptilian predators tend to feed infrequently and the likelihood of collecting specimens with empty stomachs is high (Losos and Greene, 1988; Huey *et al.*, 2001; Dalhuijsen *et al.*, 2014). Forty-one percent of our specimens contained no prey items, as is commonly reported amongst studies of reptilian diet (Patchell and Shine, 1986; Strahan *et al.*, 1998; López and Giraudo, 2004), with some studies of varanid diet reporting up to 66% of dissected specimens without stomach contents (Schmidt *et al.*, 1919; Campbell, 2005; Dalhuijsen *et al.*, 2014).

Many specimens used in this study were collected DOR, and differences in the ecological niches occupied by each species may have influenced the collection of specimens. For example, an arboreal species such as *V. tristis*, which infrequently ventures across open ground (Pianka *et al.*, 2004) has a lower likelihood of being collected DOR than would *V. gouldii* and *V. panoptes*, both of which are primarily terrestrial species (Cogger, 2014), and may preferentially scavenge along roadsides (Hastings *et al.*, 2019). Finally, diet analyses may be biased against soft-bodied prey which are more readily digestible (González-Solís *et al.*, 1997), and analysis of a species' diet through dissection may therefore provide an under-representation of its dietary breadth. However, this is unlikely to be a significant issue in our study as invertebrate prey were more commonly recorded in stomach contents than vertebrate prey items.

3.5.4 Conclusions

While we were constrained by a limited sample size for each of the three species, *V. tristis, V. gouldii,* and *V. panoptes,* our prediction that invertebrate prey is vital to the diet of varanids occupying arid habitats was supported by our data. Invertebrates, particularly Orthoptera, provide a critical dietary source for ectothermic predators in arid environments where vertebrate prey items are in limited abundance, and are important to the diet of varanids occupying arid habitats (Risbey *et al.*, 2000; Paltridge, 2002; Rouag *et al.*, 2007). The dietary overlap between each of our study species, *V. tristis*, *V. gouldii*, and *V. panoptes* is high, however all three species are generalist feeders and the consumption of high proportions of invertebrate prey has likely driven their success in low productivity habitats (Pianka, 1981; Stafford Smith and Morton, 1990). Understanding diet and feeding ecology of sympatric species is key to determining differential habitat and resource use, and in understanding population persistence in challenging environmental conditions.

3.6 References

Every reasonable effort has been made to acknowledge the owners of the copyright material. I would be pleased to hear from any copyright owner who has been omitted or incorrectly acknowledged.

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Chapter 4. Build it and some may come: restoration of discontinued mine sites initially favours herbivores

The study presented in this chapter is in preparation for submission to review within the peer-reviewed literature.

Cross, S.L, Craig, M.D, Tomlinson, S. and Bateman, P.W. Build it and some may come: restoration of discontinued mine sites initially favours herbivores. *In Preparation*.

Perentie (*Varanus giganteus*)

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4.1 Abstract

Rates of habitat destruction are increasing globally, and recent years have seen a growing focus on returning lands degraded through anthropogenic impacts to functional and sustainable ecosystems. Animals provide a range of ecological services and are critical to healthy ecosystems, yet in assessments of restoration progress animals are often overlooked and assumed to return following the return of vegetation. We used remote sensing camera traps to assess fauna community assemblages in restored and reference vegetation at a mine site in the Mid West region of Western Australia. We aimed to assess the direct and indirect impacts of mining activities on the detection likelihood of fauna, and whether animal communities (birds, mammals, and reptiles) or foraging guilds (granivores, herbivores, insectivores, omnivores, or carnivores) differed between early successional stage restored vegetation and unmined reference vegetation. Habitat restoration appeared to facilitate the return of a similar diversity of species to that found within reference vegetation. However, the restored vegetation supported different foraging guilds than those found in the reference vegetation. Early stage restoration appeared to be particularly attractive to herbivores but may lack some key resources necessary for the return of granivores, insectivores, and omnivores. Proximity of sites to the active mining operation did not appear to significantly impact animal detections. Failure to quantify the responses of the faunal community to restoration may conceal a fundamental failure in ecological restoration to establish functional community structure. Furthermore, the resulting community imbalance at the landscape scale may have widespread influences on the dynamics of surrounding, natural vegetation.

4.2 Introduction

Rates of habitat degradation and destruction are increasing globally, largely as a result of anthropogenic influences such as agriculture, forestry, mining, and urbanisation (Fahrig, 1997; Lande, 1998; Tilman *et al.*, 2001). Ecosystem functionality is intrinsically linked to the interactions of animals with their environment (Gagic *et al.*, 2015; Cross *et al.*, 2020a). Fauna from a range of trophic levels provide critical ecological services; for example earthworms and termites aid in soil decomposition and nutrient cycling (Reichle, 1977; Jouquet *et al.*, 2006), ants, birds, and some reptile species assist in seed dispersal and pollination (Frick *et al.*, 2014; Whelan *et al.*, 2015; Valido and Olsen, 2007), and apex and mesopredators regulate predator-prey dynamics (Mace *et al.*, 2012). Despite playing crucial roles in ecosystems, animals are often assumed to return passively to restored habitats following the return of vegetation ('build it and they will come'; Palmer *et al.*, 1997; Cross *et al.*, 2020a). Animals remain poorly represented in assessments of habitat restoration success following a range of degrading processes (e.g., agriculture, forestry, mining; McAlpine *et al.*, 2016; Cross *et al.*, 2019a).

Functional ecosystems rely on a variety of fauna groups, and the structure of habitats can have marked impacts on both the abundance of fauna classes and of foraging guilds (Zanette *et al.*, 2000; Razeng and Watson, 2015; Cross *et al.*, 2020a). Changes to the structure of fauna communities or foraging guilds, e.g., losses of high order predators and overabundance of subordinate species, could trigger trophic cascades and result in adverse ecological impacts (Post *et al.*, 1999; Polis *et al.*, 2000; Miller *et al.*, 2001). Vegetation can require long time periods to become established and resemble pre-disturbed communities (Grant and Loneragan, 1999; Tuff *et al.*, 2016). Restored landscapes often the lack spatial heterogeneity and comprise vegetation at earlier successional stages than undisturbed landscapes (Pywell *et al*., 2002; Baer *et al*., 2004; Cross *et al.*, 2020b), and the metabolic costs associated with use of open, homogenous landscapes can be high (Tuff *et al*., 2016). Structural heterogeneity and an abundance of microclimates and refuges in habitats is key to fauna return, particularly for ectothermic species (Tuff *et al.*, 2016, Cross *et al.*, 2020b). While species with a generalist diet may respond positively to restoration, species with more specific habitat and dietary requirements may be adversely influenced by habitat disturbance or restoration. Restored landscapes (particularly those in early stage restoration) may be unable to support species from guilds reliant upon structurally complex vegetation cover, such as those foraging or nesting in the canopy or mid-storey (Davis *et al.*, 2000; Craig *et al.*, 2015). For example, Lindenmayer *et al.* (2012) found birds occupying mid-storey and canopy niches to be most abundant within old growth woodland habitats, where tree hollows were abundant. In comparison, restored plots supported higher populations of conservation significant species, due to a higher prevalence of seedling regrowth (Lindenmayer *et al.* 2012). Restored areas may facilitate overpopulation of some groups, such as herbivores, which may be successful in exploiting resources such as seedlings and vegetation in early successional stages (Letnic and Crowther, 2013). The increased grazing pressures from herbivore species can inhibit seedling recruitment and the establishment of vegetation and reduce the survivorship of plants (Kettenring *et al.*, 2009; Letnic and Crowther, 2013).

As indicated by the radiating effects of changes in the faunal community structure, anthropogenic disturbances can have numerous indirect ecological effects. Some of these effects on animal communities, such as noise, dust, and light pollution, vibrations from heavy machinery and road use, and altered microclimates (Forman *et al.*, 2003; Raiter *et al.*, 2014) can be difficult to quantify. Such "enigmatic effects" are rarely considered in assessments of the impact of human development on animal communities and ecosystem functionality (Raiter *et al.*, 2014). While enigmatic effects can be difficult to quantify and detect, cumulatively their influence on fauna can be significant (Therivel and Ross, 2007; Canter and Ross, 2010; Raiter *et al.*, 2014). Habitats near mining activities are likely to experience increased disruption, since heavy machinery, increased traffic and road use, and blasting operations create significant noise and ground vibrations in the areas immediately surrounding mining operations (Folchi, 2003; Tripathy, 2008). These effects can significantly alter the movement, foraging, and mating behaviour of animals (Longcore and Rich, 2004; Tyler *et al.*, 2014; Raiter *et al.*, 2014), often leading them to avoid these stimuli. This avoidance may lead to animals failing to fulfil ecological roles in landscapes immediately surrounding disturbances.

Assessing the responses of animal communities (e.g., the bird, mammal, and reptile species occupying a habitat) to habitat restoration is crucial to ensuring restoration efforts are effectively returning functional and self-sustaining ecosystems. Here, we use remote sensing camera traps, an effective tool for long-term monitoring of animal communities (McDonald *et al.*, 2015), to assess the responses of animal communities to the direct and indirect effects of mining at a site in the Mid West region of Western Australia. We aim to assess i) whether the land management practice (i.e. restoration) and proximity of sites to the active mining operation influence the detectability of animals (birds, mammals, or reptiles); ii) whether early stage restoration of discontinued mine sites supports animal communities with similar composition and foraging guilds (granivore, herbivore, insectivore, omnivore, or carnivore) to the reference, unmined bushland, iii) if the structure of foraging guilds changes with proximity to the active mine, and iv) whether restoration areas in early successional stages present landscapes with higher temperatures (and hence potential increased metabolic costs) than the surrounding reference vegetation.

4.3 Methods

4.3.1 Study sites

Study sites were located within the tenement of an active mining operation in the Mid West region of Western Australia, ~430km northeast of Perth (29°08'50"S, 116°49'07"E). The study region experiences an arid climate with an average yearly rainfall of ~300mm, the majority of which falls between May and August (Australian Bureau of Meteorology, http://www.bom.gov.au/climate/data). The region has been extensively altered through several anthropogenic disturbances including rangelands agriculture and mineral extraction. Magnetite mining is the most intensive current disturbance in the landscape, with remedial action post-discontinuation a legislative requirement (Gilbert, 2000). We selected three sites of varying proximity to the active mining operation: Exploration Village Bushland (EVB), Blue Hills North (BHN), and Terapod (TP), located 3, 8, and 12km from the active mine pit, respectively. EVB comprised reference (unmined) vegetation only, and BHN and TP comprised a restored waste rock dump (~800 x 500m) surrounded by adjacent unmined reference vegetation. Vegetation within the study region largely comprised open *Eucalypt* woodlands and *Acacia* shrublands (Bamford, 2006). Restoration within each site commenced in May 2014 with the completion of all works and seeding by July 2017. Vegetation within restoration sites comprised species from the reference bushland; however, communities were at earlier successional stages than that of the unmined, reference vegetation (Fig. 4.1). Refuge piles composed of piles of sand and woody debris were incorporated into sites at the base of the restored waste rock dump along the edge of the restoration footprint (~400m length, 3m width, 3m height).

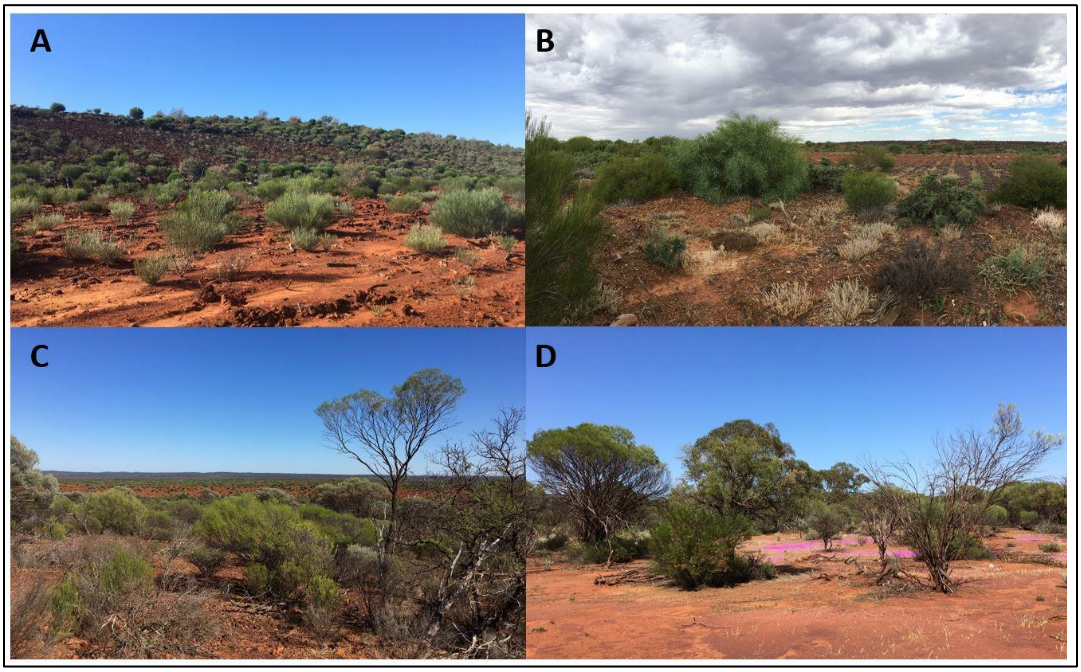

Fig. 4.1: Typical vegetation structure in study sites: A) across the base and slopes of the restored waste rock dump, B) over the top of the restored waste rock dump, C) reference shrubland communities, and D) open woodland communities in the reference vegetation.

4.3.2 Survey design

We established a network of HC600 remote sensing cameras (Reconyx Inc., United States) over the austral spring from September to November in 2017 and 2018. We assessed fauna community assemblages over spring to increase the likelihood of capturing reptiles, which are typically active in warmer months. Cameras were programmed to trigger at their highest sensitivity, capturing three pictures per trigger. To maximise the capture rate of animals, particularly for species which may be difficult to detect such as small-bodied or fast-moving animals, we set cameras to record with no time delay between triggers (Meek *et al.*, 2014; Trolliet *et al.*, 2014). Cameras were tied to trees or metal stakes and set roughly 50cm above the ground to maximise the likelihood of capturing small-bodied mammals and reptiles (Meek *et al.*, 2012).

Trapping grids comprised 20 cameras in 2017 (4 x 5 grid), and 25 cameras in 2018 (5 x 5 grid), with roughly 100m spacing between transects and camera trapping points (Fig 4.2). The total area surveyed was ~25ha within each site. Each transect ran a length of 500m, with transects in BHN and TP extending from the reference vegetation into the restoration area, such that the proportion of area surveyed within each of the reference and restoration vegetation was roughly equal. As EVB comprised only reference vegetation, we excluded this site from analyses between reference and restoration vegetation. We surveyed each site consecutively for a period of 21 days, with transects shifted ~50m along the restoration footprint at the midpoint of trapping within each site to maximise the number of trapping points surveyed within each site (Rovero *et al.*, 2013). Each site ultimately contained 40 trapping points in 2017, and 50 trapping points in 2018. We surveyed a combined total of 270 trap points over 54 transects, with a total trapping effort of 5,670 days (136,080 hrs).

We set EasyLog USB temperature loggers (Lascar Electronics Ltd., UK) at each individual camera trapping point in BHN and TP to determine if there were any differences in the thermal environments of reference and restoration sites. Loggers were suspended in open ended PVC tubes set ~50cm above the ground and were programmed to record temperatures every 15 minutes over the course of trapping at each site. We set additional data loggers at the lowest and highest elevation points in each of the reference and restoration vegetation to determine whether the increased elevation of the restored waste rock dump influenced the thermal environment.

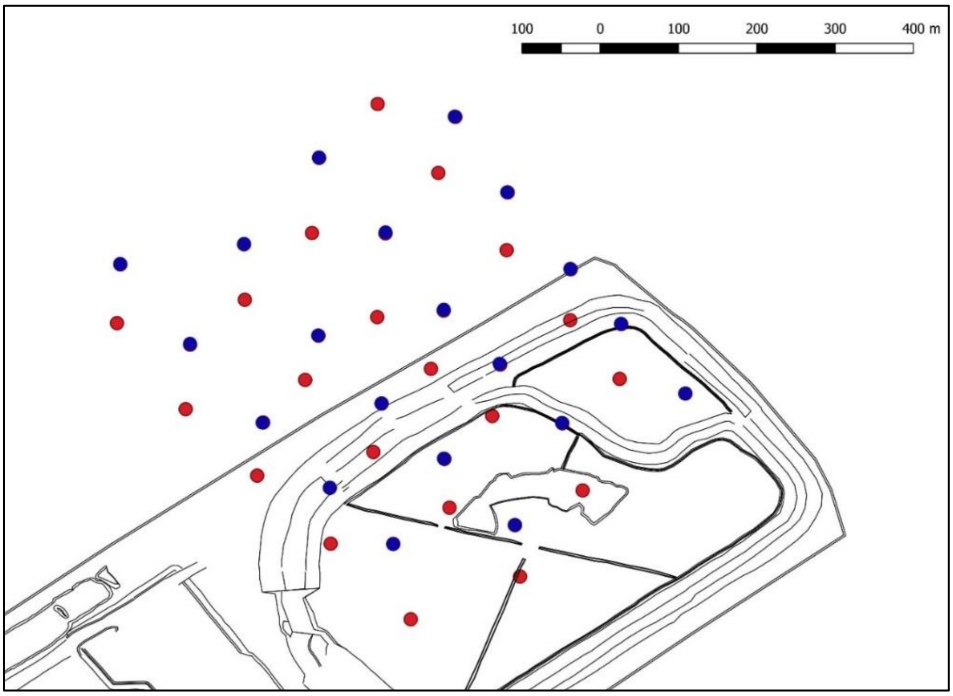

Fig. 4.2: Typical camera trapping layout as shown at the site 8km (BHN) from the active mining operation during 2017. First camera placements are shown in red and new locations following movement of cameras at the midpoint of trapping are shown

in blue. Marked outlines represent the footprint of the restored waste rock dump. Reference vegetation abuts and surrounds the restored area.

4.3.3 Statistical analyses

4.3.3.1 Species diversity

For each instance where a camera captured an image of an animal (trapping events), we identified each animal to genus, or species where possible (n = 223/439, 50.7% of records identified to species). False triggers (primarily caused by wind and vegetation) were discarded. We considered detections at each trapping point to be independent of one another if a time period of 15 minutes or greater had elapsed between images (Dorning and Harris, 2019a, b), or where animals showed clear morphological differences from each other. For each detection we recorded the site (3km; EVB, 8km; BHN, or 12km; TP), habitat (reference or restoration), species, and foraging guild of the species. We calculated Simpson's index (*D*) following Anandan *et al.* (2014), for species recorded in reference and restoration vegetation, and for proximity to mining analyses between EV, BHN, and TP, using:

$$D = \frac{\sum n(n-1)}{N(N-1)} \tag{1}$$

Where *n* = the total number of individuals of each species recorded, and *N* = the total number of all individuals recorded within each vegetation type. Simpson's Index accounts for species richness and evenness and provides a measure of diversity, where *D* ranges between 0 and 1, with 0 representing infinite diversity, and 1 representing no diversity (Anandan *et al.*, 2014; Biranvand *et al.*, 2014). We then calculated both Simpson's Index of Diversity (*SID*) and Simpson's Reciprocal Index (*SRI*) following Biranvand *et al.* (2014), using:

$$SID = 1 - D \tag{2}$$

$$SRI = \frac{1}{D} \tag{3}$$

SID is a measure of sample diversity and ranges between 0 and 1; however, a higher value of *SID* indicates greater diversity within samples, i.e., 0 indicates no diversity (Biranvand *et al.*, 2014). *SRI* is a measure of community diversity, with 1 being the lowest possible value (i.e., a community comprising one species), and increasing values of *D* indicating greater species diversity (Biranvand *et al.*, 2014).

4.3.3.2 Reference and restoration vegetation

We constructed two separate Generalised Linear Models with a Poisson distribution to determine differences in the abundance (number of individuals) of each species and foraging guild (e.g., granivore, herbivore, insectivore, omnivore, carnivore) between trapping years (2017, 2018), habitats (reference or restoration), and sites (BHN, TP). Species were grouped into five foraging guilds (see Appendix 3, Table A3.1): terrestrial insectivores (*n* = 7/20 species), terrestrial granivores (*n* = 4 species), terrestrial herbivores, terrestrial omnivores, and terrestrial carnivores (*n* = 3 species each). We analysed whether detections within each foraging guild were likely to be predicted by trapping year, site, and habitat quality at the point of detection, by performing separate binomial logistic regressions with each foraging guild as the dependent variable, and year, site, and habitat quality as the independent variables. Finally, we constructed a between-site Bray-Curtis similarity matrix for species detections between reference and restoration, visually represented using a non-metric multidimensional scaling (nMDS) and compared differences between foraging guilds using analysis of similarity (ANOSIM) in Primer 6.0 (Primer-E, 2006).

We conducted two separate two-way ANOVA to examine i) whether temperatures differed between site (BHN or TP) or habitat (reference or restoration), and ii) whether temperature differed between the highest and lowest elevation points within each site and habitat. As we were interested in the potential thermal costs of using landscapes with increased homogeneity and vegetation at early successional stages, we assessed differences in daily temperatures between reference and restoration sites (0600-1800 hrs). As EVB comprised only reference vegetation, we excluded this site from temperature analyses. The distribution of data was examined using frequency histograms, with residual and QQ plots for appropriateness of the model. Temperature data were analysed using R Studio v.1.1.383 (R Development Core Team, 2013). Data were reported as means ± 1 standard error of measurement (S.E.) unless otherwise stated.

4.3.3.3 Proximity to mining

As with analyses for detections in reference and restoration vegetation, we constructed two separate Generalised Linear Models with a Poisson distribution to determine the relationship between the abundance of species or foraging guild, and the proximity of sites to mining activities (3km, 8km, or 12km), with counts of species or foraging guild as the dependent variable and proximity to mining activities as the independent variable. Finally, we constructed a between-site Bray-Curtis similarity matrix for species detections between each site, visually represented using a non-metric multidimensional scaling (nMDS). We compared differences between foraging guilds using analysis of similarity (ANOSIM). All analyses of trapping events comparing reference and restoration vegetation, and proximity to mining were conducted using SPSS Statistics 25 (IBM, New York, United States).

4.4 Results

4.4.1 Trapping overview

We detected a total of 20 animal taxa (*Macropus* spp. and *Corvus* spp. each grouped as single entities) over 439 trapping events in EV, BHN, and TP. Mammals triggered 281 captures (64% of all captures), followed by 153 bird captures (35%), and five reptile captures (1%). Two of the mammalian species detected were invasive species; feral cat (*Felis catus*), and European rabbit (*Oryctolagus cuniculus*). One detection of a mouse species was captured, but it was not possible to distinguish between the invasive house mouse (*Mus musculus*) and the native sandy inland mouse (*Pseudomys hermannsburgensis*). Twenty-four percent of all captures were of invasive species (*n* = 105/439). We detected 18 species over 396 trapping events in reference and restoration vegetation, with 165 detections recorded in reference vegetation (42% of captures), and 231 detections within restoration vegetation (58% of captures). Proximity trapping surveys captured the same 18 species, and an additional two species (20 species in total) over 439 trapping events. Forty-three detections were recorded in EV (3km site; 10% of all captures), 204 in BHN (8km site; 46% of captures) and 192 in TP (12km site; 44% of captures).

4.4.2 Reference and restoration vegetation

Animals were 0.6 times less likely to be recorded within reference sites than within restoration areas (95% CI, 0.519 to 0.776, $P < 0.001$); however, 48% of all detections within restoration vegetation were mammals. We did not detect any significant differences in the total number of detections between 2017 and 2018 ($\chi 2 = 2.05$, d.f. =

1, *P* = 0.152), or between BHN and TP (χ2 = 0.08, d.f. = 1, *P* = 0.783), and the interaction between year and site was non-significant (χ2 = 490.0, d.f. = 1, *P* = 0.484). While there was minimal separation between sites evident in the nMDS, species detections differed significantly between restoration and reference vegetation (*r* = 0.104, *P* = 0.010; Fig. 4.3). Although we recorded 15 species within restoration vegetation, detections were primarily restricted to three species: *Macropus* spp. (*n* = 110/231, 48% of all captures), *Dromaius novaehollandiae* (emu; *n* = 57/231, 25% of captures), and *Oryctolagus cuniculus* (European rabbit; *n* = 44/231, 19% of captures). Detections for each of the three species accounted for 92% of all detections in restoration vegetation, in comparison to 63% of all detections in reference vegetation.

We detected all five foraging guilds in each of the reference and restoration habitats (Fig. 4.4); however, total numbers of detections for each guild differed significantly between habitats (χ2 = 32.32, d.f. = 1, *P* < 0.001), with no significant interaction between either year and habitat (χ2 = 2.98, d.f. = 1, *P* = 0.881), or site and habitat (χ2 = 158.07, d.f. = 1, *P* = 0.092). Herbivores were five times more likely to be recorded in the restoration vegetation than within the reference vegetation (χ2 = 99.30, d.f. = 1, *P* < 0.001). We recorded only one detection of a carnivore in each of the reference and restoration vegetation; however, detections of granivores, insectivores, and omnivores were most frequent in reference vegetation. Total detections of each foraging guild did not differ significantly between trapping years (2017, 2018; χ2 = 1.66, d.f. = 1, *P* = 0.197) or between sites (BHN or TP; χ2 = 2.71, d.f. = 1, *P* = 0.099).

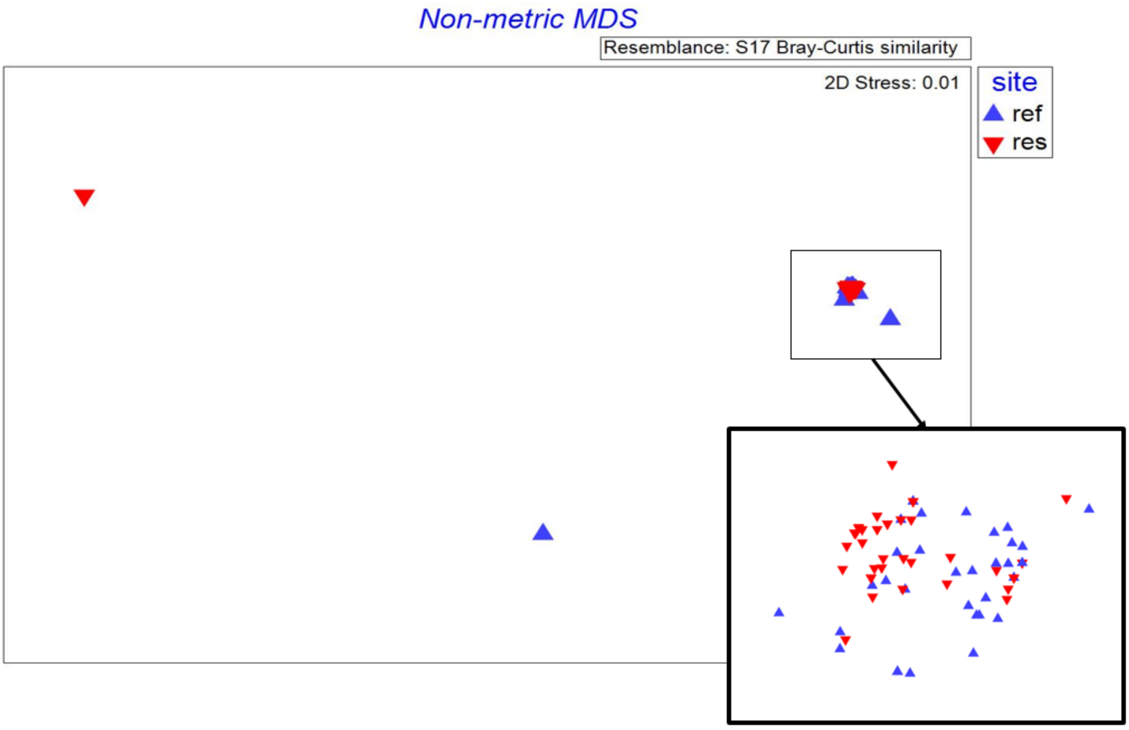

Fig. 4.3: Non-metric multidimensional scaling (nMDS) plots for the overlap in species detections between reference (▲) and restoration (▼) vegetation.

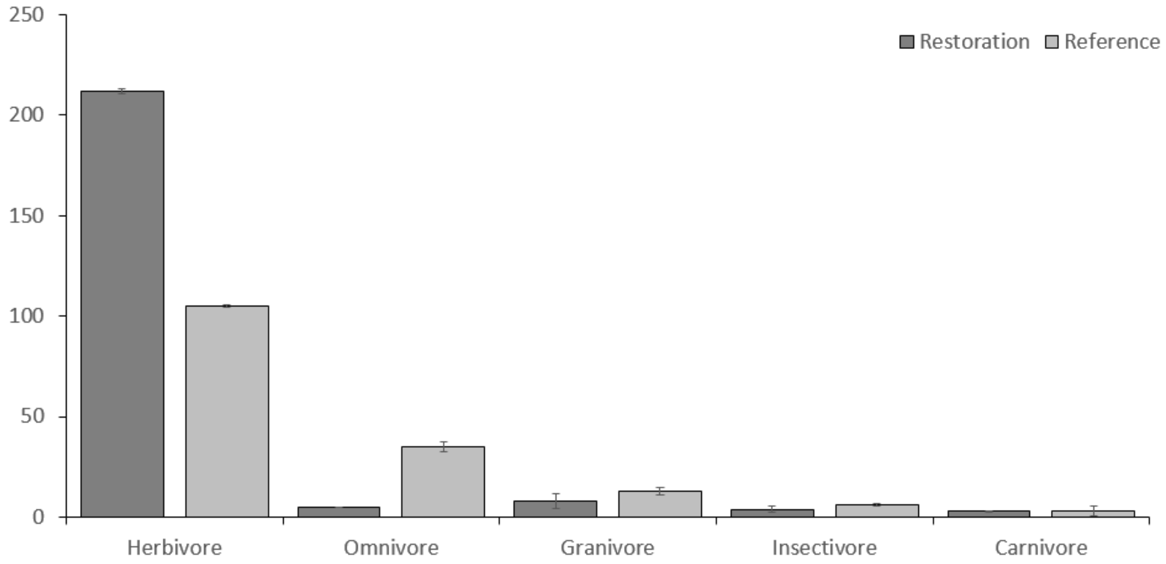

Fig. 4.4: Detections of each foraging guild recorded in reference and restoration vegetation over 2017 and 2018 trapping experiments.

Recorded temperatures ranged between 1.0 – 49.9°C in the reference vegetation, and 2.0 – 49.9 °C in the restoration vegetation, with an average temperature of 26.2°C ± 0.03 and 26.3°C ± 0.03, respectively. Restoration and reference vegetation did not differ statistically in temperatures ($F_{(1,3)} = 66.85$, $P = 0.226$); however, recorded temperatures differed statistically between BHN and TP ($F_{(1,3)} = 69364.15$, $P < 0.001$), with a significant interaction effect of site and habitat ($F_{(1,3)} = 6718.83$, $P < 0.001$). Temperatures at the highest and lowest elevation within sites averaged 26.8°C ± 0.16 and 27.4°C ± 0.19 in reference vegetation respectively, and 26.1°C ± 0.18 and 26.72 °C ± 0.14 in restoration vegetation. We did not record any significant differences in temperatures between the lowest and highest points of elevation either between sites ($F_{(1,1)} = 124.21$, $P = 0.596$), or between reference and restoration vegetation ($F_{(1,1)} = 794.45$, $P = 0.661$), with no significant interaction effect between site and elevation ($F_{(1,1)} = 613.32$, $P = 0.853$), or habitat and elevation ($F_{(1,1)} = 525.27$, $P = 0.863$).

4.4.3 Proximity to mining

Proximity of sites to active mining activities did not significantly affect animal detection likelihood when distance to mine was included as a linear variable ($\chi^2 = 2.75$, d.f. = 1, $P = 0.097$). However, when included as a non-linear variable, animals were significantly less likely to be detected at the site in closest proximity to the active mine (3km), than both the sites 8km (0.5 times less likely; $\chi^2 = 13.88$, d.f. = 1, $P < 0.001$) and 12km (0.6 times less likely; $\chi^2 = 11.18$, d.f. = 1, $P < 0.001$) from active mining activities. We did not detect any significant differences in the total number of detections between trapping years (2017 and 2018; $\chi^2 = 2.89$, d.f. = 1, $P = 0.089$), and we did not record any interaction effect between year and proximity to mining ($\chi^2 = 2.068$, d.f. = 2, $P = 0.356$). Detections for each species differed significantly between sites, with detections significantly less likely to be recorded within the 3km site than the sites 8km and 12km from the active mine ($\chi2 = 35.72$, d.f. = 2, $P < 0.001$). While there was minimal separation between sites evident in the nMDS, species detections differed significantly between sites ($r = 0.062$, $P = 0.010$; Fig. 4.5).

Apart from the site 3km from the active mining operation, which lacked carnivores, we detected all foraging guilds across all sites (Fig. 4.6). While the overall detections did not differ between the 8km and 12km sites, total detections for carnivores ($\chi2 = 12.17$, d.f. = 1, $P < 0.01$), herbivores ($\chi2 = 156.052$, d.f. = 1, $P < 0.001$), and insectivores ($\chi2 = 7.85$, d.f. = 1, $P < 0.005$) differed significantly between each site. Herbivores occurred most frequently within the 8km site; however, both carnivores and insectivores were recorded most frequently in the 12km site. Only carnivore ($\chi2 =

2.36, d.f. = 2, *P* = 0.307) detections were not significantly predicted by proximity to mining. Detections for each of the remaining classes were significantly more likely to occur within the sites 8km and 12km from the active mining operations (omnivores: $\chi 2 = 6.93$, d.f. = 2, $P = 0.031$; granivores: $\chi 2 = 9.18$, d.f. = 2, $P = 0.010$; insectivores: $\chi 2 = 20.78$, d.f. = 2, $P < 0.001$; and herbivores: $\chi 2 = 30.38$, d.f. = 2, $P < 0.010$).

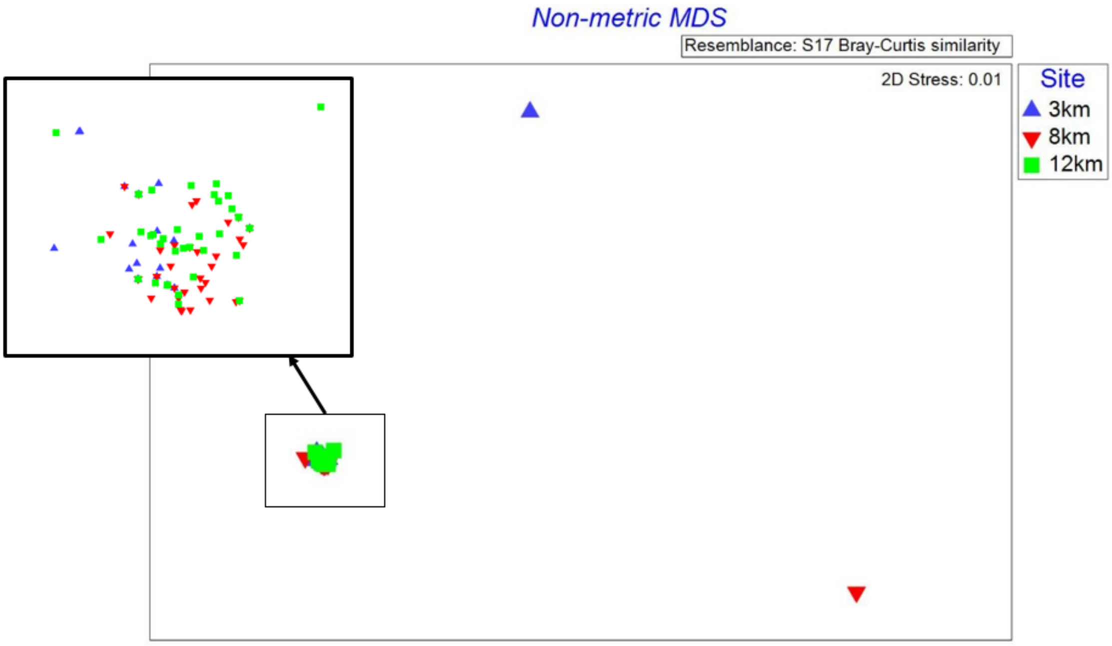

Fig. 4.5: Non-metric multidimensional scaling (nMDS) plots for the overlap in species detections between sites of varying proximity to the active mining operations (3km, 8km, 12km from the active mining operation).

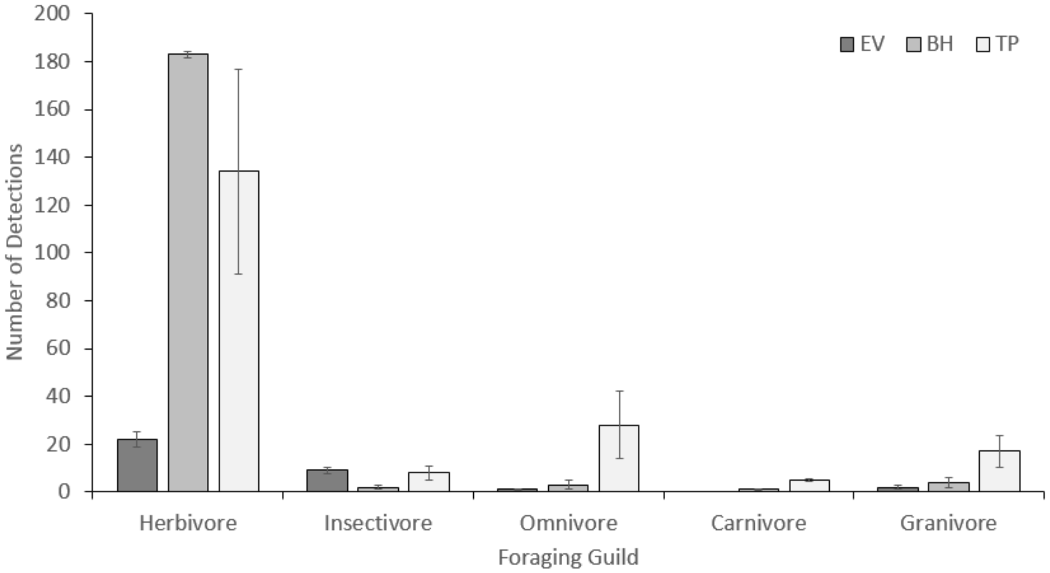

Fig. 4.6: Detections of each foraging guild recorded in EVB, BHN, and TP over 2017 and 2018 trapping experiments.

4.5 Discussion

Early stage restoration within the study area is facilitating the return of animals at a similar species diversity to the reference bushland. While restoration sites are aligning along a trajectory towards a state comparable with the reference bushland, early stage restoration facilitated disproportional return of some foraging guilds in comparison to the reference vegetation. Restoration appears to be particularly effective for herbivores, predominantly macropods; however, these areas were used infrequently by granivores, insectivores, and omnivores. It is likely that habitats in early stage restoration lack some key resources necessary for the return of representative faunal communities. Community structure of animals may further align towards a state comparable to the reference bushland as vegetation becomes established. We infrequently detected carnivores across all sites; however, within the arid regions of Australia the monitor lizards often fill apex predator roles in the broad absence of apex mammalian species (Read and Scoleri, 2015; Cross *et al.*, 2019b). Low detection likelihood may be an inherent limitation for the application of camera traps in reptile detection (Hobbs and Brehme, 2017; Dundas *et al.*, 2019). While we were limited in our ability to make inferences about the return of carnivorous species to habitats undergoing restoration, previous research has highlighted distinct disparities in the numbers of reptiles recorded in restored landscapes in comparison to within the reference bushland (Cunningham *et al.*, 2007; Michael *et al.*, 2011, 2018). We did not note any significant impact of the proximity of sites to active mining on the structure of animal communities.

4.5.1 Usage of reference and restoration vegetation

Altered vegetation structure and community dynamics can influence the abundance, richness, and fitness of vertebrate fauna within ecosystems (Olsson *et al.*, 2002; Lindell, 2008), and we detected significant differences in the abundances of each foraging guild between reference and restoration vegetation. Our interpretation that early stage restoration is facilitating the return of fauna communities is predicated on two assumptions: first, that we adequately surveyed the fauna community using our camera traps; and second, that there is a substantial difference in habitat quality between the restoration areas and the surrounding habitat. Further research into of microhabitat metrics (e.g., density of woody debris, availability of refuges) and vegetation structure may provide an increased understanding of the quality of restored landscapes. While our inferences may be limited by a low replication of restored sites, we conclude that early stage restoration efforts are facilitating the return of fauna communities. However, these areas may be less suited to use by non-herbivorous fauna. Our capacity to draw any solid conclusions on either of these speculations, however, is limited by the detectability of many key fauna groups using camera traps.

Although animals were more likely to be detected within restored areas, these areas facilitated a high level of use by herbivores such as macropods. Macropods are a dominant herbivore group in Australia and tend to occur at high density in landscapes where primary productivity is high and seedlings are abundant (Letnic and Crowther, 2013). As restored landscapes often comprise vegetation at early successional stages (e.g., seedlings and saplings; Pywell *et al.*, 2002; Baer *et al.*, 2004), these areas may be more likely to attract a high density of macropods through increased availability of grazing resources. Habitat choice may be influenced by the availability of food resources; however, increased presence of macropods within restoration habitats may also represent a lower perceived predation risk in these habitats (Chace and Goldizen, 2003). Macropods have previously been shown to select foraging sites based upon food availability and predation risk, and open landscapes with high visibility present lower predation risks (Carter and Goldizen, 2003; Blumstein *et al.*, 2003). The high mobility of macropods and increased visibility in restored landscapes may allow for opportunistic use of such areas, and population density recorded in these areas may be artificially high. Overabundance of fauna in landscapes can create imbalances in ecosystems and result in altered trophic structure and ecosystem function, and losses of species from habitats (Lumney *et al.*, 2007; Yugovic, 2019). These imbalances may adversely impact community dynamics of fauna both within restored landscapes and in the surrounding undisturbed landscape. The imbalance of herbivores in restored landscapes may be problematic at multiple spatial and trophic levels, and increased use of restored landscapes by macropods may present issues for conservation and management of landscapes undergoing restoration. For example, increased gazing pressures may inhibit seedling recruitment and the recovery and re-establishment of native vegetation (Meers and Adams, 2003). Overconsumption of vegetation and herbivore imbalance may alter habitats and result in losses of both flora and the fauna species reliant on them (Yugovic, 2019). Unfortunately, there is very little empirical understanding of the carrying capacity of arid ecosystems for kangaroos (but see Cairns and Grigg, 1993 and Meers and Adams, 2003), especially in Western Australia.

Spatially heterogenous and structurally diverse habitats are key for many species (Adamík *et al.*, 2003; Tews *et al.*, 2004; Cross *et al.*, 2020b). We did not note any significant differences in the thermal landscape between reference and restoration vegetation. However, spatially homogenous landscapes can present high metabolic costs for animals, even in cases where temperatures between landscapes do not differ significantly (Sears *et al.*, 2011; Cross *et al.*, 2020b). Restoration vegetation was used infrequently by granivorous, insectivorous, and omnivorous species. Seeds and invertebrates are staple food resources in arid environments (Brown *et al.*, 1979; Morton, 1979; Stafford Smith and Morton, 1990; Cross *et al.* 2019b). Habitats with early successional stage vegetation are unlikely to contain abundant seed resources, or insects, which are often associated with leaf litter and microclimate abundance (Şekercioḡlu *et al.*, 2002). Granivorous species rely upon seed production from a variety of perennial and annual plant species (Brown *et al.*, 1979; Price and Joyner, 1997), and early stage restoration is unlikely to contain resources suitable for supporting granivorous populations. Coarse woody debris is a vital component of ecosystems, providing critical habitat for many invertebrate and vertebrate species (Jacobs *et al.*, 2007; Christie *et al.*, 2012; Craig *et al.*, 2014). Coarse woody debris requires a considerable length of time to develop and is often a limited resource in habitats undergoing restoration (Craig *et al.*, 2014). Species reliant on woody debris, such as insectivores, may be limited in their ability to utilise habitats undergoing restoration, particularly those in early stage restoration.

4.5.2 Enigmatic impacts on animal detection likelihood

We did not record a strong relationship between the proximity of sites to active mining activities and the detection likelihood of species or foraging guilds. The enigmatic impacts of mining may cause significant disturbances to animal communities in close proximity to human disturbances (Folchi, 2003; Longcore and Rich, 2004; Tripathy, 2008; Tyler *et al.*, 2014; Raiter *et al.*, 2014), and there is some indication that detections of omnivores and granivores may be influenced by proximity to disturbance. However, differences between sites appear to be primarily driven by differences in the detection of herbivorous species. We detected herbivores significantly more frequently in the sites 8km and 12km from the active mining operation; however, both sites comprised restoration habitat in addition to reference bushland. Herbivores are typically abundant within sites with increased primary productivity and grazing resources (Letnic and Crowther, 2013). As restoration habitat largely comprised vegetation in early successional stages (Pywell *et al.*, 2002; Baer *et al.*, 2004), these sites are likely to contain an abundance of grazing resources and hence support a higher density of herbivore species.

Although we do not note any strong influence of proximity of sites to disturbance on fauna communities, previous research has highlighted significant influences of human disturbance on animal behaviour, movement, and fitness (Longcore and Rich, 2004). For example, light pollution alters the activity periods, mating, and foraging behaviours of species typically diurnally active (Schwartz and Henderson, 1991; Derrickson, 1988). Although altered activity periods can be beneficial to species foraging in artificial light, prey species are often adversely affected by altered activity resulting from anthropogenic light sources (Longcore and Rich, 2004). Future research should consider the influence of the enigmatic effects of mining and other disturbances, such as noise, dust, and vibrations on fauna communities and detection likelihood of animals. Understanding the effects of habitat disturbance that may alter the behaviour, community structure, and abundance of fauna within habitats is critical to determining whether habitats are supporting self-sustaining, functional, and representative fauna populations.

4.5.4 Conclusions

Habitat restoration following the discontinuation of mining activities within the study area results in a similarly rich and diverse fauna community. However, early stage restoration heavily favours dominant herbivore species and species well-adapted to anthropogenically modified landscapes. Restoration appears to be particularly successful for returning macropod species, but these landscapes may facilitate overpopulation and overgrazing through an increased prevalence of seedlings within these habitats (Pywell *et al.*, 2002; Baer *et al.*, 2004; Letnic and Crowther, 2013). Overabundance of certain fauna groups or foraging guilds may trigger trophic cascades and community imbalances, and ultimately result in non-functional and unsustainable ecosystems (Lumney *et al.*, 2007; Yugovic, 2019). Restored landscapes may lack some key resources necessary for the return of representative animal communities, for example coarse woody debris and consequently microclimates, refuges, food, and spatial heterogeneity (Craig *et al.*, 2014; Tuff *et al.*, 2016; Cross *et al.*, 2020b). Coarse woody debris was incorporated into restoration sites following revegetation; however, increasing the availability of woody debris and refuges within areas undergoing restoration may further aid the return of fauna to restored landscapes (e.g., Manning *et al.*, 2013). Some critical resources may re-establish naturally with time but recovering representative and established vegetation communities and ecological processes may require decades (Grant and Loneragan, 1999; Munro *et al.*, 2012). Assessing the responses of fauna from a range of guilds and trophic levels to habitat restoration over a range of spatial scales is critical to determining whether habitat restoration is effectively returning functional and self-sustaining animal communities.

4.6 References

Every reasonable effort has been made to acknowledge the owners of the copyright material. I would be pleased to hear from any copyright owner who has been omitted or incorrectly acknowledged.

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Chapter 5. Using monitors to monitor ecological restoration: presence may not indicate persistence

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'Kev' the resident yellow spotted monitor (*Varanus panoptes*)

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5.1 Abstract

Habitat loss is a leading cause of biodiversity declines globally, and there has been increasing recognition in recent years of the importance of restoring degraded habitats to functional ecosystems to ameliorate this loss. Despite the critical roles animals play in ecosystems, animals are often overlooked in assessments of ecological restoration success, particularly beyond their presence or absence in these habitats. Apex predators are critical to ecosystems, regulating predator-prey dynamics, and in arid Australia, monitor lizards (Reptilia: Varanidae) often fill high order predatory roles. Varanids are highly diverse in size and occupy a variety of ecological niches, providing an ideal group for assessing habitat change over multiple spatial scales. Here, we assess the responses of varanids to early stage habitat restoration following the discontinuation of mining activities, by mapping behavioural signs of habitat usage including burrows, tracks, and diggings. We recorded burrow size and track measurements to gauge the size of varanids utilising reference and restored habitats, and mapped tortuosity of tracks to assess their movement through habitats. Restored areas had significantly fewer signs of varanid presence than the reference bushland and largely appeared to be just traversed or used only by larger individuals. Restored landscapes, particularly those in early successional stages, often lack established vegetation cover and present increased metabolic costs and predation risks. Providing fauna refuges (e.g., hollow logs) to mitigate the metabolic costs and predation risks in areas undergoing restoration may aid in facilitating the return of varanids and of other animal populations, particularly during the early stages of vegetation establishment. Understanding the behavioural responses and movement ecology of animals within landscapes undergoing restoration is key to facilitating the conservation of self-sustaining and functional ecosystems.

5.2 Introduction

Habitat loss and degradation through anthropogenic activities such as agriculture, urbanisation, and mining, is a leading driver of species extinctions worldwide (Fahrig, 1997; Lande, 1998; Tilman *et al.*, 2001; Cristescu *et al.*, 2012). Consequently, there has been growing recognition of the importance of returning degraded habitats to pre-disturbance conditions (Standards Reference Group SERA, 2017; Miller *et al.*, 2017;

Gann *et al.*, 2019; Cross *et al.*, 2020a). Historically, there has been an emphasis placed upon assessing vegetation structure and community dynamics in post-restoration monitoring (Ruiz-Jaen and Mitchell Aide, 2005; Koch *et al.*, 2010). Consequently, animal taxa are often overlooked in assessments of restoration success (Lindell, 2008; Cross *et al.*, 2019a, 2020) despite their role in providing critical ecological services, such as soil decomposition (Jouquet *et al.*, 2006; Lavelle *et al.*, 2006), pollination (Phillips *et al.*, 2010; Menz *et al.*, 2014; Frick *et al.*, 2014), and regulation of predator-prey dynamics (Mace *et al.*, 2012).

The 'Field of Dreams' hypothesis, or 'build it and they will come' (Palmer *et al.*, 1997), assumes that animal taxa will return to habitats following the restoration of vegetation (Block *et al.*, 2001; Cristescu *et al.*, 2012; Cross *et al.*, 2020a). Few studies, however, have demonstrated that the return of vegetation and habitat structure promote the unassisted return of fauna, or their re-integration into ecological processes, to a level comparable to that of the pre-disturbance habitats (Hilderbrand *et al.*, 2005; Cross *et al.*, 2020a). Among existing studies of fauna responses to restoration there is a strong focus towards assessments of species' presence, absence, or abundance within restored habitats (Lindell, 2008; Cross *et al.*, 2019). While providing important tools for assessing population dynamics and habitat quality (Mackenzie, 2005), such studies risk failing to provide sufficiently detailed information on the complexities of ecological interactions within habitats (Aldridge and Boyce, 2007; Lindell, 2008; Cross *et al.*, 2020a), and have a high chance of missing rare or cryptic species due to capturing only a 'snapshot' of biodiversity (Chiarucci *et al.*, 2011). Successful restoration of degraded sites may hinge on both presence and abundance of key resources in restoration (Lindell, 2008). Presence/absence studies are unlikely to identify the fundamental resource and habitat requirements that support reproductive populations, and hence be able to show whether habitat restoration is facilitating the return of self-sustaining, functional populations (Maron *et al.*, 2005; Lindell, 2008; Cross *et al.*, 2019).

Understanding the behavioural responses of animals to habitat change and restoration, and the complex environmental factors facilitating their persistence and viability in habitats, is fundamental to achieving successful restoration outcomes (Sutherland, 1998; Lindell, 2008; Hale and Swearer, 2017; Cross *et al.*, 2019, 2020a). Monitoring visible signs of the presence of animals in habitats, such as tracks, burrows, and diggings, provides an effective method for indirectly assessing behavioural and movement ecology (Gese, 2001; Jewell *et al.*, 2001; Stephens *et al.*, 2006), particularly for shy or cryptic species where visual observations can be challenging (Silveira *et al.*, 2003; Balme *et al.*, 2009). Monitoring habitat use can provide important insights into habitat quality, resource availability, and predator-prey dynamics (Lindell, 2008; Salo *et al.*, 2008; Van Beest *et al.*, 2013). Habitats with few signs of animal activity often lack fundamental resources and may impose high metabolic costs and predation risks, subsequently impacting foraging efficiency (e.g., Tomlinson *et al.*, 2017). Areas of higher foraging and burrow activity are more likely to contain an abundance of food and thermal refuges, and may have decreased predation risk and competition pressures (Lindell, 2008; Bruton *et al.*, 2016).

Movement and habitat use by animals is influenced by the perceived risks associated with their surrounding landscape (Fahrig, 2007). Within habitats perceived to be of lower quality, such as fragmented, or spatially and structurally homogenous landscapes, movement tends to be direct and without deviation to minimise time spent within these areas (Haynes and Cronin, 2006; Fahrig, 2007). Ectothermic species, such as reptiles, rely on an availability of suitable microclimates and thermal refuges for thermoregulation (Basson *et al.*, 2017; Lindell, 2008). Tracks crossing directly through a habitat with minimal tortuosity may indicate that these areas are lacking key resources (e.g., thermal refugia and less diverse microhabitats; Bruton *et al.*, 2016) and are consequently thermally unsuitable or expose individuals to high predation risk. The identification of key resources and microhabitats required to support populations with diverse demography is vital to understanding the suitability of restoration to support self-sustaining faunal populations (Craig, 2002; Craig *et al.*, 2007; Fahrig, 2007).

High order predators play critical roles in ecosystems, maintaining top-down control through predator-prey dynamics (Post *et al.*, 1999; Miller *et al.*, 2001). Declines or losses of apex predators from habitats can have significant flow-on effects in ecosystems including increased prey populations and a reduction of species diversity through competitive exclusion of subordinate species (Miller *et al.*, 2001). Within arid Australia, varanids (monitor lizards; Reptilia: Varanidae: *Varanus*) often fill high order predatory roles, occurring at relatively high species richness (Pianka, 1994; Read and Scoleri, 2015). Varanids have highly generalist diets, preying on a variety of invertebrate and vertebrate items (Losos and Greene, 1988; Cross *et al.*, 2020b). Consumption of prey from multiple trophic levels may facilitate the support of increased biodiversity through increased stability of food webs (Gross *et al.*, 2009; Bird *et al.*, 2013). Varanids occupy a wide range of habitats, including aquatic, terrestrial, and arboreal niches. This diversification has led to the largest range of sizes within a single genus of any vertebrate taxon (~20cm; *V. sparnus*, to ~3m; *V. komodoensis*; Pianka *et al.*, 2004; Doughty *et al.*, 2014). Australian varanids encompass almost the entirety of this size breadth, with the largest species, *V. giganteus*, growing to around 2.5m long (Pianka *et al.*, 2004). Due to their diverse range of body sizes and, therefore, home ranges (King *et al.*, 1989), varanids present an ideal group to monitor habitat change and restoration over a range of spatial scales.

Here, we analyse habitat use and movement by varanids within reference (unmined) and early stage restoration vegetation at a mine site in the Mid West region of Western Australia, approximately 415km northeast of Perth. We aim to assess: i) if reference and restoration sites differ in total or type of habitat usage (total of all recorded tracks, diggings, and burrows); ii) whether restoration vegetation supports burrowing and foraging behaviour (diggings), or if usage is restricted to transitory movement through these areas, and iii) whether reference and restoration vegetation present different thermal environments.

5.3 Methods

5.3.1 Study site and species

Study sites were located in the semi-arid shrubland communities within the tenement of a magnetite extraction operation in the Mid West region of Western Australia (29°11'31"S, 116°45'36"E). Five sympatric *Varanus* species co-exist within the study region; the stripe-tailed goanna (*V. caudolineatus*, arboreal, total length [TL] 0.32m), black-headed monitor (*V. tristis*, primarily arboreal, TL 0.76m), Gould's goanna (*V. gouldii*, primarily terrestrial, TL 1.2m), yellow-spotted monitor (*V. panoptes*, terrestrial, TL 1.4m), and the perentie (*V. giganteus*, terrestrial, TL 2.5m) (Wilson and Swan, 2003; Pianka *et al.*, 2004). Territoriality has not been documented among studies tracking the movement and activity of varanids (e.g., Green and King, 1978; Auffenberg, 1981; Stanner and Mendelssohn, 1987; Case and Schwaner, 1993) and home ranges of *Varanus* species often overlap considerably (King and Green, 1993). The broad niche overlap of varanids in the Mid West region of Australia suggests little interspecific exclusion (Cross *et al.*, 2020b). The study area comprised two sites previously directly impacted by mining activities, located eight and 12km from current active mining operations. Both sites were characterised by a restored waste rock dump (an area of ~800 x 500m with sandy/rocky loam soils) and were adjacent to reference bushland (unmined, largely flat landscape with sandy loam soils). Restoration of the waste rock areas in each site commenced in 2014 with completion in 2017. The dominant vegetation types within the study area are *Acacia* shrubland and open *Eucalyptus* woodland, with sandy rocky-loam soils (Bamford, 2006). Restoration sites comprise a similar species composition to the reference habitat, however the vegetation is at earlier successional stages than the reference community (Fig. 5.1a,b).

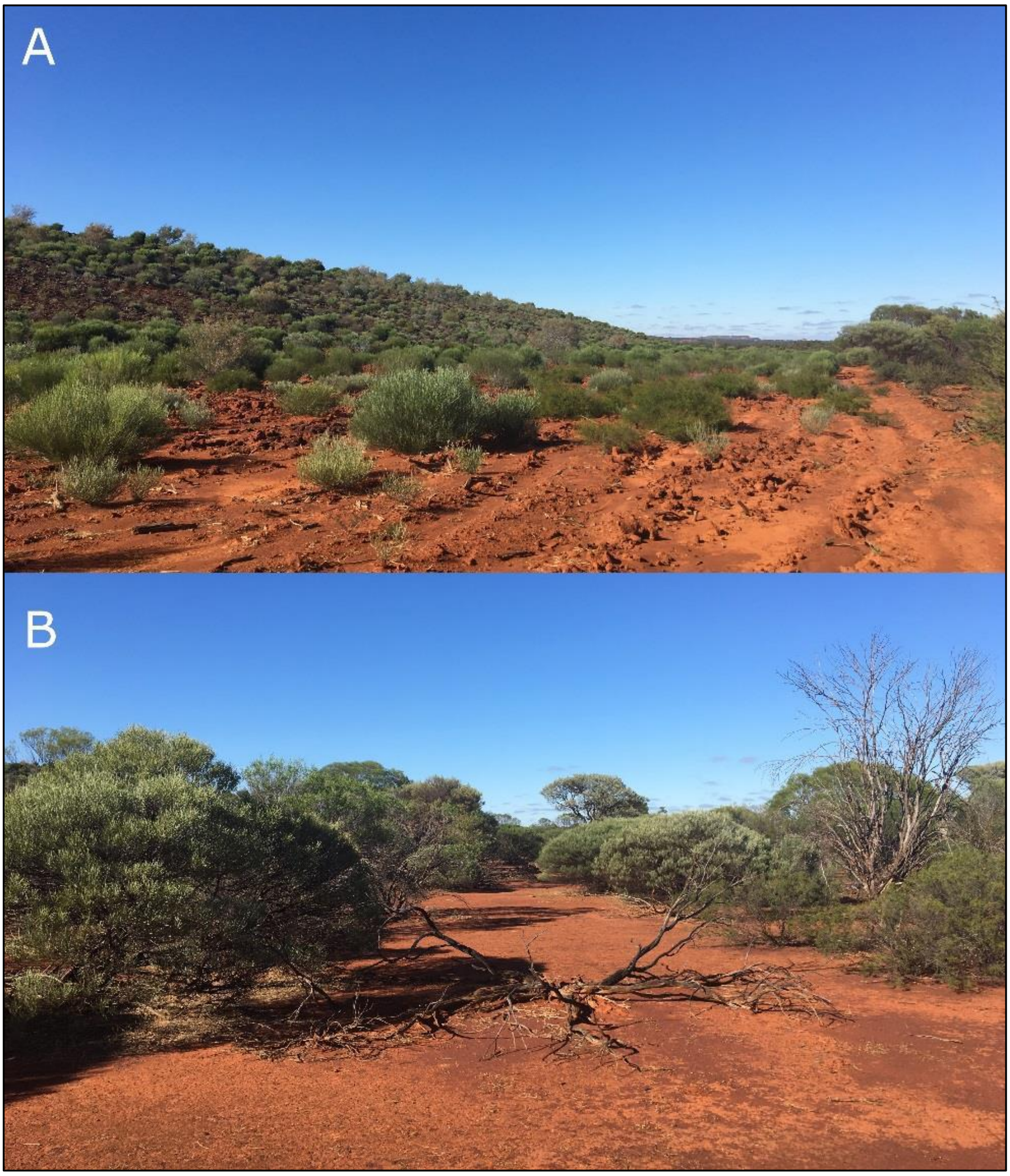

Fig. 5.1: Vegetation community structure as shown at the site 8km from active mining operations: A) restoration vegetation, where communities are early successional stages, and B) reference vegetation, where vegetation is well established, and the landscape has increased spatial and structural heterogeneity.

5.3.2 Survey design

Favoured activity temperatures for many *Varanus* species average around 35°C (King and Green, 1999), and as such, we surveyed sites consecutively between September and October 2018, where daily maximum temperatures average between 36.2 – 41.2°C (Bureau of Meteorology, http://www.bom.gov.au/climate/data/). Sites were surveyed for a total of 16 days each, with reference and restoration areas within each site surveyed concurrently. The footprint of restoration activities within each site was ~800m x 500m and we surveyed each site using marked transects spaced at 100m intervals. Each transect ran the width of the restoration area (500m) and extended the equivalent distance into the reference vegetation (1km in total). We walked two groups of five transects such that each site contained 10 transects. Each transect group was walked on alternating days, with transects in the second group also spaced 100m apart but shifted 50m further along the restoration footprint from transects in the first group. To maximise the ground covered, we surveyed a width of 25m either side of each transect (10 transects of 50m width, 1km length).

5.3.3 Thermal environment

To determine whether the thermal environment differed between reference and restoration sites, we set 10 EasyLog USB temperature loggers (Lascar Electronics Ltd., UK) in each site. We aimed to assess the general differences in ambient temperatures between sites, and as such loggers were placed randomly along transects, such that reference and restoration areas each contained five temperature loggers (five transects each with two loggers, one in each of the reference and restoration sites). Loggers were suspended ~30cm above the ground within open ended PVC tubes attached to wooden stakes to capture ambient temperature in reference and restoration sites. Loggers were programmed to record temperature at 15-minute intervals for the duration of the study period (16 days per site), such that the number of recordings per site totalled 15,360 readings (7680 temperature readings in each of the reference and restoration areas at each site).

5.3.4 Mapping habitat use

To determine how varanids move through and use reference and restoration vegetation, we GPS mapped all signs of varanid habitat use (tracks, diggings, and burrows) along To determine how varanids move through and use reference and restoration vegetation, we GPS mapped all signs of varanid habitat use (tracks, diggings, and burrows) along transects. Varanids create visible and distinctive tracks (Blamires, 2000), and in the absence of similar-sized burrowing animals in the study region (Bamford, 2006), burrows and diggings were easily identifiable. We marked burrows and diggings as fresh activity if there were signs of recently disturbed soil or visible tracks around each use, and tracks were recorded as fresh if footprints were clearly visible. To gauge the size of varanids utilising reference or restoration areas, we recorded the height and width of burrows, and measurements of tracks, including total track length, tail width (TW; width at the thickest section of the tail mark), stride length (SL; distance between the base of the pad of the forelimb and tip of the middle claw of the hindlimb; Fig. 5.2a), and foot length (FL; base of the pad to the tip of the longest claw; Fig. 5.2b, c). Where possible, we measured the FL of the hindlimb and SL of five imprints along each track to obtain an average measurement for analyses. We controlled for independence in samples by including tracks only if they were separated by at least 100m from any other track, or if they were recorded on different dates and therefore verified as fresh usage. While assessments of habitat use by animals using tracks are infrequent, some studies of wide-ranging fauna have suggested 100m as a suitable distance for independence of samples to reduce the risk of spatial autocorrelation (e.g., Bowman and Robitaille, 1997; Proulx and O'Doherty, 2006; Proulx *et al.*, 2006).

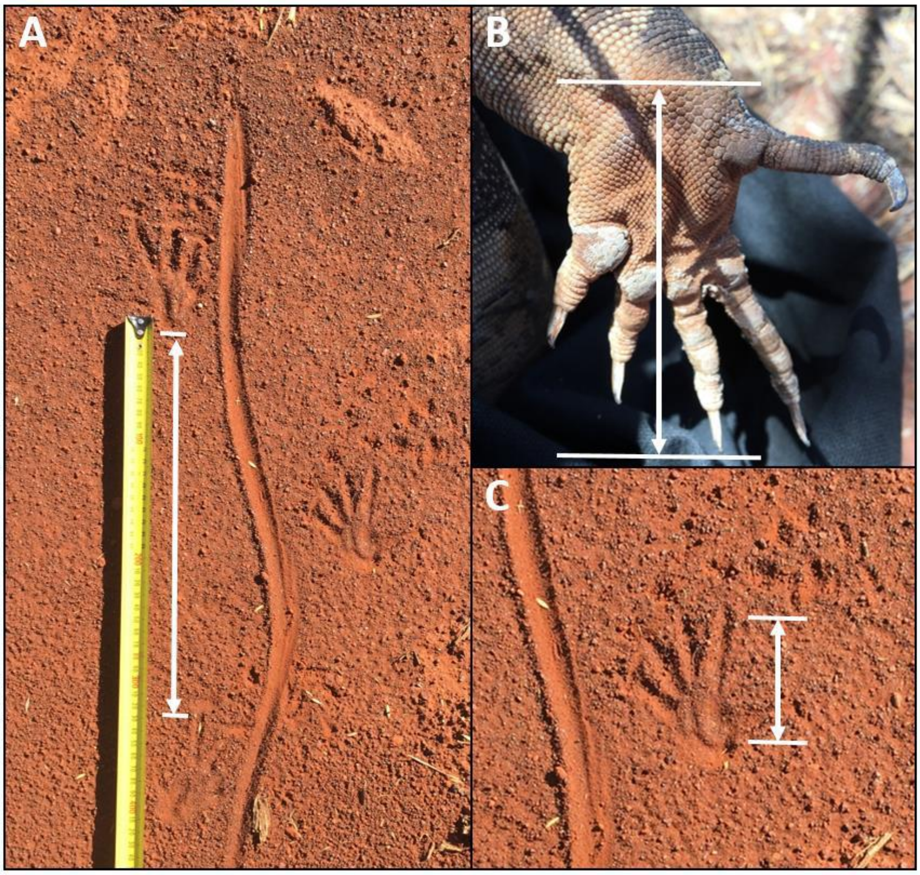

Fig. 5.2: Measurements of varanid tracks: A) stride length from the base of the pad of the forelimb, to the middle claw tip of the hindlimb, B) foot length of the hindlimb as shown on a *Varanus giganteus*, from the base of the pad to the claw tip of the middle digit, and C) the resultant foot imprint and length of measurement taken.

5.3.5 Statistical analyses

5.3.5.1 Thermal environment

To determine whether there were any differences in the thermal environment between reference and restoration vegetation, we used paired *t*-tests with temperature as the dependent variable, and vegetation type (reference or restoration) as the independent variable. As varanids are active diurnally (King and Green, 1993), we assessed whether reference and restoration sites presented thermally heterogenous environments by calculating the coefficient of variation for average day time temperatures between 0700 and 1800. As varanid activity is typically highest at temperatures around 35°C, to determine the frequency at which recorded temperatures exceeded optimal activity temperature in reference and restoration vegetation, we calculated the total and average number of days, and minutes each day, where temperatures (recorded by dataloggers within each site) exceeded 35°C.

5.3.5.2 Habitat usage

We used log-linear models to analyse differences in the total and type of habitat use between reference and restoration vegetation, with vegetation type (reference or restoration), and site (8km or 12km) as the independent variables, and the total number of each habitat usage (numbers of tracks, diggings, or burrows) as the dependent variable. We repeated analyses with only fresh habitat usage included, and then with only old usage, to assess whether each vegetation type was more likely to contain fresh or old habitat usage. We used separate two-way generalised linear models with a Gaussian distribution to assess whether track measurements (total length, TW, FL, or SL), or the size of burrows differed between reference and restoration vegetation, with vegetation type and site as the independent variables, and track length, TW, FL, SL, or burrow size as the dependent variables. As tracks were marked as ‘old usage’ if there were no visible foot imprints, we included only fresh tracks in measurement analyses.

5.3.5.3 Usage measurements

To assess whether varanids show selectivity in movement, we calculated standard proportions of travel (SP_t) in both reference and restoration vegetation for each identifiably unique trackway, using:

$$SP_t = \frac{Total\ distance\ traveled}{Shortest\ distance\ from\ point\ A\ to\ B}$$

where total distance travelled refers to the total recorded track length, and shortest distance is the straight-line distance from point A (track start) to point B (track end). As with assessments for burrow and track measurements, we analysed differences in proportions of travel between reference and restoration vegetation using two-way generalised linear models with a Gaussian distribution, with vegetation type and site as the independent variables and standard proportion of travel as the dependent variable. All analyses were conducted using R Studio (RStudio, Inc, Boston, United States, 2019).

5.4 Results

5.4.1 Temperature

Recorded temperatures ranged between 5.5 – 42.8°C in reference vegetation, and 5.75 – 46.8°C in restoration vegetation but reference and restoration sites did not differ significantly in either the average number of days (t = -5.80, d.f. = 1, P = 0.109), or average number of daily minutes (t = -10.98, d.f. = 1, P = 0.058) exceeding 35°C. Although average temperatures did not differ statistically (Fig. 5.3a), reference vegetation had a higher level of variability in recorded temperatures than restoration vegetation, where the thermal environment was relatively homogenous (Fig. 5.3b).

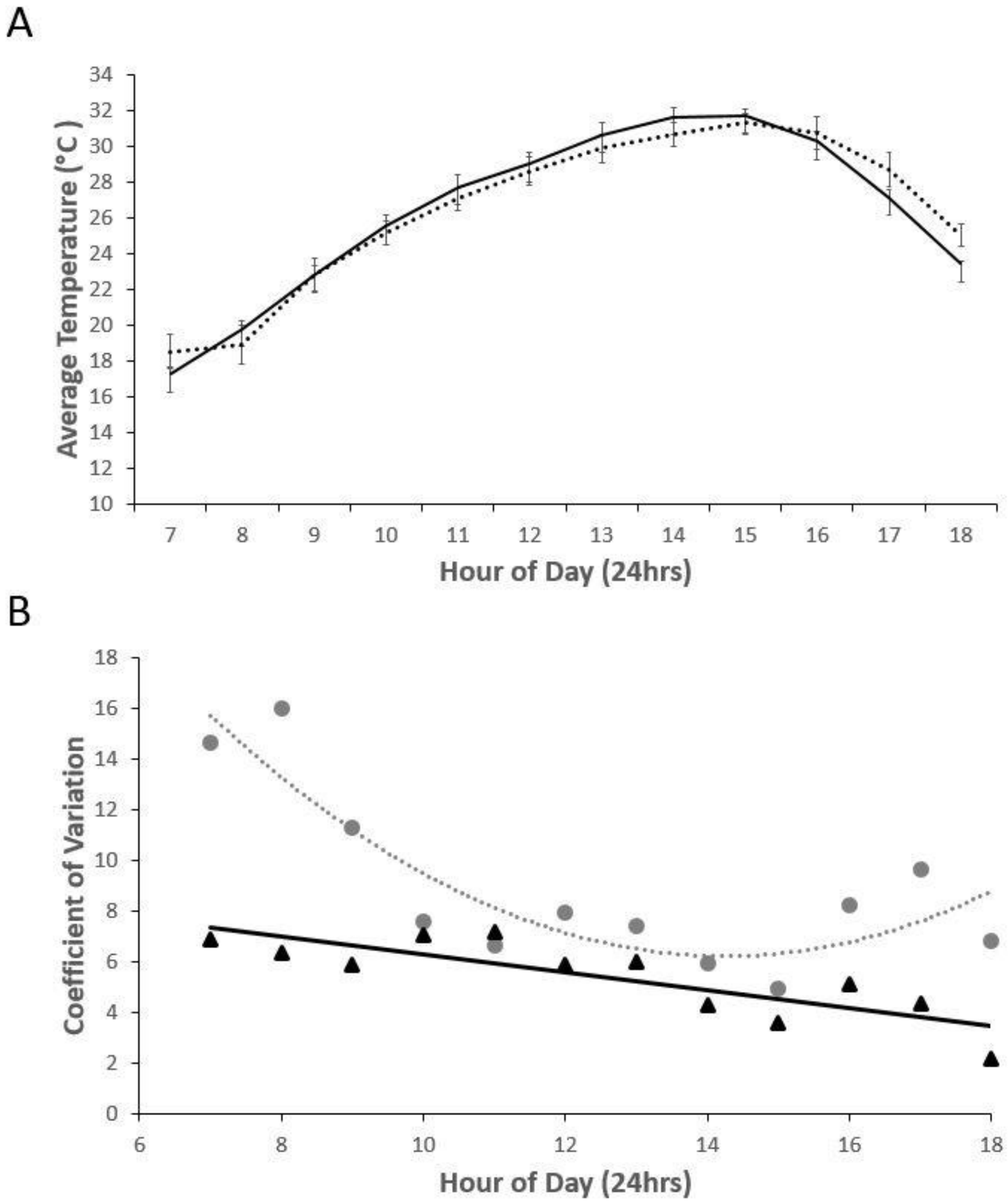

Fig. 5.3: Thermal environment of reference and restoration vegetation during daylight hours (0700 – 1800): A) average hourly temperatures in reference (···) and restoration (-) vegetation, and B) coefficient of variability in hourly temperatures in reference (•) and restoration (▲) sites. Trendlines in reference (···) and restoration (-) vegetation do not relate to a statistical function but emphasise differences in thermal patterns between reference and restoration areas.

5.4.2 Usage of reference and restoration sites

We recorded a total of 138 signs of habitat usage across all sites, with 80% (*n* = 110) of all usage recorded within the reference habitat (Fig. 5.4). Tracks were the most frequently recorded evidence of varanid presence within habitats (*n* = 60, 44%), with diggings and burrows recorded at equal frequencies (*n* = 39, 28%). Total habitat usage (i.e., the combined total of all recorded burrows, diggings, and tracks within each vegetation type) was significantly higher in the reference vegetation, both including ($\chi2 = 48.72$, d.f. = 2, $P < 0.001$) and excluding points of old habitat usage ($\chi2 = 17.09$, d.f. = 2, $P < 0.001$). Reference vegetation contained a high proportion of old habitat usage (*n* = 46/110, 42% of all recorded usage); however, we rarely recorded old habitat usage within restoration vegetation (*n* = 3/28, 10.7% of all recorded usage). Restoration vegetation rarely contained signs of burrowing or foraging activity, with tracks recorded significantly more frequently than both diggings and burrows in these areas ($\chi2 = 6.50$, d.f. = 2, $P = 0.038$). Reference vegetation supported movement, foraging activity, and burrow use at similar frequencies, with no significant differences in the type of habitat usage recorded in reference areas ($\chi2 = 2.96$, d.f. = 2, $P = 0.227$). We did not note any significant interaction effect between site (8 or 12 km from active mining) and total habitat usage in reference and restoration vegetation ($\chi2 = 0.73$, d.f. = 1, $P = 0.392$).

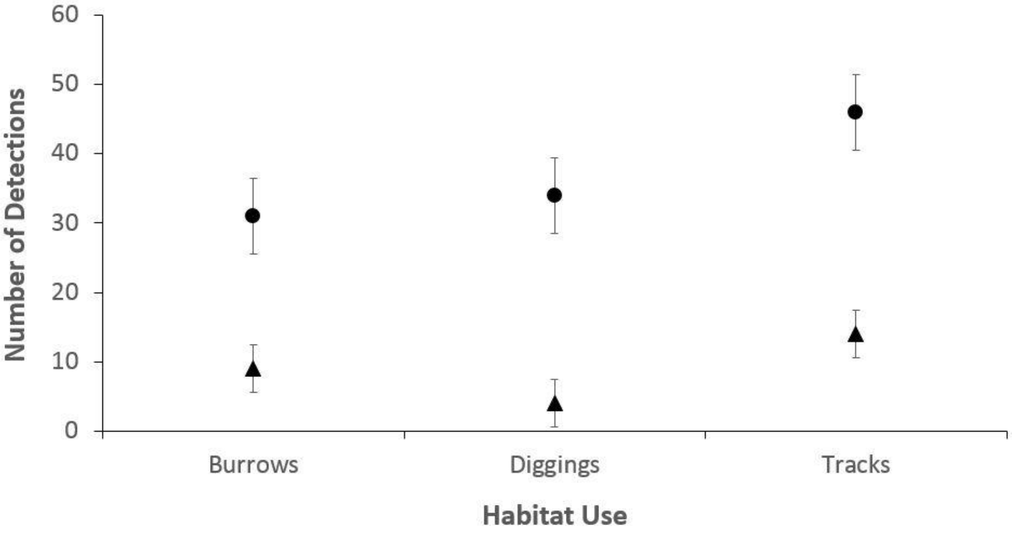

Fig. 5.4: Total recorded varanid habitat usage ± standard error, including burrows, diggings, and tracks, in reference (•) and restoration (▲) sites.

5.4.3 Burrow and track measurements

Burrows within the restoration vegetation were significantly larger 234.0 ± SE 22.4 mm height, 321.20 ± 24.70 mm width) than those within the reference vegetation (176.0 mm ± SE 62.10 height, 251.0 ± 10.0 mm; $F_{(1, 35)} = 6.06$, $P = 0.019$; Fig. 5.5). However, we did not detect any significant difference between each area for total track length ($F_{(1, 46)} = 0.12$, $P = 0.733$), TW ($F_{(1, 46)} = 0.34$, $P = 0.546$), FL ($F_{(1, 46)} = 0.79$, $P = 0.376$), or SL of tracks ($F_{(1, 46)} = 0.29$, $P = 0.595$). Fifty-five percent of all tracks crossed through any given area without deviation, and we did not detect any significant difference between reference and restoration vegetation in the proportion of travel ($F_{(1, 56)} = 2.41$, $P = 0.126$). While there were no significant differences in proportion of travel of varanids between reference and restoration sites, tracks in the reference habitat displayed the largest range of travel proportion, ranging between 1 (straight line, no deviation) and 4.27 (proportion travelled ~4 times greater than the straight-line distance between the track start and end), in comparison to restoration areas, which only ranged between 1 and 1.26.

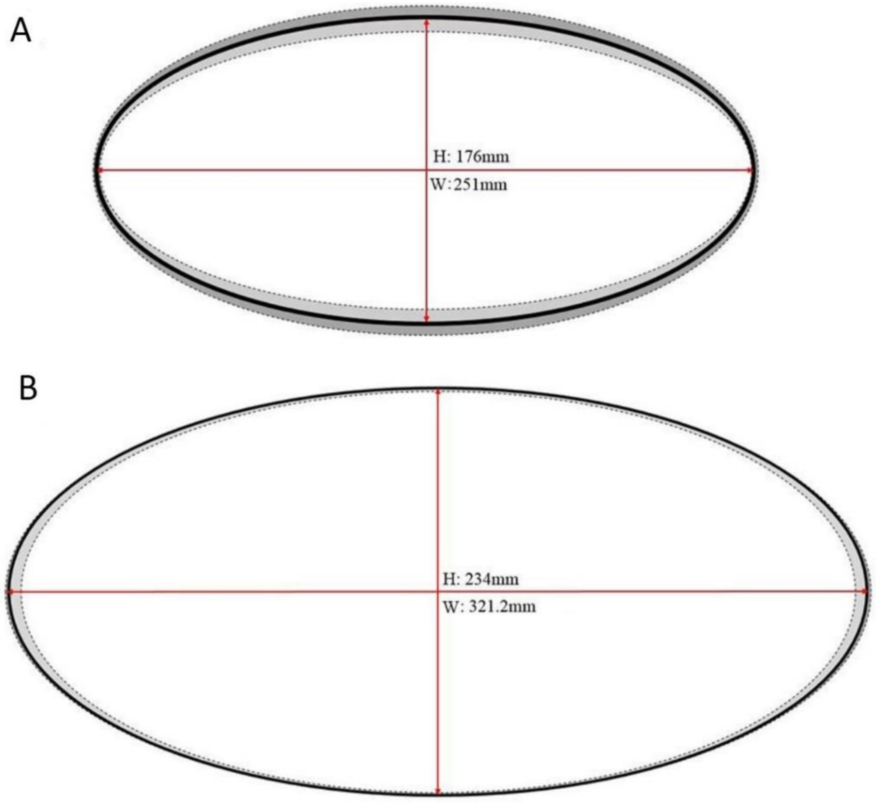

Fig. 5.5: Variability in burrow width and height (mm) between a) reference vegetation, and b) restoration vegetation, drawn to scale. The middle oval in each figure represents average burrow size, and dashed lines and shaded areas show the average plus/minus one standard error.

5.5 Discussion

Ambient temperature ranges did not differ significantly between reference and restoration sites; however, early stage restoration lacked the thermal variability present within reference vegetation. Restoration vegetation in the study area is being used by varanids; however, usage appears to be infrequent and largely opportunistic. The metabolic costs of thermoregulating within spatially homogenous landscapes can be significantly higher than those within heterogenous landscapes, even in cases where recorded temperatures do not differ between landscapes (Sears *et al.,* 2011; Tuff *et al.*, 2016). We recorded tracks significantly more frequently than both burrows and diggings within restoration vegetation, in comparison to the reference vegetation where tracks, diggings, and burrows were recorded in similar proportions. The lack of diggings and burrowing activity in restoration vegetation may indicate that these areas lack key resources, such as food, thermal refuges, and a diversity of microclimates, and may have increased metabolic costs associated with their use. While burrows were infrequently recorded in restoration vegetation, their use appeared to be restricted to larger-bodied varanids. Early stage habitat restoration may be more restrictive to smaller-bodied varanids, which have lower thermal tolerances, lower thermal inertia, and increased reliance on an availability of thermal refuges and a diversity of microhabitats (Huey and Bennett, 1990). However, it is also possible that larger bodied varanids colonise restoration first and restrict use by smaller individuals through competitive exclusion, although evidence in the literature to support this is sparse.

5.5.1 Habitat usage

We recorded signs of varanid activity significantly less frequently within restoration habitat than within the reference habitat, with restoration areas containing just 20% of all recorded habitat usage. Sites undergoing restoration following discontinuation of mining activities can require long periods of time for vegetation to become established, or for vegetation to approach a state comparable to pre-disturbance structure and floristics (Grant and Loneragan, 1999; Tuff *et al.*, 2016). Furthermore, restoration, particularly early stage restoration, is typically more homogeneous than reference habitats, as the reinstatement of vegetation structure can be a slow process (Pywell *et al.*, 2002; Baer *et al.*, 2004).Survival of reptile populations is often dependent on spatially heterogenous habitats with an abundance of microclimates to support foraging and thermoregulatory behaviours (Hertz *et al.*, 1993; Basson *et al.*, 2017). Reptile populations are negatively impacted by a loss of native vegetation and decreasing habitat structure and complexity (Smith *et al.*, 1996; Cunningham *et al.*, 2007; Brown *et al.*, 2008). Therefore, spatially and structurally homogenous landscapes often present increased metabolic costs through a lack of suitable thermal refuges and diverse microhabitats (Attum and Eason, 2006).

Reptiles experience trade-offs between time spent foraging, and time spent engaged in thermoregulatory behaviour (Tuff *et al.*, 2016). This trade off can be particularly high in hot, open landscapes that impose increased metabolic costs and predation risk (Tuff *et al.*, 2016), and we rarely recorded burrows or diggings in restored vegetation. The lower variability in daily temperatures in restored areas indicate that these sites present increased homogeneity in the thermal landscape and may present increased thermal costs (Sears and Angilletta, 2015; Cross *et al.*, 2020c). Foraging densities can be indicative of both prey abundance and habitat quality, with habitat patches of higher quality and an increased abundance of resources tending to support increased foraging activity (Lindell, 2008; Wellenreuther and Connell, 2002; Kilgo, 2005). Mapping diggings may not fully encapsulate foraging activity by varanids in reference and restored habitats as larger-bodied varanids can hunt and capture vertebrate prey items without digging. However, many varanid species are primarily insectivorous and often dig for food resources (Losos and Greene, 1988). Varanids of a range of body sizes in the Mid West region of Western Australia primarily prey upon invertebrates and small reptilian species (Cross *et al.*, 2020b), suggesting diggings accurately represented foraging activity. Invertebrate richness has previously been reported to be positively correlated with increasing vegetation structure and diversity (Muren *et al.*, 2003; Robinson *et al.*, 2018). Increased metabolic costs and a reduction in vegetation structure in restoration areas may present less favourable habitat for both predator and prey species, limiting foraging efficiency by varanids in these areas. High metabolic costs associated with homogenous landscapes can be particularly restrictive for smaller reptile species, which rapidly reach a temperature equilibrium with the surrounding environment (Huey and Bennett, 1990). As with signs of foraging, we rarely recorded burrows in restoration vegetation; however, when present, burrows in these areas appeared to be restricted to larger bodied varanids. Larger bodied reptiles have greater thermal inertia, requiring longer time-periods to reach maximum thermal levels (Cowles and Bogert, 1944), and are typically able to withstand greater temperature fluctuations than smaller individuals (Spotila *et al.*, 1973; Stevenson, 1985; Huey and Bennett, 1990).

5.5.2 Movement ecology

Tracks were the most frequently recorded sign of varanid presence within restoration vegetation; however, we recorded tracks in these areas significantly less frequently than within the reference vegetation. The structural complexity of landscapes, and the perceived costs associated with their use, have substantial impacts on the movement ecology of animals (Morales and Ellner, 2002; Jeanson *et al.*, 2003; Fahrig, 2007). Boundaries of habitat patches can impose hard constraints to movement and dispersal, with animals unlikely to leave habitat patches if the surrounding habitat is of lower quality (Fahrig, 2007). The decreased availability of refuges and established vegetation cover within restoration areas may account for the reduction in movement of varanids within these areas. Animals minimise time spent in high risk environments, tending to cross these areas rapidly and infrequently, in contrast with higher quality habitats that facilitate slower and non-uniform movement and can be crossed with less selectivity (Fahrig, 2007). While we did not detect any significant differences in the proportion of travel between reference and restoration vegetation, tracks within restoration areas rarely deviated from straight-line movement, whereas tracks in the reference bushland had greater variability in the proportion of travel.

While use of longer-term signs of habitat usage (e.g., burrows), may be restricted to larger bodied varanids, we did not find any significant difference in the size of varanids traversing through restoration and reference areas. Early stage restoration appears to support infrequent, opportunistic use by individuals of a range of body sizes, with use by smaller-bodied varanids largely confined to simply traversing through the restoration habitat. This disparity in use may result in restored landscapes serving as an ecological trap, with smaller-bodied varanids capable of moving through these landscapes but not persisting within restored areas. Ecological traps may affect the long-term viability and persistence of populations within habitats (Battin, 2004). However, our study represents a snapshot of usage in early stage habitat restoration and it is likely that habitat usage by both predator and prey species will increase as vegetation structure becomes established and heterogeneity increases, creating an availability of suitable microhabitats.

5.5.3 Study limitations

There are some limitations to assessments of behaviour and movement ecology of animals through indirect measures, such as monitoring habitat usage. Although providing an effective method of assessing habitat usage by populations, determining usage by individuals can be challenging, and habitat usage is likely to vary among individuals of different ages and sexes (Garshelis, 2000). While some studies suggest 100m as an appropriate distance between tracks for sample independence (e.g., Bowman and Robitaille, 1997; Proulx and O'Doherty, 2006; Proulx *et al.*, 2006), varanids can move over large distances (e.g., Green *et al.*, 1986; Cross *et al.*, 2020c) and difficulties in identifying between usage of individuals risks introducing spatial autocorrelation. Furthermore, given the varying effects of environmental factors, such as fluctuating temperatures, on small and large-bodied reptiles (Spotila *et al.*, 1973; Stevenson, 1985; Huey and Bennett, 1990), it is likely that *Varanus* species, or juveniles and adults of the same species, are impacted by habitat degradation and restoration to varying extents. Conclusions drawn at only a population level may therefore not fully represent the ecological impacts of habitat degradation and subsequent restoration on their behaviour and movement ecology. This method may also present a bias towards animals occupying terrestrial niches, with habitat usage by primarily arboreal species less likely to be recorded on the ground. Furthermore, substrate and vegetation density can impact the detectability of habitat usage (Garshelis, 2000). Soil compaction following the use of heavy machinery is common in areas undergoing restoration (Bradshaw, 1997) and may have reduced the probability of detecting use by varanids in these areas. Lastly, our results may have been affected by the differing proximity of our sites to the active mine pit. However, as we did not detect any interaction effects between distance of sites from active mining (8 or 12km) and habitat usage of reference and restoration vegetation, we concluded that proximity to the active mine pit was unlikely to have influenced our results.

5.5.4 Conclusions

Restoration of discontinued mine sites within the study area appeared to be supporting a level of usage by varanids. However, usage appeared to be largely movement through restored areas, or where burrows were present, usage was restricted primarily to larger-bodied individuals. Our data suggest that restoration areas may contain a paucity of some fundamental resources, such as food resources, thermal refuges, and a diversity of microclimates. Decreased spatial heterogeneity in restoration likely presents unfavourable thermal conditions, reducing the abundance of both varanids, and the prey they are reliant on. Returning fauna refuges, for example hollow logs and debris piles, may aid in facilitating colonisation and long-term use of restoration sites by varanids, and increase their resilience to habitat disturbance, particularly during the initial stages of vegetation establishment (Koch, 2007; Robinson *et al.*, 2013; Christie *et al.*, 2013; Connell and Keppel, 2016). Most studies of wildlife responses to mine site restoration only consider presence or abundance of animals (Cross *et al.*, 2019, 2020a). Had this study considered only the presence of varanids as being indicative of habitat use, we might have concluded that restoration and reference sites were utilised similarly. However, by assessing the features associated with movement ecology and burrow use, we show that restoration sites may lack some key resources required to sustain reptile populations, particularly small-bodied varanids. Further research to identify the key resources promoting and aiding the return of fauna groups from a variety of taxa and trophic levels over multiple temporal scales is key to returning functional and diverse fauna populations to habitats undergoing restoration. Understanding how animals respond to habitat change and restoration is critical to their conservation in the face of ever-increasing rates of habitat degradation and loss.

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Chapter 6. The Time Local Convex Hull (T-LoCoH) method as a tool for assessing responses of fauna to habitat restoration: a case study using the perentie (*Varanus giganteus*: Reptilia: Varanidae)

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'Petra' the perentie (*Varanus giganteus*) and her mate

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6.1 Abstract

Understanding the behavioural responses of animals to habitat change is vital to their conservation in landscapes undergoing restoration. Studies of animal responses to habitat restoration typically assess species presence/absence; however, such studies may be restricted in their ability to show whether restoration is facilitating the return of self-sustaining and functional fauna populations. We present a case study using VHF/GPS tracking of a young adult perentie (*Varanus giganteus*), to demonstrate the range of applications of the Time Local Convex Hull method of home range construction in analysing the behavioural responses of fauna to habitat change and restoration. Presence/absence studies provide single point locations of an animal, and the Minimum Convex Polygon method provides an invariant estimate of habitat use across the whole home range. However, the Time Local Convex Hull method provides a useful method for assessing movement and behavioural responses of fauna to habitat change and restoration, and the specific habitat requirements for the long-term support of populations. The breadth and multidimensionality of data generated indicates strongly that understanding the complex interactions between animals and their environment is fundamental to their conservation in the face of ever-increasing rates of human induced habitat change and degradation.

6.2 Introduction

Habitat loss and degradation are leading drivers of species declines globally (Dobson *et al.*, 1997). As we enter the Earth's sixth mass extinction event (largely anthropogenically driven; Kingsford *et al.*, 2009), conservation efforts to aid in the mitigation of human-induced landscape change are becoming increasingly vital. Reptiles are experiencing global declines, largely driven by a loss and degradation of suitable habitat, through human influences such as industrialisation, urbanisation, mining, and agriculture (Böhm *et al.*, 2013; Webb *et al.*, 2014). Ectothermic animals, such as reptiles, rely on their habitat to regulate body temperature, and are thought to be at especially high risk of declines through changes in habitats and thermal environments (Ihlow *et al.*, 2012). Reptiles are often overlooked in assessments of habitat change and restoration, and among existing studies there is a focus towards assessments for species diversity, presence, or abundance (Lindell, 2008; Cross *et al.*, 2019). These studies have several limitations, namely in their inability to account for the varying life history strategies of animals that alter their behavioural responses to habitat change (Lindell, 2008; Chiarucci *et al.*, 2011). Understanding the processes driving the responses of reptiles to habitat alteration and restoration, in addition to assessments of their presence or absence from these systems, is vital to their conservation in the face of ever-increasing rates of habitat degradation and loss (Heard *et al.*, 2004; Seebacher and Franklin, 2012).

Behavioural ecology and habitat selection are key to understanding the fundamental resource requirements for long term persistence of fauna within ecosystems (Ims, 1995; Mauritzen *et al.*, 2003). Understanding the factors behind how fauna adapt and respond to varying habitat and climatic conditions, such as increasing temperatures and habitat fragmentation, is essential in forming predictions of how they may cope with habitat change and degradation into the future (Tuff *et al.*, 2016). This has critical applications for reptiles as climate change alone is predicted to drive extinction rates as high as 20% by 2080 (Sinervo *et al.*, 2010) and, in addition to a shifting climate, changing vegetation cover can markedly alter the thermal landscape and the availability of refuges for thermoregulation (Attum and Eason, 2006). Thermoregulatory ability of reptiles is strongly influenced by microclimates within their habitat, and their survival is highly dependent on an availability of spatially heterogeneous habitats with high resource abundance (Hertz *et al.*, 1993; Basson *et al.*, 2017). Predation pressures and metabolic costs can strongly impact the fitness and foraging efficiency of animals (Lima and Dill, 1990). Lima and Bednekoff (1999) propose that animals will preferentially forage within habitats perceived as ‘low-risk’ environments and exhibit antipredator behaviour and reduced foraging activity in higher-risk areas. Therefore, particularly for ectothermic species, in hot, open landscapes (higher risk environments), the trade-offs between time spent in thermoregulatory behaviour and time spent foraging can be high (Tuff *et al.*, 2016).

Biotelemetry using VHF (Very High Frequency) and GPS (Global Positioning System) tracking provides an effective method of recording data for assessments of animal behaviour and movement patterns over multiple spatial scales (Obbard *et al.*, 1998). Biotelemetry can provide detailed information on an animal’s ecology, including its home range, use of the habitat, responses to the thermal environment, and activity, with minimal stress or intrusion to the animal (Cooke, 2008). Previous methods of analysing data from bio-telemetry approaches have largely been restricted to the construction of home ranges using the minimum convex polygon method (MCP; Worton, 1987). While providing important ecological data the MCP method does not account for the influence of environmental factors on home range and movement, such as temperature and vegetation structure, and does not show differential use of the home range or whether the entirety of the home range is in use (Worton, 1987). Other attempts of modelling home ranges have sought to define core or larger space-use areas (e.g., Kernel Density Estimates (KDE); Laver and Kelly, 2008), however the assumptions concerning the distribution of the point data are often violated, thereby over-estimating home ranges (Getz *et al.*, 2007; Laver and Kelly, 2008).

Methods for analysing home-range data are constantly evolving, and recent years have seen the implementation of increasingly complex statistical algorithms for assessing home range and habitat selection by animals (Lele *et al.*, 2013), including methods such as the Brownian Bridge Movement Model (BBMM), and the Time Local Convex Hull (T-LoCoH) method of home range construction, both of which have been used successfully to elucidate movements of animals within their home range (Byrne *et al.*, 2014; Lyons *et al.*, 2013; Tarszisz *et al.*, 2018). T-LoCoH and the BBMM method not only include points of presence of an animal, but also include spatial and temporal information, allowing for a complex understanding of habitat use and behavioural ecology (Byrne *et al.*, 2014). However, BBMM estimates movement paths between recorded locations of an animal and the probable location of an animal at any given time and may present an inaccurate reflection of behaviour and movement (Kranstauber *et al.*, 2012; Ofstad *et al.*, 2019). In comparison, T-LoCoH uses observed movement and can measure behaviour of animals in their habitat and uses a kernel modelling approach to identify areas that an animal visits repeatedly, which may represent core usage areas of the habitat, of which may be more profitable than others (i.e., contain an abundance of resources or refuges for thermoregulation; Lyons *et al.*, 2013; Tuff *et al.*, 2016) and are therefore visited frequently and for longer durations than non-core areas (Benhamou and Riotte-Lambert, 2012). Data extracted with either approach may be overlaid with environmental factors to show complex interactions of animals with their environment, providing a detailed understanding of behavioural and ecological responses to habitat change. Compared to assessments for species presence or absence, or home range analyses such as the MCP and KDE methods, T-LoCoH can aid in understanding the behavioural responses of fauna, including ectothermic animals, not only to habitat change and fragmentation, but to differing habitats and thermal environments. With globally increasing rates of anthropogenically-driven habitat change, understanding ecological, behavioural, and thermoregulatory responses are key to understanding how reptiles may respond to future changes, and to aiding future conservation efforts (Heard *et al.*, 2004).

The perentie (*Varanus giganteus*, Reptilia: Varanidae) is Australia's largest lizard species, growing up to two and a half meters in length and weighing around 17kg at maturity (Pianka, 1994; Pianka *et al.*, 2004). Occupying terrestrial niches, perenties occur throughout much of arid Australia, from the Mid West region of Western Australia to far western Queensland (Pianka *et al.*, 2004; Cogger, 2014). An apex predator, the perentie is active almost exclusively diurnally (during daylight hours) with activity typically highest during late spring and summer at temperatures averaging around 36°C (King *et al.*, 1989; King and Green, 1993). Perenties tend to have large home ranges and forage over large areas (King *et al.*, 1989), with distances of up to 900m between consecutive locations of tracked perenties previously reported (Green *et al.*, 1986). Importantly, due to their high mobility, they present an ideal study species to monitor thermal influences and habitat change over relatively large spatial scales.

Here, we report on the range of applications of the T-LoCoH method of home range construction for understanding the responses of fauna to habitat change and restoration using a young adult female *V. giganteus* as a case study. T-LoCoH has been used successfully in previous studies of animal movement, however these have been limited to either simulated data or mammalian studies, and have not sought to determine differential habitat use in altered or restored landscapes, but primarily spatial utilisation of home ranges (e.g., Lyons *et al.*, 2013; Stark *et al.*, 2017), with one ecophysiological study of movement and seed dispersal (Tarszisz *et al.*, 2018). Here we discuss how T-LoCoH can be applied to ectothermic animals and studies of conservation, habitat alteration, and ecophysiology. We aim to assess how T-LoCoH may be applied in assessments of habitat use, movement, and activity of a perentie within reference (bushland not directly impacted by mining activities) and anthropogenically-altered habitats. We hypothesise that the T-LoCoH method will provide an effective measure for understanding the mechanisms driving behavioural and ecological responses of fauna to altered habitats, which should facilitate their future conservation in the face of increasing rates of habitat degradation and loss.

6.3 Methods

6.3.1 Study Site and Species

We conducted our study in the Mid West region of Western Australia at an area undergoing restoration within a major magnetite mining operation, approximately 430 km northeast of Perth (29°08'50.3"S 116°49'07.5"E, Fig. 6.1A, B). Restoration of a waste rock dump (an area of ~0.5 km^2), situated 8 km north of the current mining activities within the study area, commenced in May 2014 (earthworks and seeding), with the completion of all seeding by July 2017. The study site (Fig. 6.2) experiences an arid climate, with temperatures in late spring typically ranging between 27°C and 37°C (Australian Bureau of Meteorology, http://www.bom.gov.au/climate/data/). Vegetation within the study area largely comprises *Acacia* shrublands and open *Eucalyptus* woodlands (Bamford, 2006), with the restoration habitat comprising a similar species composition. However, vegetation in the restoration area is at varying stages of establishment and lacks the spatial heterogeneity of the reference habitat (Fig. 6.3). We sought to catch adult perenties between October and November 2018, using cage trapping, noosing, and other methods of hand capture (Flesch *et al.*, 2009; García-Muñoz and Sillero, 2010). We imposed a constraint that the ratio of tracker to body weight was < 4% of the varanid's total body weight to minimise adverse effects to the animal's fitness or a reduction in movement (Flesch *et al.*, 2009). Shy and elusive, perenties can be difficult to capture (Pianka, 1994; Moro and MacAulay, 2014), and we only succeeded in capturing a single *V. giganteus* individual large enough to outfit for telemetry; a young adult female (head width: 35 mm, head length: 85 mm, snout-vent length: 495 mm, total length: 1215 mm, body weight: 2.1 kg). We tracked its movements for a period of 20 days from the 8 – 28 of November 2018, ending when it shed the transmitter.

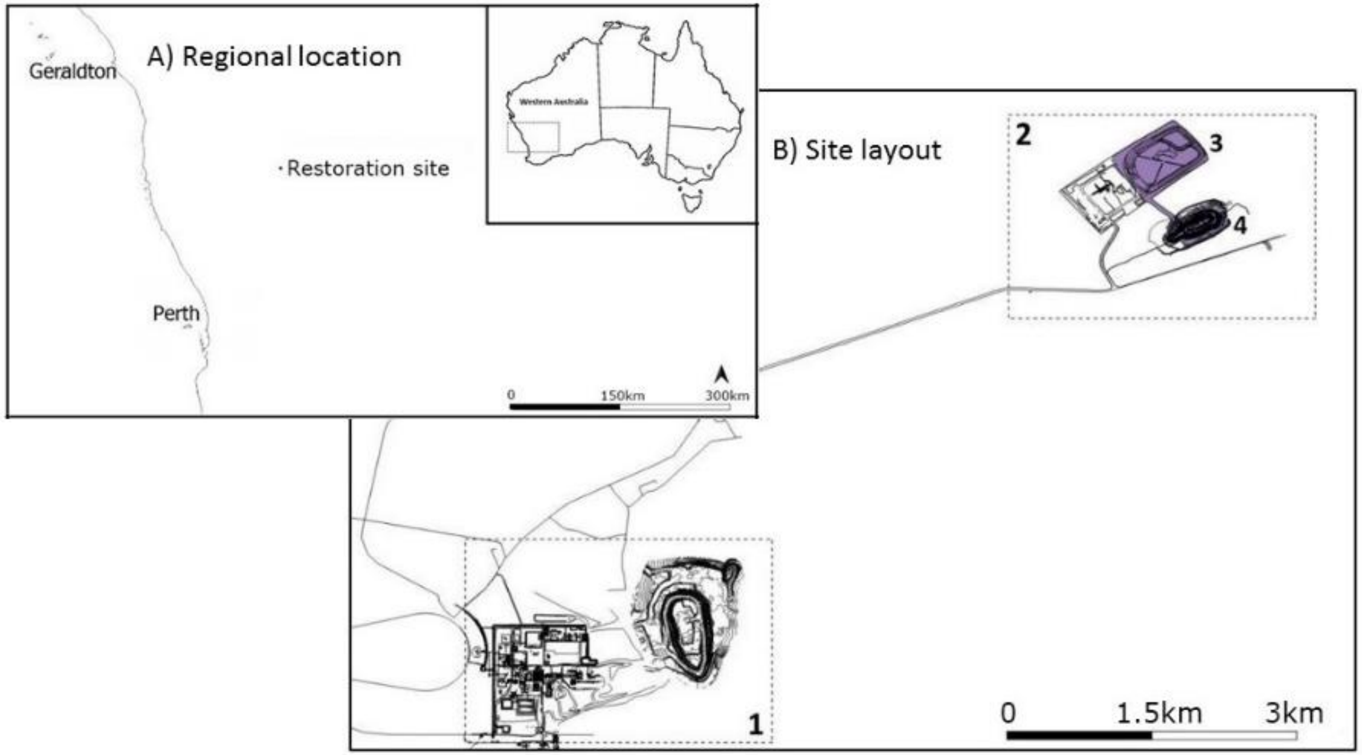

Fig. 6.1: A) The location of the study site in the Mid West region of Western Australia, and B) the site layout, comprising the location of active mining activities (1), and the restoration project (2), which is characterised by two areas of anthropogenic disturbance, the restoration of a waste rock dump (3) and a disused mine pit void (4), surrounded by otherwise unmined habitat. The study site is roughly 8km distant from the current active mining operations and our findings are unlikely to have been influenced directly by that activity.

Fig. 6.2: An aerial view of the study site, characterised by (1) the restoration waste rock dump, and (2) the disused mine pit, surrounded by reference habitat.

Fig. 6.3: Typical vegetation structure within restoration and reference habitats. Restoration vegetation is at varying stages of establishment and cover is reduced in comparison to the reference vegetation.

6.3.2 Transmitter Attachment and Tracking

The perentie was fitted with a W510 Wildlink Wireless GPS/VHF tracking unit (50 x 40 x 10mm, 65g; Advanced Telemetry Systems, Australia). The tracker was affixed to its dorsal surface with non-toxic, quick drying superglue (Fig. 6.4), and the edges of the tracker were covered with cloth tape to reduce the likelihood of the tracker catching on debris or being dislodged (Price-Rees and Shine, 2011). As perenties are not considered to be nocturnally active (King and Green, 1993), the GPS logger was programmed to take 58 fixes per day, with quarter hourly fixes during daylight hours (between 0600hrs and 1900hrs), and one fix every two hours overnight (between 1900hrs and 0600hrs). As we were tracking a single animal, the GPS was programmed to record the maximum number of fixes during periods of high activity, to attain a thorough and consistent dataset of the perentie's movements throughout its home range. Transmitters recorded date and time of fix, activity of the perentie (the number of minutes the perentie was active for during the 15 minutes prior to each fix, recorded as a percentage), and ambient temperature (temperature of the tracking device at the time of the GPS fix recording). Activity was recorded internally within the GPS unit via a switch activated upon movement of the animal, which continually reset every 15 minutes when a new fix was recorded.

Fig. 6.4: The tracking unit affixed to the back of a perentie (*Varanus giganteus*). Left: the unit attached on the middle of the dorsal surface, set just behind the line of the forelimbs to minimise the likelihood of the varanid dislodging the tracker, and right: the perentie observed in situ basking next to a fauna refuge in the restoration area, with the telemetry package and cloth tape covering (Photographed by S. Cross).

Following tracker attachment, the perentie was released at the point of capture and VHF tracked with visual sightings for the remainder of the day to ensure its welfare. To maximise the likelihood of recapturing the perentie, attaining a download of fixes, or finding the tracker if it was shed, we conducted twice daily VHF tracking with an early morning track prior to 0800hrs to observe the lizard as it emerged from its burrow, and a late afternoon track to identify its nocturnal refuge. GPS fixes were remotely downloaded each morning upon location of the animal. Tracking ceased upon shedding of the transmitter (9.40am on 28 November 2018), at which time we observed the perentie engaged in mating behaviour and were able to confirm its sex as female. We concluded the tracker had no adverse effects on the perentie's health.

6.3.3. Home Range and Movement Ecology

We analysed the data on the movement patterns of the perentie using the T-LoCoH method to determine its home range, and influence of vegetation quality (reference or restoration) and temperature on movement within its habitat (Lyons *et al.*, 2013). Analyses included all GPS points for which a successful fix could be attained via four or more satellites (Stark *et al.*, 2017) and ultimately comprised 865 successful fixes (79% of all data points) over a 20-day duration. Excluded data comprised 'zero fixes' where the perentie was underground in a burrow and GPS fix could not be recorded. As we were assessing daily movement patterns of the perentie, for the purposes of analysing frequency and duration of visits, we applied an inter-visit gap (IVG) of 12 hours following Lyons (2014) and, as such, points were only considered separate visits if a period of 12 hours or greater had passed from the previous visit.

T-LoCoH offers two methods in kernel modelling, the '*k'* method, which standardises the number of nearest neighbours, and the '*a'* method, which accounts for single excursions by an animal within its habitat and decreases sampling bias by 'reducing the number of nearest neighbours used in areas where points are thin and scattered' (Lyons, 2014). As our data comprised both heavily and sparsely utilised regions of the home range, we applied the *a* (adaptive) method in the construction of hulls, following Lyons (2014). The *a* method decreases the likelihood of over-estimating use of particular areas of the home range (Lyons *et al.*, 2013; Lyons, 2014). We computed hulls for a variety of *a* values, and selected an *a*- value of 8000 as the model-of-best-fit based upon isopleth models and edge: ratio curves (Lyons, 2014). We used chi square tests to compare observed frequency (NSV; Number of Separate Visits) and duration of visits (MNLV; Mean Number of Locations in the Hull per Visit) within each area of the habitat with expected values, based on the proportional area of reference (0.87 km^2) and restoration habitats (0.31 km^2) within the perentie's home range.

6.3.4 Activity and Temperature

As data were not normally distributed or independent, we used a repeated measures ANOVA with a Greenhouse-Geisser correction to analyse differences in activity and temperature of the perentie between reference and restoration habitats. We recorded a high level of GPS fixes with an activity level of zero (where GPS recorded no active minutes in the 15 minutes prior to a fix), which largely represented time the perentie spent in or around a burrow. As such, we repeated our analyses with zero activity data points removed, to determine differences between usage of restoration and references habitats during the periods of time when the perentie was active within each habitat type. While ambient temperature recorded by the tracking device may not fully represent the perentie's body temperature, we used recorded temperatures to demonstrate how ecophysiological data may be integrated with T-LoCoH to understand environmental constraints to movement. All analyses for home range and movement ecology were conducted in the *R* v3.4.4 statistical environment (R Core Team, 2016), implemented using RStudio (RStudio, Inc, Boston, United States, 2019). All other analyses were conducted using SPSS Statistics version 25 (IBM, New York, United States).

6.4 Results

6.4.1 Home Range and Movement Ecology

We attained a total of 865 fixes, 503 within reference habitat and 362 within restoration habitat. Through the application of the T-LoCoH method, we identified the perentie's home range to cover 1.18km^2, in comparsion to the MCP which overestimates the home range at 2.002 km^2 (Fig. 6.5). The perentie's home range had three distinct areas of core usage; one each within reference and restoration, and one on the edge of the reference and restoration habitats (Fig. 6.6A). These areas included the perentie's burrows and refuges, and appeared to be in alternating use, with each area of the core home range visited in rotation during the tracking period. However, usage of the refuge area on the border of reference and restoration vegetation (a designated fauna refuge area composed of piles of logs, vegetation, and soil) appeared to be opportunistic, used for only one time block (three days, 10th to 13th November 2018). Use of this area was associated with a storm and flash flooding event (38mm rainfall on 10 November 2018). Core areas of the home range were characterised by a high frequency of short duration visits (Fig. 6.6B, C). Usage of the remainder of the perentie's home range radiated from core usage areas. Points with long duration visits tended to occur on the edges of the perentie's home range, with shortest duration visits centring within the core usage areas.

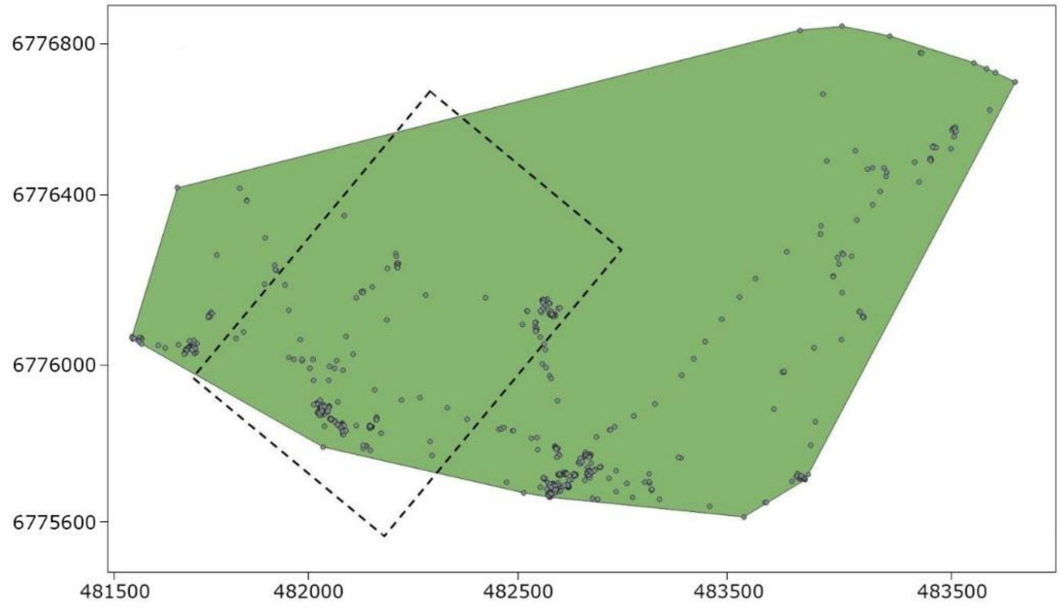

Fig. 6.5: An estimation of the perentie's home range using the Minimum Convex Polygon Method. The dashed rectangle comprises the points of utilization within the restoration habitat. X and Y axes represent UTM coordinates.

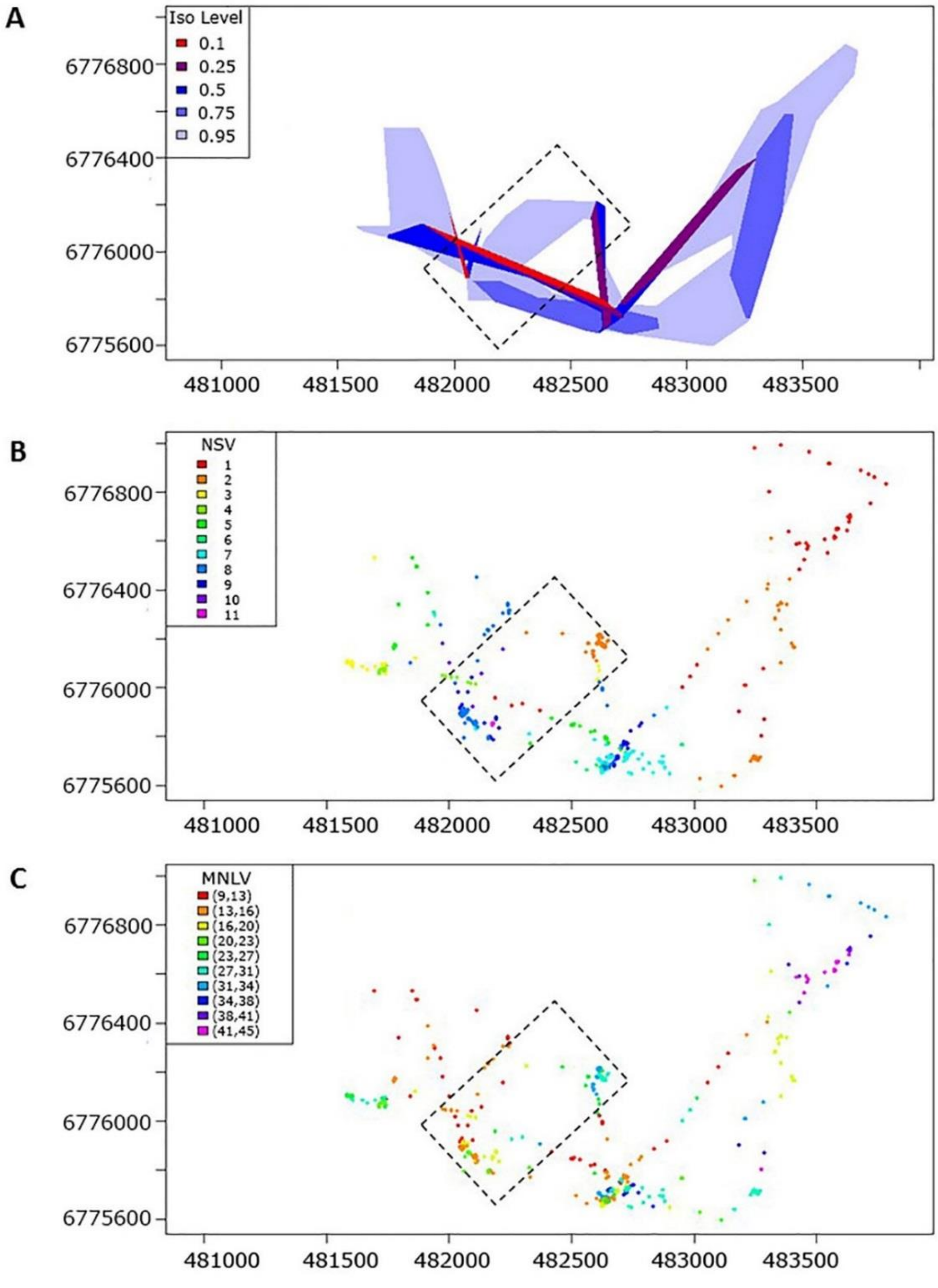

Fig. 6.6: Movement and behaviour of an individual *V. giganteus*: A) behaviour isopleths (density calculated from hull metrics: average point density falling within each hull) and core home range, B) number of visits (NSV; number of separate visits) to each point within the home range, and C) duration of visits (MNLV; mean number of locations in the hull per visit). Points are considered 'separate visits' if the IVG ≥ 12hrs. Behaviour isopleths show utilisation of regions of the home range, where higher iso levels indicate an increased likelihood of the varanid visiting a point within the hull (i.e. an iso level of 0.95 indicates a 95% chance of the varanid being located within this area at any given time within the sampling period). Density isopleths, frequency, and duration of visits were calculated using the *a*-method (s = 0.0075, *a* = 8000). X and Y axes represent UTM coordinates. The dashed rectangle within each figure comprises the points of utilisation within the restoration habitat.

Both reference and restoration habitats were utilised by the perentie; however, movement and activity varied considerably between the two habitats. Both frequency (NSV) and duration (MNLV) of visits differed significantly between reference and restoration habitats, with points in restoration visited frequently, but for shorter durations (NSV; $\chi2 = 62.98$, d.f. = 1, $P < 0.001$, MNLV; $\chi2 = 60.32$, d.f. = 1, $P < 0.001$). The perentie exhibited increased selectivity in use of restoration vegetation, typically traversing this area quickly and infrequently, venturing only short distances from the core usage area (Fig. 6B, C). The opposite was true within the reference habitat where usage was less selective. The perentie visited points within the core area of the restoration vegetation significantly more frequently than core areas in the reference vegetation ($\chi2 = 351.33$, d.f. = 1, $P < 0.001$), however for shorter durations ($\chi2 = 400.85$, d.f. = 1; $P < 0.001$). The same pattern was observed between non-core areas between reference and restoration areas of the habitat (NSV: $\chi2 = 12.43$, d.f. = 1, $P < 0.001$; MNLV: $\chi2 = 398.68$, d.f. = 1, $P < 0.001$). Overall, points in core areas were consistently frequented more often, and for a reduced duration compared to those within non-core areas (Fig. 6.7). The area within and immediately surrounding the disused mine pit was completely avoided.

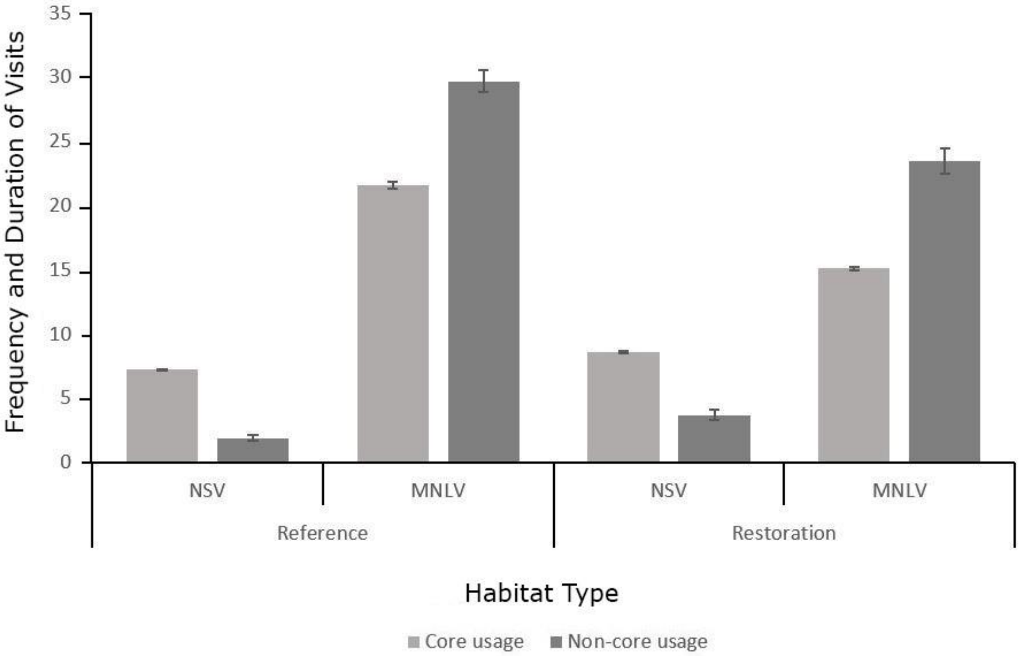

Fig. 6.7: Frequency (NSV: Number of Separate Visits) and duration (MNLV: Mean Number of Locations per Visit) of visits by the perentie to core and non-core usage areas within restoration and reference habitat.

6.4.2 Temperature and Activity

Recorded active temperatures ranged from 21°C to 40°C in reference habitat and 23°C to 40°C in restoration habitat, averaging 34.1 ± 0.14 (SE)°C and 33.8 ± 0.21 (SE)°C respectively. Ambient temperatures at times of GPS fixes did not differ significantly between reference and restoration habitat ($F_{(1,361)} = 0.60$, $P = 0.439$); however, activity level of the perentie was significantly higher in reference habitat ($F_{(1,361)} = 95.60$, $P < 0.001$). Activity within reference habitat ranged from 0 – 78% (up to 12 of 15 minutes active), with an average of 6.7 ± 0.56 (SE)%, while activity within the restoration vegetation ranged between from 0 – 64% (up to 9.5 minutes active), averaging 5.5 ± 0.57 (SE) % activity. The perentie was significantly more active within non-core usage areas within both reference ($F_{(1,114)} = 91.62$, $P < 0.001$) and restoration habitats ($F_{(1,89)} = 45.33$, $P < 0.001$), however between the two areas, activity in non-core areas tended to be higher in reference habitat, while activity in core areas was higher within restoration habitat (Fig. 6.8). The perentie spent approximately two thirds of its time in core-usage areas within each habitat. Daily patterns for temperature and activity of the perentie within reference and restoration vegetation is summarised in Fig. A4.1a,b, c (Appendix 4).

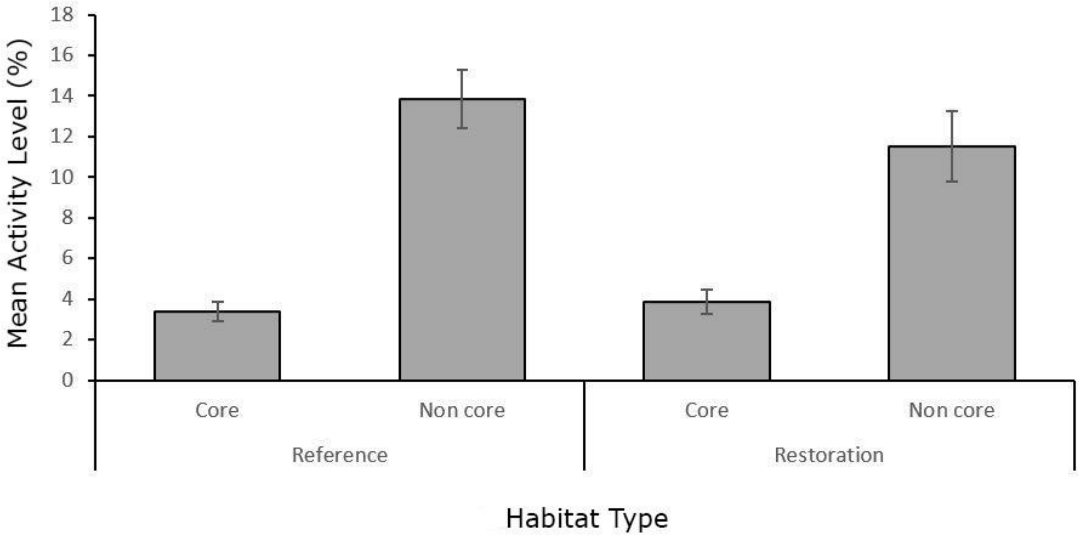

Fig. 6.8: Mean activity levels of the perentie within core and non-core usage areas of the reference and restoration habitat.

6.5 Discussion

The extent to which conclusions for ecological responses to habitat alteration may be drawn from a single animal are limited; however, T-LoCoH outputs provide a breadth of data and can aid in identifying potential mechanisms driving complex ecological interactions between animals and their environment. The movement ecology of Australian varanids has attracted sporadic attention over several decades (e.g., King, 1980; Guarino, 2002; Flesch *et al.*, 2009); however, little is understood of how these animals use either reference habitats, or anthropogenically impacted landscapes. While we are limited in drawing ecological implications by our sample size, application of T-LoCoH not only allowed for the construction of the total home range of the perentie, but in understanding the complex spatial and temporal use of restoration and reference areas habitat, such as frequency and duration of visits to points in the home range, and temperature and activity at each point, aiding in understanding the potential motivators of the responses of the perentie to habitat alteration and restoration.

6.5.1 Home Range, Movement, and Behaviour

The perentie's home range comprised three main areas of usage; the main burrow within reference habitat, a burrow on the southern side of the restoration waste rock dump, and a burrow on the edge between reference and restoration vegetation. Several *Varanus* species have previously been documented using multiple burrows or refuges such as hollow logs, termite mounds, and trees (for example, *V. panoptes, V. bengalensis*, and *V. salvator*; Wikramanayake and Dryden, 1993; Doody *et al.*, 2014). In addition to providing overnight refuges, burrows aid in the regulation of body heat and water and provide a refuge during periods of aestivation (Green, 1972; Vernet *et al.*, 1988; Doody *et al.*, 2014). Both reference and restoration habitats contained core usage points; however, actual usage across the home range differed significantly between each habitat. Despite 42% of all GPS fixes originating within restoration habitat, spatial use of this area was restricted and 65% smaller than usage in reference habitat (0.31km^2 in restoration vegetation, compared to 0.87 km^2 in reference vegetation). Outside of the core area, usage of restoration was limited with many areas either traversed infrequently, or completely avoided. Assessing the frequency and duration of visits an animal makes to points in the habitat is vital to understanding movement within, and use, of its home range (Benhamou and Riotte-Lambert, 2012).

Restoration can require significant periods of time before it resembles reference habitats, and habitats lacking established vegetation cover typically present unfavourable conditions and inadequate refuges or microclimates for thermoregulation, all of which may have made the restoration less attractive to the perentie, and account for its movement patterns (Tuff *et al.*, 2016). King (1980) notes that the activity patterns of several *Varanus* species may be strongly impacted by vegetation cover, with open and homogenous landscapes presenting increased thermal costs and restricting movement during peak temperatures.

Temperature and its effects on thermoregulation and fitness are well-studied in many reptile species (e.g., Chelazzi and Calzolai, 1986; Hertz *et al.*, 1988; Schwanz and Janzen, 2008). Some studies have assessed the relationship between body temperature and activity of varanids (e.g., Christian and Weavers, 1994, 1996); however, the relationship between landscape degradation and subsequent restoration, and thermoregulatory behaviour of ectothermic animals has rarely been studied (Tuff *et al.*, 2016). Assessing the relationship between thermal landscapes and thermoregulatory behaviours of reptiles at a variety of spatial scales is important to understanding habitat use and the factors impacting habitat selection (Row and Blouin-Demers, 2006). By overlaying environmental temperature recordings with our perentie's movement data in T-LoCoH, we demonstrated an effective method for analysing how habitat degradation and restoration may impact thermoregulatory behaviour and movement. Studies that do not account for environmental factors may conclude that, due to both habitats recording similar temperatures, temperature is not a constraint within restored habitats. Although ambient temperature may not fully represent body temperature, our data suggest that movement and activity patterns of the perentie within, and between, each habitat type differed significantly, despite ambient temperature not differing significantly between the habitats. We note high selectivity of habitat use by the perentie in restoration habitat, with this area functioning largely as a shelter site, whereas reference habitat facilitated increased movement and longer distance and duration foraging trips. Sears *et al*. (2011) notes that spatial heterogeneity in the thermal landscape can influence thermoregulatory costs, even in situations where temperatures within a landscape do not differ statistically. Particularly for ectothermic animals, where temperature and metabolic costs can be significant constraints to movement, understanding how temperature affects movement and activity within altered habitats is vital to their conservation.

6.5.2 Benefits of T- LoCoH and GPS Technology

Compared to the insights that we gained using the T-LoCoH method, conclusions drawn from our data based upon only points of presence of the perentie would have resulted in a significant misrepresentation of the perentie's home range (activity area). To begin with, the home range estimated by the MCP method was 170% larger than that estimated by T-LoCoH, with almost the entirety of the restoration area included in the home range. Over-estimation of home ranges is a common issue reported for studies using the MCP method (Pimley *et al.*, 2005; Ciofi *et al.*, 2007; Downs and Horner, 2008). Furthermore, roughly half of all GPS fixes for the perentie's movements were recorded within restoration, and a study of species presence alone would conclude similar use of both habitats. Through applying the T-LoCoH method, we identified that, although facilitating return of the perentie, the restoration was used with greater selectivity in comparison to reference habitat, supporting shorter distance and duration movement. Presence/absence studies, by comparison, are limited in their ability to provide an understanding of the environmental influences driving behavioural responses to habitat alteration, or in understanding movement of animals through landscapes (Mackenzie, 2005; Cross *et al.*, 2019). T-LoCoH provides a reliable method for identifying points of the habitat that are used frequently compared to those that are visited infrequently or only in opportunistic use. For example, our data suggested that while the perentie frequented parts of the restoration vegetation, several areas including the top of waste rock dump appeared to be avoided. This area of the habitat tended to comprise sparse vegetation and increased spatial homogeneity, and as such was likely avoided by the perentie due to a lack of appropriate microclimates for thermoregulatory behaviours. Understanding the behavioural responses of animals to changing habitats, particularly those which may have increased susceptibility to fluctuating temperatures, is key to facilitating their conservation in altered landscapes undergoing restoration (Cross *et al.*, 2019).

In assessments of home ranges and movements of animals, VHF tracking alone is less effective than when combined with GPS technology, due to its tendency to cause significant disruption to the study animal and its natural behaviour (Cooke *et al.*, 2004;

Thomas *et al.,* 2011). VHF tracking also requires multiple relocations of the animal each day to understand movement ecology, and hence constant human presence can disrupt natural behaviour and may result in a false representation of habitat use. Combining GPS technology with the T-LoCoH method allowed the collection of continuous data on the perentie's movements with minimal impact to the natural behaviour of the animal, due to requiring comparatively infrequent locations by people. Furthermore, T-LoCoH reduces the impacts of temporal autocorrelation on data through the incorporation of an IVG component, and visits to points of the habitat are only considered separate if a period greater than the IVG has elapsed between samples (Lyons, 2014).

Ecological and behavioural responses may differ between individuals and our data may not reflect responses of the wider population of *V. giganteus*, particularly since our focal animal was a young adult, and hence may have a smaller home range than that of an adult individual, as has previously been reported among studies of reptile home range (e.g., Semlitsch, 1981; Diemer, 1992; Rocha, 1999). Regardless, our study demonstrates how VHF and GPS tracking provides a depth of data and a measure of continued assessments of movement ecology which, through the application of the T-LoCoH method, can aid in understanding movement and behavioural responses of animals to habitat change and restoration. In assessments of faunal responses to habitat change and restoration, measures of species presence, absence, or abundance, or construction of simple spatial home ranges are unlikely to adequately show whether restoration sites are facilitating long-term use and population persistence (Cross *et al.*, 2019). However, T-LoCoH can show whether habitat restoration is supporting natural behaviour, or whether it provides suboptimal habitat for support of animal populations. Although conclusions drawn concerning the success or failure of restoration efforts from a single animal may not be valid, data from our study suggest that while restoration is facilitating return, these areas are used with increased selectivity, and behaviour of the perentie differed significantly to that within reference vegetation.

6.5.3 Conclusions

T-LoCoH is a highly effective method for constructing complex home ranges of animals, and for understanding how movement and habitat use may be influenced or constrained by environmental factors. The T-LoCoH method allowed for not only the construction of the perentie's total home range, but understanding the complex behavioural responses of the perentie to habitat alteration. The use of T-LoCoH is not restricted to assessments of thermal responses and behaviour of ectothermic animals but may be applied to a wide range of fauna, habitats, and disturbances. Contrary to conclusions drawn from assessments of presence and absence, which may conclude roughly equal use between restoration and reference habitats by our perentie, T-LoCoH analyses allowed for the identification of differential habitat use, core usage areas, and areas with limited use, or that were avoided completely. We highlight that although restoration may be facilitating return of a high order reptilian predator, behavioural use of restoration differs from that in reference habitat. Understanding the behavioural responses animals, in addition to their presence or absence from habitats, is key to facilitating their conservation in the face of increasing rates of habitat degradation. The T-LoCoH method of home range construction provides a useful measure for understanding the complex interactions of animals with their environments.

6.6 References

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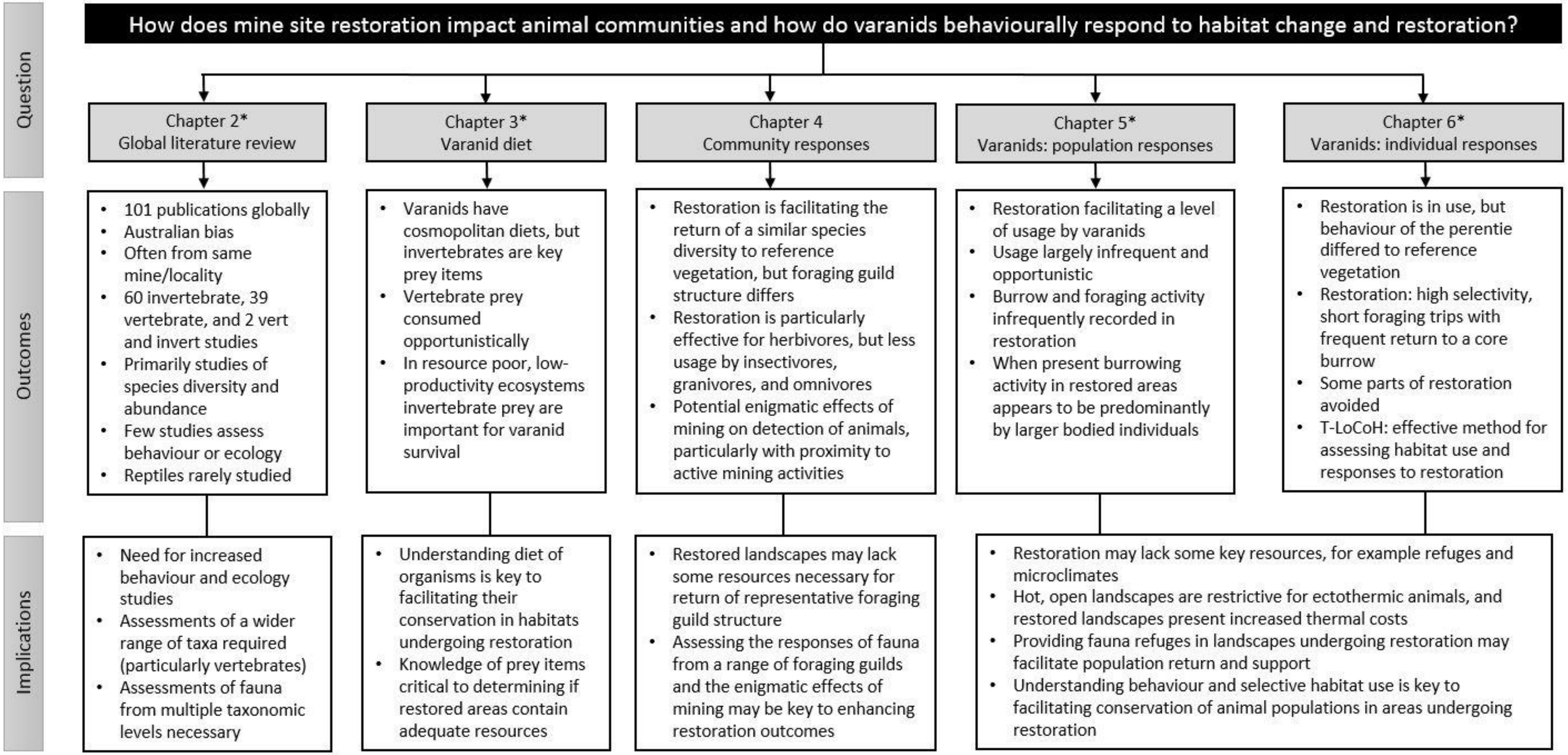
Chapter 7. General Discussion

‘Petra’ the perentie

7.1 Summary of findings

This thesis represents the first study of the behavioural and ecological responses of fauna, primarily varanids, to habitat restoration following the cessation of mining activities. In this thesis, I demonstrate how assessments of animal behaviour and ecology can provide critical insights into ecosystem functionality and assess the community responses of animals, and the behavioural and ecological responses of varanids to habitat restoration. I also provide a critical analysis of a novel method for analysing home range data to understand habitat use and the behavioural responses of varanids to habitat change and restoration. The tool should be broadly applicable to other taxa. In this discussion I summarise the main findings of this thesis and discuss how the results of this research may be used to inform industry and restoration practitioners on conservation and management strategies to increase the likelihood of achieving successful restoration outcomes for fauna (Fig. 7.1). This thesis demonstrates that understanding the complex mechanisms underpinning the responses of fauna to habitat change and restoration is key to determining whether restoration efforts effectively support the return of fauna populations. I advocate for increased consideration and monitoring of fauna, particularly of their behavioural responses, in assessments of habitat restoration success following the discontinuation of mining activities.



How does mine site restoration impact animal communities and how do varanids behaviourally respond to habitat change and restoration?

Chapter 2*
Global literature review

- 101 publications globally
- Australian bias
- Often from same mine/locality
- 60 invertebrate, 39 vertebrate, and 2 vert and invert studies
- Primarily studies of species diversity and abundance
- Few studies assess behaviour or ecology
- Reptiles rarely studied

- Need for increased behaviour and ecology studies
- Assessments of a wider range of taxa required (particularly vertebrates)
- Assessments of fauna from multiple taxonomic levels necessary

Chapter 3*
Varanid diet

- Varanids have cosmopolitan diets, but invertebrates are key prey items
- Vertebrate prey consumed opportunistically
- In resource poor, low-productivity ecosystems invertebrate prey are important for varanid survival

- Understanding diet of organisms is key to facilitating their conservation in habitats undergoing restoration
- Knowledge of prey items critical to determining if restored areas contain adequate resources

Chapter 4
Community responses

- Restoration is facilitating the return of a similar species diversity to reference vegetation, but foraging guild structure differs
- Restoration is particularly effective for herbivores, but less usage by insectivores, granivores, and omnivores
- Potential enigmatic effects of mining on detection of animals, particularly with proximity to active mining activities

- Restored landscapes may lack some resources necessary for return of representative foraging guild structure
- Assessing the responses of fauna from a range of foraging guilds and the enigmatic effects of mining may be key to enhancing restoration outcomes

Chapter 5*
Varanids: population responses

- Restoration facilitating a level of usage by varanids
- Usage largely infrequent and opportunistic
- Burrow and foraging activity infrequently recorded in restoration
- When present burrowing activity in restored areas appears to be predominantly by larger bodied individuals

Chapter 6*
Varanids: individual responses

- Restoration is in use, but behaviour of the perentie differed to reference vegetation
- Restoration: high selectivity, short foraging trips with frequent return to a core burrow
- Some parts of restoration avoided
- T-LoCoH: effective method for assessing habitat use and responses to restoration

- Restoration may lack some key resources, for example refuges and microclimates
- Hot, open landscapes are restrictive for ectothermic animals, and restored landscapes present increased thermal costs
- Providing fauna refuges in landscapes undergoing restoration may facilitate population return and support
- Understanding behaviour and selective habitat use is key to facilitating conservation of animal populations in areas undergoing restoration

Fig. 7.1: Conceptual framework of the outcomes and implications of the research of this thesis. *Published papers

In Chapter 2 I investigated the extent to which fauna are considered in assessments of restoration success globally. Animals are broadly overlooked in assessments of mine site restoration success; I identified 101 published studies globally assessing the responses of animals to mine site restoration (Chapter 2; Cross *et al.*, 2019a). I identified several biases and shortcoming among the existing literature, most notably that over half of all studies originated within Australia. Australia is a global leader in fauna monitoring in assessments of mine site restoration success and is one of the few countries with comprehensive legislative requirements and non-compliance penalties post-restoration and non-compliance penalties (Gilbert, 2000; Clark and Clark, 2005; Cross *et al.*, 2019a). Although Australia leads the world in post-mining restoration research, including the study of fauna, a high proportion of the literature originates from a single mining operation in the southwest region of Western Australia. The trend of large volumes of research originating from small numbers of sites is, however, the global norm, with much of the literature either stemming from a single mine site, or a single locality within a country (Chapter 2; Cross *et al.*, 2019a). It is critical that research be expanded across a variety of ecosystems globally, to gain a more diverse understanding of how habitat restoration impacts animals and ecosystem function in a range of climates and habitats (Chapter 2; Cross *et al.*, 2019a). Another significant limitation among the existing literature reporting faunal responses to mine site restoration is the strong focus towards assessments of species richness, presence, or absence in habitats (Chapter 2; Cross *et al.*, 2019a). Such studies may be limited in their ability to assess the long-term functionality of restored habitats, or whether these areas are supporting animal populations or are only in opportunistic use (Chapter 2; Cross *et al.*, 2019a). Assessments of behaviour and ecological responses of animals to habitat restoration are critical to understanding whether habitat restoration facilitates the return of functional, self-sustaining animal populations (Lindell, 2008; Cross *et al.*, 2019a, 2020a).

Diet is a fundamental component of animal ecology and can provide insight into their fundamental resource requirements (Sih and Christensen, 2001; Cross *et al.*, 2019b). In Chapter 3, I assessed the diets of three common sympatric species of *Varanus* (*V. tristis*, *V. gouldii*, and *V. panoptes*) in the Mid West region of Western Australia. I identified significant overlap in the diets of each of the three species, invertebrate prey (particularly Orthoptera) appeared to be critical to all three species (Chapter 3; Cross *et al.*, 2019b). Unlike large mammalian predators, which require frequent consumption of high energy prey, varanids can survive on relatively infrequent feeds, and aestivate when environmental conditions become unfavourable (Morton and James, 1988; Christian *et al.*, 1999; Doody *et al.*, 2014), which has likely driven their success in arid, resource poor habitats (Pianka, 1981; Stafford Smith and Morton, 1990). Understanding the diets of co-existing, sympatric species, particularly those living in resource poor, low productivity habitats, is key to determining how populations can thrive in challenging environmental conditions (Chapter 3; Cross *et al.*, 2019b).

Understanding how animals from a wide range of taxa respond to habitat restoration is critical in determining whether restoration efforts facilitate the return of fauna communities. In Chapter 4, I assessed how the direct and indirect effects of mining influenced animals and foraging guild structure in restored and reference vegetation. Habitat restoration following the cessation of mining activities appeared to facilitate the return of a similar species diversity to the reference bushland; however, the foraging guilds present in restoration vegetation differed significantly to within the reference community (Chapter 4). Restoration heavily favoured macropod herbivores and species well-adapted to anthropogenically influenced landscapes. Restored habitats often comprise vegetation at early successional stages and have an increased abundance of seedlings and saplings (Pywell *et al.*, 2002; Baer *et al.*, 2004). Being at early successional stages, restoration habitats may lack some key resources necessary for the return of representative fauna communities, for example refuges, vegetation cover, and coarse woody debris (Tuff *et al.*, 2016; Craig *et al.*, 2014; Cross *et al.*, 2020b). I detected higher abundances of all foraging guilds in the sites distant from the active mining operation compared to a site nearby, suggesting some effects in habitats immediately surrounding disturbances (Folchi, 2003; Longcore and Rich, 2004; Tyler *et al.*, 2014; Raiter *et al.*, 2014).

While the assessment of animal communities to habitat change can be informative, it has its limits. In Chapters 5 and 6, I assessed how large, predatory reptiles responded behaviourally to habitat change and restoration, to identify the specific habitat requirements for their return following restoration. In Chapter 5, I identified that, while the restoration of degraded lands within the study region appeared to support varanids, they used restoration landscapes infrequently and opportunistically, rarely foraging or burrowing. Furthermore, it appeared that only large varanids burrowed in restoration landscapes, which I ascribed to their greater temperature tolerance than small varanids (Cowles and Bogert, 1944). My data suggest that restored habitats may lack the structural heterogeneity and microhabitats necessary to support the return and use by varanids of diverse sizes and ages (Chapter 5; Spotila *et al.*, 1973; Stevenson, 1985; Huey and Bennett, 1990).

In Chapter 6, I reported on the Time Local Convex Hull (T-LoCoH) method of home range analysis to estimate an animal's home range and to assess the complex interactions of animals with their environment (Chapter 6; Cross *et al.,* 2020b). I identified that while restored habitats facilitated the return of perenties (*V. giganteus*), their behaviour and movement in the restored landscape differed to that in the reference vegetation (Chapter 6; Cross *et al.,* 2020b). Spatially homogenous restoration landscapes can present significantly higher metabolic costs for reptiles than heterogenous landscapes, even when these landscapes do not differ statistically in temperature (Sears *et al.*, 2011; Tuff *et al.*, 2016). While restored landscapes were used by my focal perentie, these areas and tended to be crossed infrequently and quickly, with frequent returns to a core area of the home range necessary for thermoregulation (Chapter 6; Cross *et al.*, 2020b). Understanding how animals respond behaviourally to habitat change and restoration is key to identifying critical resources for population support (Lindell, 2008; Cross *et al.*, 2020a), and the T-LoCoH method provides an effective method for understanding how animals respond to habitat change and restoration, and their behaviour and movement ecology within habitats (Chapter 6; Cross *et al.*, 2020b).

7.2 Implications for industry, management, and conservation

Increased focus has recently been placed upon assessing fauna return to restored landscapes (Majer, 1989; Lindell, 2008; Majer, 2009; Cross *et al.*, 2019a, 2020a); however, despite numerous calls for their greater consideration in restoration monitoring, animals remain broadly overlooked in assessments of mine site restoration success (Cross *et al.*, 2019a, 2020a). The results from this thesis demonstrate a clear need for the incorporation of fauna responses in assessments of mine site restoration success. Although assessments of species presence or absence in restored landscapes provides valuable ecological information, such studies are limited in their ability to demonstrate whether restoration supports fauna populations or is only opportunistically used by a select few taxa (Lindell, 2008; Cross *et al.*, 2019a, 2020a). Given the inherent complexity of ecosystems and the interactions between plants and animals, the assumption of the unassisted return of fauna communities and restitution of population dynamics post restoration is unlikely without informed ecological interventions (Cross *et al.*, 2020a). Understanding how animals respond behaviourally to habitat change is key to determining critical resources supporting populations, and to returning functional, biodiverse communities to restored landscapes. Without consideration of the complex interactions between animals and their environment, restoration efforts may ultimately fail in returning fully functional, representative, and diverse ecosystems.

7.3 Thesis conclusions and future directions

Restoration of a discontinued mine site within the Mid West region of Western Australia has facilitated some return and usage by animals. However, the restored landscapes appear to lack some key resources necessary for the return of animal community dynamics and return of varanids. Reptiles require spatially heterogenous landscapes with an abundant refuges and microclimates for thermoregulatory behaviours (Tuff *et al.*, 2016), and the increased homogeneity and thermal costs of restored landscapes restricts movement and usage of such landscapes by varanids (Cross *et al.*, 2020b). Providing more fauna refuges, such as hollow logs, in restoration areas may aid the return of fauna populations, particularly for ectothermic species (Cross *et al.*, 2020b). Future research should consider assessments of the selective habitat use, movement, and behaviour of animals from a wide range fauna groups and consider the enigmatic impacts of mining (for example noise, dust, vibrations) that may alter their behaviour and restrict return to restored landscapes. Future research should also consider the long-term responses of fauna to habitat change and restoration, as restoration success and the return of different taxa may depend on the time since restoration. Fauna are integral to functioning ecosystems and restoration monitoring and planning must consider fauna to a greater extent, both in their presence and behaviour, if the goal of restoration is to achieve not only representative vegetation communities and structure, but biodiverse, functional, and self-sustaining ecosystems.

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Appendix 1. Copyright Statements

Chapter 2: Overlooked and undervalued: the neglected role of fauna and a global bias in ecological restoration assessments

To Whom it May Concern, I Sophie Louise Cross collected and analysed the data; I led the writing of the manuscript; all authors contributed to the writing and revisions of the manuscript for the following publication:

> **Cross, S.L.**, Tomlinson, S., Craig, M.D., Dixon, K.W. and Bateman, P.W. (2019). Overlooked and undervalued: the neglected role of fauna and a global bias in ecological restoration assessments. *Pacific Conservation Biology* **25**, 331-341. doi: 10.1071/PC18079.

Signature: Sophie Cross Date: 17/02/2020

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

Assoc. Prof Philip W. Bateman

Signature: Date: 17/02/2020

Dr Sean Tomlinson

Signature: Sean Tomlinson Date: 18/02/2020

Dr Michael D. Craig

Signature: Michael Craig Date: 21/02/2020

Prof Kingsley W. Dixon

Signature: Date: 17/02/2020

Chapter 3: I don’t like crickets, I love them: invertebrates are an important prey source for varanid lizards

To Whom it May Concern, I Sophie Louise Cross collected the data, I analysed the data with guidance from MDC, ST, and PWB; I led the writing of the manuscript; all authors contributed to the writing and revisions of the manuscript for the following publication:

> **Cross, S.L.**, Craig, M.D., Tomlinson, S. and Bateman, P.W. (2019). I don’t like crickets, I love them: invertebrates are an important prey source for varanid lizards. *Journal of Zoology*. Online Early. doi: 10.1111/jzo.12750

Signature: Sophie Cross Date: 17/02/2020

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Assoc. Prof Philip W. Bateman

Signature: Date: 17/02/2020

Dr Sean Tomlinson

Signature: Sean Tomlinson Date: 18/02/2020

Dr Michael D. Craig

Signature: Michael Craig Date: 21/02/2020

Chapter 4: Build it and some may come: restoration of discontinued mine sites initially favours herbivores

To Whom it May Concern, I Sophie Louise Cross collected the data, I analysed the data with guidance from MDC, ST, and PWB; I led the writing of the manuscript; all authors contributed to the writing and revisions of the manuscript for the following publication:

> **Cross, S.L.**, Craig, M.D., Tomlinson, S. and Bateman, P.W. Build it and some may come: restoration of discontinued mine sites initially favours herbivores. *In preparation for submission*.

Signature: Date: 17/02/2020

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

Assoc. Prof Philip W. Bateman

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Dr Michael D. Craig

Signature: Date: 21/02/2020

Chapter 5: Using monitors to monitor ecological restoration: presence may not indicate persistence

To Whom it May Concern, I Sophie Louise Cross collected the data; I analysed the data with guidance from MDC, ST, and PWB**;** I led the writing of the manuscript; all authors contributed to the writing and revisions of the manuscript for the following publication:

> **Cross, S.L.**, Tomlinson, S., Craig, M.D., Dixon, K.W. and Bateman, P.W. Using monitors to monitor ecological restoration: presence may not indicate persistence. *Austral Ecology*. **In Press.**

Signature: Date: 17/02/2020

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

Assoc. Prof Philip W. Bateman

Signature: Date: 17/02/2020

Dr Sean Tomlinson

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Dr Michael D. Craig

Signature: Date: 21/02/2020

Prof Kingsley W. Dixon

Signature: Date: 17/02/2020

Chapter 6: The Time Local Convex Hull (T-LoCoH) method as a tool for assessing the responses of fauna to habitat change: a case study using the perentie (*Varanus giganteus*: Reptilia: Varanidae).

To Whom it May Concern, I Sophie Louise Cross collected the data, I analysed the data with guidance from MDC, ST, and PWB; I led the writing of the manuscript; all authors contributed to the writing and revisions of the manuscript for the following publication:

> **Cross, S.L.**, Tomlinson, S., Craig, M.D. and Bateman P.W. (2020). The Time Local Convex Hull (T-LoCoH) method as a tool for assessing the responses of fauna to habitat change: a case study using the perentie (*Varanus giganteus*: Reptilia: Varanidae). *Australian Journal of Zoology*. Online Early. doi: 10.1071/ZO19040.

Signature: Date: 17/02/2020

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

Assoc. Prof Philip W. Bateman

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Dr Sean Tomlinson

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Dr Michael D. Craig

Signature: Date: 21/02/2020

Appendix 2. Overlooked and undervalued: the neglected role of fauna and a global bias in ecological restoration assessments

Table A2.1: Summary of the 94 peer-reviewed publications used in this review.

Authors	Date	Continent	Group	Target	Mineral
Andersen	1993	Australasia	Invertebrate	Formicidae	Uranium
Andersen and Sparling	1997	Australasia	Invertebrate	Formicidae	Uranium
Andersen *et al*.	2003	Australasia	Invertebrate	Formicidae	Coal
Andres and Mateos	2006	Europe	Invertebrate	Multiple	Limestone
Armstrong and Nichols	2000	Australasia	Vertebrate	Multiple	Bauxite
Becker *et al.*	2013	South America	Vertebrate	Multiple	Not stated
Bisevac and Majer	1999a	Australasia	Invertebrate	Formicidae	Sand
Bisevac and Majer	1999b	Australasia	Invertebrate	Multiple	Sand
Brady and Noske	2010	Australasia	Vertebrate	Multiple	Bauxite
Brandle *et al*.	2000	Europe	Invertebrate	Coleoptera	Coal
Bulluck and Buehler	2006	North America	Vertebrate	Multiple	Coal
Burgar *et al.*	2017	Australasia	Vertebrate	*Chiroptera*	Bauxite
Comer and Wooller	2002	Australasia	Vertebrate	Multiple	Sand
Craig *et al.*	2007	Australasia	Vertebrate	*Pogona minor*	Bauxite
Craig *et al.*	2010	Australasia	Vertebrate	Multiple	Bauxite
Craig *et al.*	2011	Australasia	Vertebrate	*Egernia napoleonis*	Bauxite
Craig *et al.*	2014	Australasia	Vertebrate	Multiple	Bauxite
Craig *et al.*	2015	Australasia	Vertebrate	Multiple	Bauxite
Cristescu *et al.*	2013	Australasia	Vertebrate	*Phascolarctos cinereus*	Sand
Cuccovia	1999	Australasia	Invertebrate	Acarine	Bauxite
Curry and Nichols	1985	Australasia	Vertebrate	Multiple	Bauxite
Cusser and Goodell	2013	North America	Invertebrate	Multiple	Coal
Davis *et al.*	2003	Africa	Invertebrate	Coleoptera	Sand
Davis *et al.*	2013	Africa	Invertebrate	Coleoptera	Not stated
Doherty *et al.*	2016	Australasia	Vertebrate	*Calyptorhynchus* sp.	Bauxite
Dominguez-Haydar and Armbrecht	2011	South America	Invertebrate	Formicidae	Coal
Dunger *et al.*	2001	Europe	Invertebrate	Multiple	Coal
Dunger *et al.*	2004	Europe	Invertebrate	Collembola	Coal

Ferreira and Van Aarde	1996	Africa	Vertebrate	Multiple	Not stated
Fox and Fox	1978	Australasia	Vertebrate	Multiple	Sand
Fox and Fox	1984	Australasia	Vertebrate	Multiple	Sand
Fox and Twigg	1991	Australasia	Vertebrate	Mice	Sand
Frick *et al.*	2014	Australasia	Vertebrate	Multiple	Sand
Galan	1997	Europe	Vertebrate	Multiple	Coal
Gould	2011	Australasia	Vertebrate	Multiple	Bauxite
Gould and Mackey	2015	Australasia	Vertebrate	Multiple	Bauxite
Greenslade and Majer	1980	Australasia	Invertebrate	Collembola	Bauxite
Greenslade and Majer	1993	Australasia	Invertebrate	Collembola	Bauxite
Hamilton and Vimmerstedt	1981	America	Invertebrate	Annelida	Not stated
Hendrychová et al.	2012	Europe	Invertebrate	Multiple	Coal
Hill and Phinn	1993	Australasia	Vertebrate	Swamp Wallabies	Sand
Holl	1995	North America	Invertebrate	Lepidopteran	Coal
Holl	1996	North America	Invertebrate	Lepidopteran	Coal
Jackson and Fox	1996	Australasia	Invertebrate	Formicidae	Sand
Kielhorn *et al.*	1999	Europe	Invertebrate	Coleoptera	Coal
Koch et al.	2010	Australasia	Invertebrate	Multiple	Bauxite
Kumssa *et al.*	2004	Africa	Invertebrate	Multiple	Multiple
Lannoo *et al.*	2009	North America	Vertebrate	Multiple	Coal
Lee *et al.*	2013	Australasia	Vertebrate	*Calyptorhynchus* sp.	Multiple
Letnic and Fox	1997a	Australasia	Vertebrate	Multiple	Sand
Letnic and Fox	1997b	Australasia	Vertebrate	Multiple	Sand
Lythe *et al.*	2017	Australasia	Invertebrate	Multiple	Bauxite
Madden and Fox	1997	Australasia	Invertebrate	Multiple	Sand
Majer	1981	Australasia	Invertebrate	Multiple	Bauxite
Majer	1984a	Australasia	Invertebrate	Formicidae	Bauxite
Majer	1984b	Australasia	Invertebrate	Formicidae	Bauxite
Majer	1985	Australasia	Invertebrate	Formicidae	Sand
Majer	1996	South America	Invertebrate	Formicidae	Bauxite
Majer and Nichols	1998	Australasia	Invertebrate	Formicidae	Bauxite
Majer *et al.*	1982	Australasia	Invertebrate	Formicidae	Sand
Majer *et al.*	1985	Australasia	Invertebrate	Formicidae	Iron ore
Majer	1989	Australasia	Invertebrate	Multiple	Bauxite
Majer *et al.*	2013	Australasia	Invertebrate	Formicidae	Bauxite
Mawson	1986	Australasia	Invertebrate	Araneae	Bauxite

Moir *et al*.	2005	Australasia	Invertebrate	Hemiptera	Bauxite
Nichols and Bamford	1985	Australasia	Vertebrate	Multiple	Bauxite
Nichols and Burrows	1985	Australasia	Invertebrate	Multiple	Bauxite
Nichols and Nichols	2003	Australasia	Both	Multiple	Bauxite
Nichols and Watkins	1984	Australasia	Vertebrate	Multiple	Bauxite
Ottonetti *et al.*	2006	Europe	Invertebrate	Formicidae	Coal
Passell	2000	Asia	Vertebrate	Multiple	Tin
Petersen *et al.*	2016	North America	Vertebrate	Sage Grouse	Coal
Pižl	2001	Europe	Invertebrate	Oligochaeta	Coal
Purger *et al.*	2004a	Europe	Vertebrate	Pheasant, Nightjar, Yellowhammer	Coal
Purger *et al.*	2004b	Europe	Vertebrate	Quail	Coal
Purger *et al.*	2007	Europe	Invertebrate	Multiple	Coal
Redi *et al*.	2005	Africa	Invertebrate	Diplopoda	Multiple
Ribas *et al.*	2012	South America	Invertebrate	Formicidae	Gold
Rufaut *et al.*	2006	Australasia	Invertebrate	Multiple	Coal
Rufaut *et al.*	2015	Australasia	Invertebrate	Coleoptera	Coal
Seifert and Prosche	2017	Europe	Invertebrate	Formicidae	Not stated
Sieg *et al.*	1987	North America	Invertebrate	Multiple	Bentonite
Simmonds *et al.*	1994	Australasia	Invertebrate	Araneae	Bauxite
Taillefer and Wheeler	2012	North America	Invertebrate	Multiple	Peat
Tajovský	2001	Europe	Invertebrate	Multiple	Coal
Taylor and Fox	2001	Australasia	Vertebrate	Multiple	Sand
Thompson and Thompson	2003	Australasia	Vertebrate	*Pogona minor*	Not stated
Thompson and Thompson	2005	Australasia	Vertebrate	Multiple	Gold
Tizado and Núñez-Pérez	2016	Europe	Invertebrate	Multiple	Coal
Topp *et al.*	2001	Europe	Invertebrate	Multiple	Coal
Topp *et al.*	2010	Europe	Invertebrate	Coleoptera	Coal
Triska *et al.*	2016	Australasia	Vertebrate	Multiple	Bauxite
Twigg and Fox	1991	Australasia	Vertebrate	Multiple	Sand
van Arde *et al.*	1996a	Africa	Both	Multiple	Multiple
van Arde *et al.*	1996b	Africa	Invertebrate	Diplopoda	Multiple
Van Schagen	1986	Australasia	Invertebrate	Formicidae	Coal

Wassenaar *et al.*	2005	Africa	Invertebrate	Multiple	Multiple
Watts *et al.*	2008	Australasia	Invertebrate	Coleoptera	Peat
Wheater and Cullen	1997	Europe	Invertebrate	Multiple	Limestone
Wykes	1985	Australasia	Vertebrates	Aves	Bauxite
Zeppelini *et al.*	2009	America	Invertebrate	Collembola	Sand

Table A2.2: Summary of the accessible grey literature surrounding fauna and mine site restoration

Authors	Date	Document type	Mention to fauna	Direct reference to fauna monitoring in restoration?
Glenn *et al.*	2014	Bulletin	Only mentions current monitoring methods overlook fauna, and fauna monitoring is not common	No
Guinea	2007	Management Plan	Management plan for marine turtles	No
Knuckey	2017	Monitoring plan	Monitoring plan for ghost bats	No
McLaughlin	2012	Rehabilitation strategy	Return of habitat corridors necessary for fauna populations, document references attempt of the project to minimise impacts to threatened fauna	No
Weipa	2015	Management plan	Management plan for feral pigs	No
Brennan *et al.*	2005	Guidance document	Methods for increasing fauna return to rehabilitating sites	Yes- techniques for promoting fauna return to rehabilitated sites following mining
Majer	1997	Book chapter	Invertebrates in the restoration process	Yes- An Australian perspective on the role of invertebrates in the mining restoration process
Majer	1998	Conference proceedings	Return of animals to reclaimed mine lands	Yes- Long term patterns of fauna recolonization after mine rehabilitation, and potential influencing factors

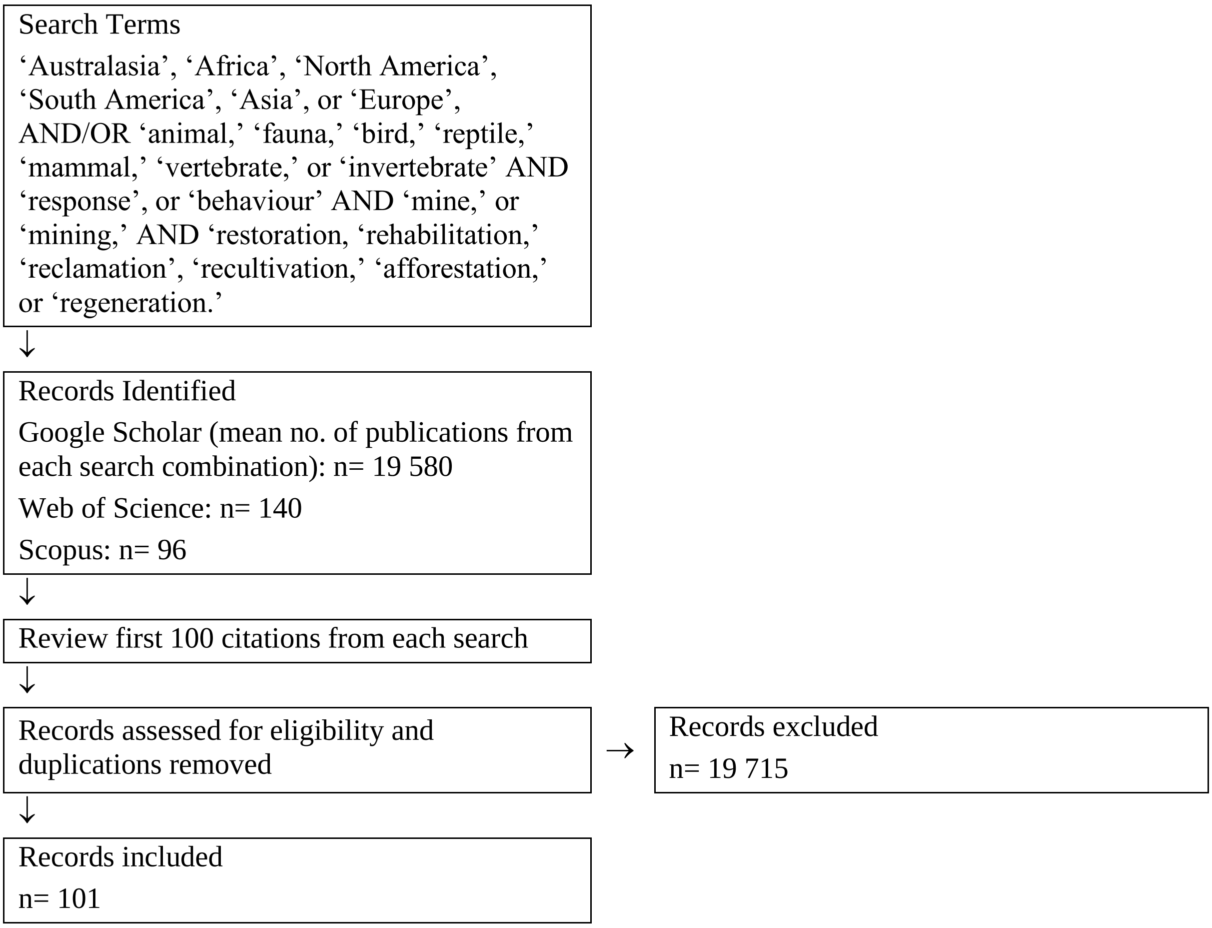
McKee	2007	Conference presentation	Establishment of burrows and nests in reclaimed mine sites to promote growth of raptors and mountain plovers	Yes- construction of burrows and nests in reclamation sites
Moloney *et al.*	1998	Conference proceedings	Mammal distribution after fire and mining	Yes- study of small mammal recolonization in revegetated mined areas
Nawrot and Klimstra	1989	Book chapter	Wetland habitats on developed mines	Yes- Reintroduction of birds in reclaimed mines
Nickel and Claremont	2015	Monitoring plan	Monitoring plan for conservation significant fauna.	Yes- monitoring to be conducted in and around rehabilitation
Tibbett	2015	Book chapter	Mentions vertebrate studies from Alcoa, and the key role fauna play in ecosystem processes	Yes- basic overview of percentage of vertebrates recolonising rehabilitated sites (from published literature)

Table A2.3: Breakdown of terminologies used across literature assessing fauna responses to mining.

Main Terminology	Definition	Literature
Rehabilitation	"Direct or indirect actions with the aim of reinstating a level of ecosystem functionality where ecological restoration is not sought but rather renewed and ongoing provision of ecosystem goods and services" – McDonald *et al.*, 2016	Andersen *et al.*, 2003, Armstrong and Nichols, 2000, Bisevac and Majer, 1999a,b, Brady and Noske, 2010, Comer and Wooller, 2002, Cristescu *et al.*, 2013, Cuccovia, 1999, Curry and Nichols, 1986, Davis *et al.*, 2013, Dominguez-Haydar and Armbrecht, 2011, Ferreira and Van Aarde, 1996, Gould, 2011, Gould and Mackey, 2015, Greenslade and Majer, 1993, Hill and Phinn, 1993, Jackson and Fox, 1996, Koch *et al.*, 2010, Kumssa *et al.*, 2004, Lee *et al.*, 2013, Letnic and Fox, 1997b, Majer, 1981, 1984a,b, 1985, 1996, Majer and Nichols, 1998, Majer *et al.*, 1982, 1984, 1985, Mawson, 1986, Nichols and Bamford, 1985, Nichols and Burrows, 1985, Nichols and Nichols, 2003, Nichols and Watkins, 1984, Ottonetti *et al.*, 2006, Redi *et al.*, 2005, Ribas *et al.*, 2012, Rufaut *et al.*, 2006, 2015, Simmonds *et al.*, 1994, Thompson and Thompson, 2003, 2005, Van Aarde *et al.*, 1996a,b, Van Schagen, 1986, Wassenaar *et al.*, 2005, Wykes, 1985
Restoration	"The process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed" – Clewell *et al.*, 2004; McDonald *et al.*, 2016	Andersen, 1993, Andersen and Sparling, 1997, Andres and Mateos, 2006, Becker *et al.*, 2013, Burgar *et al.*, 2017, Craig *et al.*, 2007, 2011, 2014, 2015, Frick *et al.*, 2014, Lythe *et al.*, 2017, Majer *et al.*, 2013, Moir *et al.*, 2005, Passell, 2000, Petersen *et al.*, 2016, Taillefer and Wheeler, 2012, Tizado and Núñez-Pérez, 2016, Triska *et al.*, 2016, Watts *et al.*, 2008, Wheater and Cullen, 1997, Zeppelini *et al.*, 2009
Regeneration	"Germination, birth, or other recruitment of biota including plants, animals, and microbiota, whether arising from colonisation or in situ process" – McDonald *et al.*, 2016	Bulluck and Buehler, 2006, Davis *et al.*, 2003, Fox and Fox, 1978, 1984, Fox and Twigg, 1991, Letnic and Fox, 1997a, Madden and Fox, 1997, Rufaut *et al.*, 2006, Taylor

		and Fox, 2001, Twigg and Fox, 1991
Reclamation	[1]*No SER Definition Provided* “To re-establish some sort of vegetation cover on a degraded land service” – Bradshaw, 1983	Cusser and Goodell, 2013, Greenslade and Majer, 1980, Holl, 1995,1996, Kielhorn *et al*., 1999, Pižl, 2001, Sieg *et al*., 1987, Topp *et al*., 2010
Revegetation	“Establishment by any means, of plants on sites (including terrestrial, freshwater, and marine areas) that may or may not involve local or native species” – McDonald *et al.*, 2016	Craig *et al.*, 2010, Doherty *et al.*, 2016, Galan 1997, Lannoo *et al.*, 2009
Recultivation	*No SER Definition Provided* “Replantation of an area of land with tree or shrub species” – adapted from Hüttl *et al.*, 1996	Dunger *et al.*, 2004, Purger *et al.*, 2004a, b, 2007, Seifert and Prosche 2017, Tajovský, 2001, Topp *et al.*, 2001
Afforestation	*No SER Definition Provided* “Conversion of large areas to forests” – Nosetto *et al.*, 2005	Dunger *et al.*, 2001, Hamilton and Vimmerstedt, 1981, Hendrychová *et al.*, 2012
Unclear		Brändle *et al.*, 2000
Multiple terminologies		Becker *et al.*, 2013, Brandle *et al.*, 2000, Bulluck and Buehler 2006, Craig *et al.*, 2010, Davis *et al.*, 2013, Dominguez-Haydar and Armbrecht, 2011, Dunger *et al.*, 2004, Hendrychová *et al.*, 2012, Hill and Phinn, 1993, Jackson and Fox, 1996, Letnic and Fox, 1997a, Lythe *et al.*, 2017, Majer and Nichols, 1998, Majer *et al.*, 2013, Nichols and Burrows, 1985, Ottonetti *et al.*, 2006, Petersen *et al.*, 2016, Pižl, 2001, Ribas *et al.*, 2012, Rafaut *et al.*,. 2006, 2015, Seifert and Prosche, 2017, Tajovský, 2001, Topp *et al.*, 2001, 2010, Wassenaar *et al.*, 2005, Wheater and Cullen, 1997, Zeppelini *et al.*, 2009

[1] SER: Society for Ecological Restoration, International Standards for the Practice of Ecological Restoration. McDonald *et al.*, 2016.



Search Terms

‘Australasia’, ‘Africa’, ‘North America’, ‘South America’, ‘Asia’, or ‘Europe’, AND/OR ‘animal,’ ‘fauna,’ ‘bird,’ ‘reptile,’ ‘mammal,’ ‘vertebrate,’ or ‘invertebrate’ AND ‘response’, or ‘behaviour’ AND ‘mine,’ or ‘mining,’ AND ‘restoration, ‘rehabilitation,’ ‘reclamation’, ‘recultivation,’ ‘afforestation,’ or ‘regeneration.’

Records Identified

Google Scholar (mean no. of publications from each search combination): n= 19 580

Web of Science: n= 140

Scopus: n= 96

Review first 100 citations from each search

Records assessed for eligibility and duplications removed

Records excluded

n= 19 715

Records included

n= 101

Fig. A2.1: PRISMA 2009 flow diagram

Moher, D., Liberati, A., Tetzlaff, J. and Altman, D.G. (2009). Preferred reporting items for systematic reviews and meta-analyses: the PRISMA statement. *Annals of Internal Medicine* **151**, 264-269.

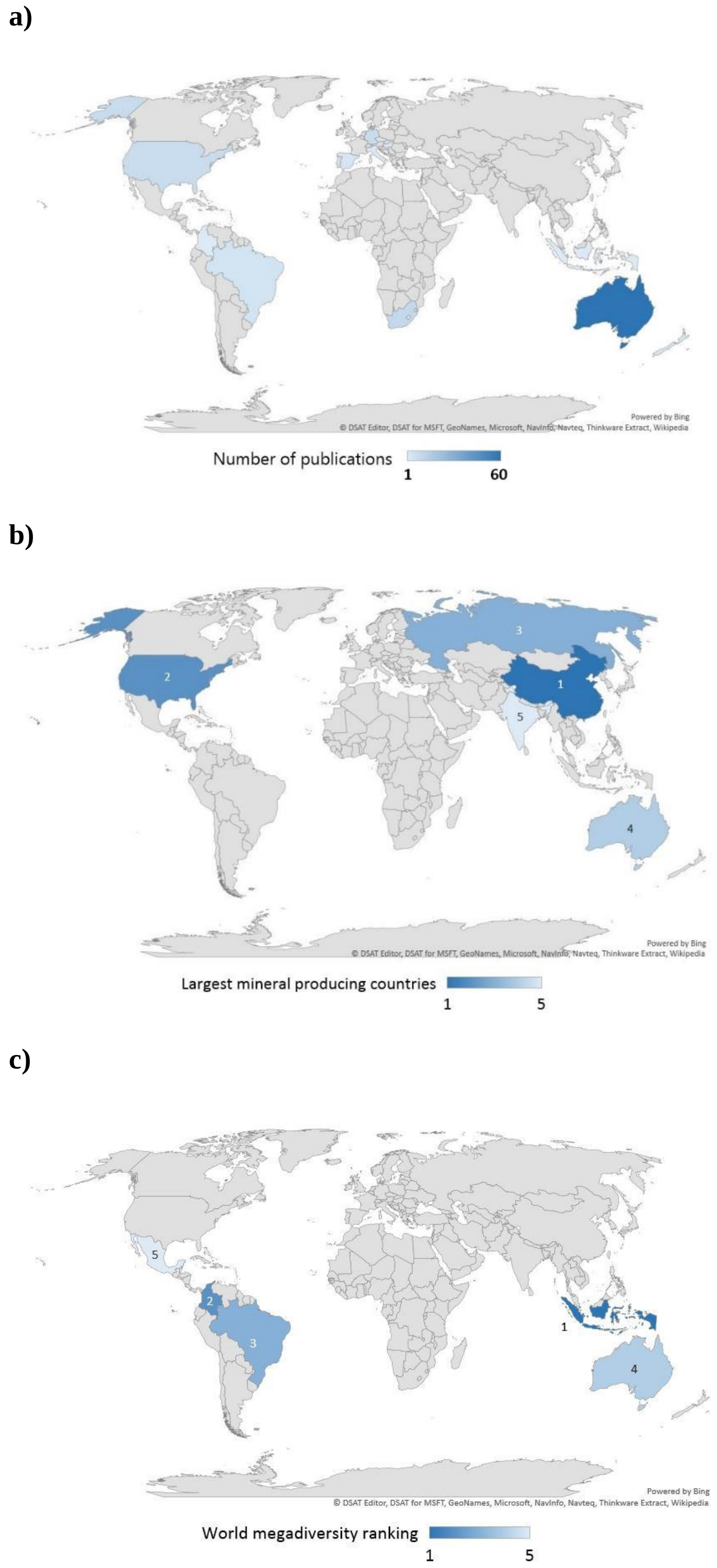


Fig. A2.2: Countries ranked by a) publication output of fauna responses to mine site restoration studies, b) mineral production output (metric tons; Reichl *et al.*, 2014), and c) megadiversity ranking (Médail and Quézel, 1999).

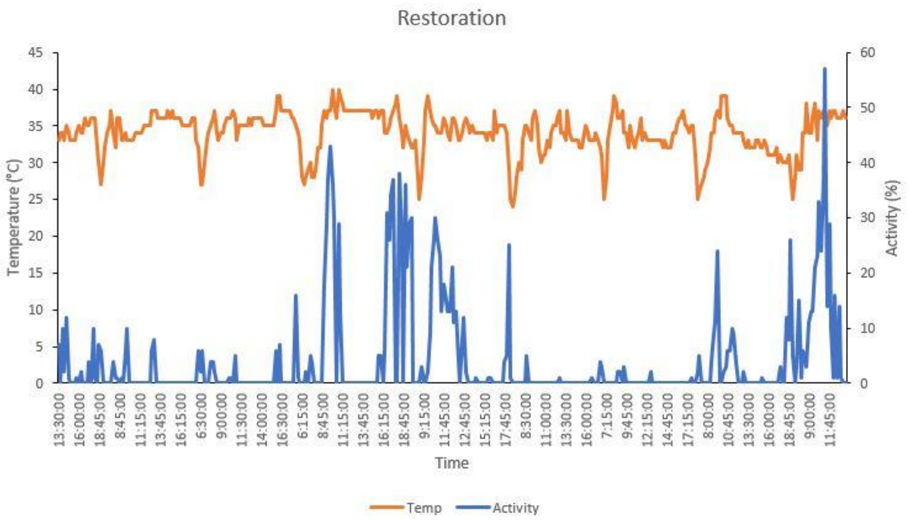
Appendix 3. Build it and some may come: restoration of discontinued mine sites initially favours herbivores

Table A3.1: The foraging guild occupied by each species recorded on camera traps.

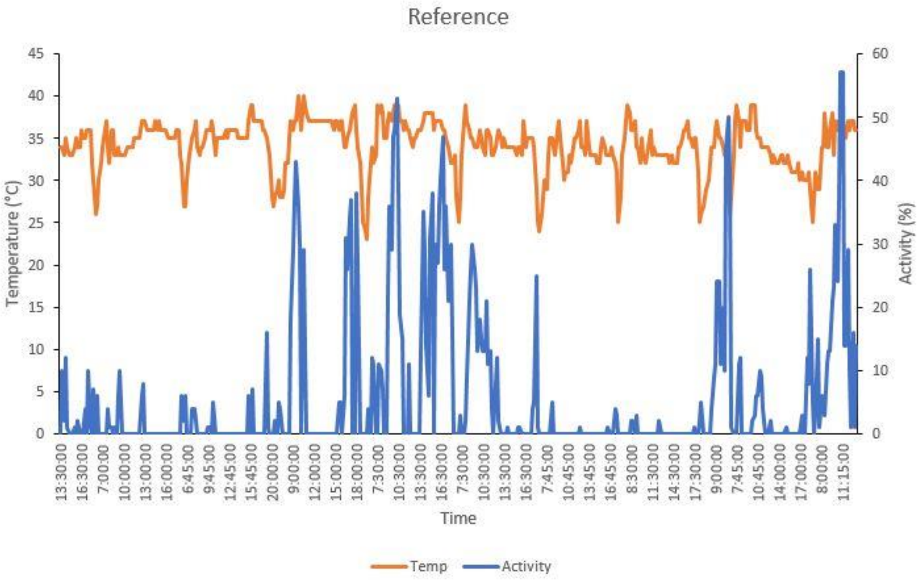
Species	Foraging guild
Aves	
Calyptorhynchus banksii (Red-tailed black cockatoo)	Granivore
Cinclosoma castanotum (Chestnut quail-thrush)	Insectivore
Corvus spp. (Crows and ravens)	Omnivore
Dromaius novaehollandiae (emu)	Herbivore
Epthianura albifrons (White fronted chat)	Insectivore
Leipoa ocellata (Malleefowl)	Granivore
Megalurus mathewsi (Rufous songlark)	Insectivore
Oreoica gutturalis (Crested bellbird)	Insectivore
Pomatostomus superciliosus (White-browed babbler)	Insectivore
Phaps chalcoptera (Common bronzewing)	Granivore
Psephotus varius (Mulga parrot)	Granivore
Strepera versicolor (Grey currawong)	Omnivore
Mammalia	
Felis catus (Feral cat)	Carnivore
Macropus spp. (Kangaroo)	Herbivore
Mus sp.	Omnivore
Oryctolagus cuniculus (European rabbit)	Herbivore
Tachyglossus aculeatus (short-beaked echidna)	Insectivore
Reptilia	
Ctenophorus scutulatus (Lozenge marked dragon)	Insectivore
Varanus giganteus (Perentie)	Carnivore
Varanus panoptes (Yellow-spotted monitor)	Carnivore

Appendix 4. The Time Local Convex Hull (T-LoCoH) method as a tool for assessing responses of fauna to habitat restoration: a case study using the perentie (*Varanus giganteus*: Reptilia: Varanidae)

a)



b)



c)

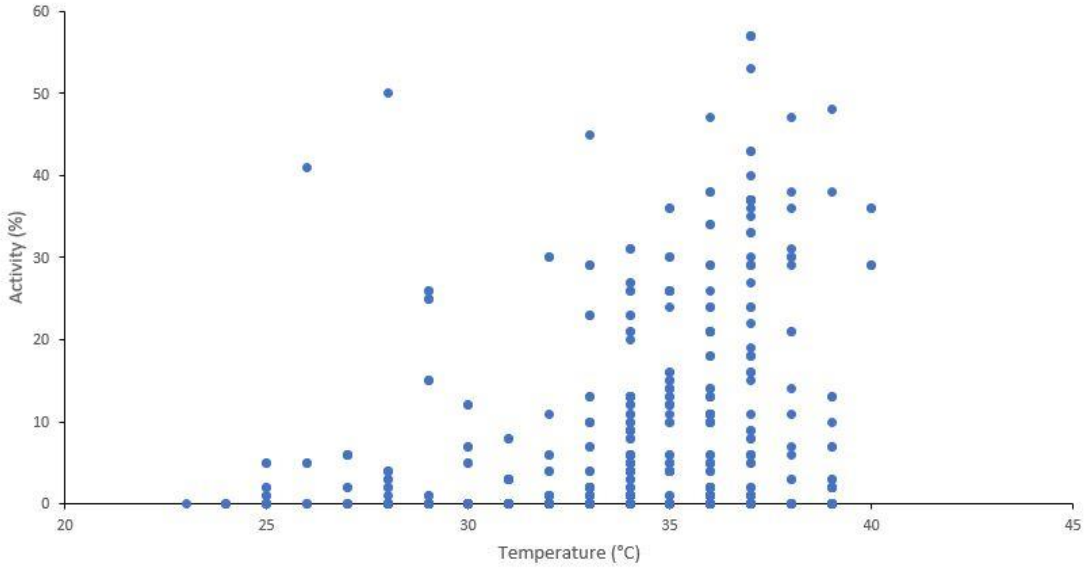


Fig. A4.1: The patterns for activity and temperature of a young adult V. giganteus in a) restoration vegetation, and b) reference vegetation. Activity levels increase with increasing temperatures c), with activity tending to be highest around 35°C, as is reported across the literature.

Appendix 5. Publications arising from this research

Appendix 5.1 Peer-reviewed papers

Cross, Sophie L., Craig, M.D., Tomlinson, S., Dixon, K.W. and Bateman, P.W. Using monitors to monitor ecological restoration: Presence may not indicate persistence. *Austral Ecology*. ***In Press***.

Cross, Sophie L., Tomlinson, S., Craig, M.D. and Bateman, P.W. (2020). The Time Local Convex Hull method as a tool for assessing responses of fauna to habitat restoration: a case study using the perentie (*Varanus giganteus*: Reptilia: Varanidae). *Australian Journal of Zoology*. Online Early. doi: 10.1071/ZO19040.

Cross, Sophie L., Craig, M.D., Tomlinson, S. and Bateman, P.W. (2019). I don't like crickets, I love them. Invertebrates are an important prey source for varanid lizards. *Journal of Zoology*. Online Early. doi: 10.1111/jzo.12750

Cross, Sophie L., Tomlinson, S., Craig, M.D., Dixon, K.W. and Bateman P.W. (2019). Overlooked and undervalued: the neglected role of fauna and a global bias in ecological restoration assessments. *Pacific Conservation Biology* **25**: 331-341.

Appendix 5.2 Peer-reviewed papers not included in the thesis*

*Abstract appended to the thesis.

Cross, Sophie L., Bateman, P.W. and Cross, A.T. (2020). Restoration goals: Why are fauna still overlooked in the process of recovering functioning ecosystems and what can be done about it? *Ecological Management & Restoration* **21**, 4-8.

Appendix 5.3 Conference presentations

Cross, Sophie L., Tomlinson, S., Craig M.D. and Bateman, P.W. (2020). Living La Vida T-LoCoH: Using the T-LoCoH method to predict reptile responses to habitat change. 9th World Congress of Herpetology, Dunedin, New Zealand.

Cross, Sophie L., Tomlinson, S., Craig, M.D. and Bateman, P.W. (2020). Using monitors to monitor restoration: how does Australia's largest lizard respond to mine site restoration? 9th World Congress of Herpetology, Dunedin, New Zealand.

Cross, Sophie L., Tomlinson, S., Craig, M.D. and Bateman, P.W. (2019). Return of a giant: responses of a large predatory reptile (*Varanus giganteus*) to mine site restoration. MLS HDR Science Symposium 2019. Curtin University.

Cross, Sophie L., Tomlinson, S., Craig, M.D. and Bateman, P.W. (2019). Living La Vida T-LoCoH: Using the T-LoCoH method to predict reptile responses to habitat change. MLS HDR Science Symposium 2019. Curtin University.

Cross, Sophie L., Tomlinson, S., Craig, M.D. and Bateman, P.W. (2019). Using monitors to monitor restoration: how does Australia's largest lizard species respond to mine site restoration? 8th World Conference on Ecological Restoration, Society for Ecological Restoration, Cape Town, South Africa.

Cross, Sophie L., Tomlinson, S., Craig, M.D., Dixon, K.W. and Bateman, P.W. (2019). Using monitors to monitor restoration: presence may not indicate persistence. Research Rumble, Curtin University, Perth, Australia.

Cross, Sophie L., Tomlinson, S., Craig, M.D., Dixon, K.W. and Bateman, P.W. (2018). Using monitors to monitor restoration: presence may not indicate persistence. Joint meeting of the Australian Society of Herpetologists and the Society for Research on Amphibians and Reptiles in New Zealand, Redland Bay, Queensland, Australia.

Cross, Sophie L., Tomlinson, S., Craig, M.D., Dixon, K.W. and Bateman, P.W. (2018). How does Australia's largest lizard respond to mine site restoration? Joint meeting of the Australian Society of Herpetologists and the Society for Research on Amphibians and Reptiles in New Zealand, Redland Bay, Queensland, Australia.

Cross, Sophie L., Tomlinson, S., Craig, M.D., Dixon, K.W. and Bateman, P.W. (2018). How do animals respond to mine site restoration? Society for Ecological Restoration Australasia, Brisbane, Australia.

Cross, Sophie L., Tomlinson, S., Craig, M.D., Dixon, K.W. and Bateman, P.W. (2018). How do animals respond to mine site restoration? Royal Society of Western Australia Symposium, Perth, Australia.

Cross, Sophie L., Tomlinson, S., Craig, M.D., Dixon, K.W. and Bateman, P.W. (2017). Behavioural responses of varanids to mine site restoration. West Australian Herpetology Society, Perth, Australia.

Appendix 5.4 Restoration goals: Why are fauna still overlooked in the process of recovering functioning ecosystems and what can be done about it?

COMMENT PIECE

doi: 10.1111/emr.12393

Restoration goals: Why are fauna still overlooked in the process of recovering functioning ecosystems and what can be done about it?

By Sophie L. Cross, Philip W. Bateman and Adam T. Cross

Sophie L. Cross is a PhD student, Philip W. Bateman is an associate professor, and Adam T. Cross is a post-doctoral research fellow with the ARC Centre for Mine Site Restoration, School of Molecular and Life Sciences, Curtin University (Kent Street, Bentley WA 6102, Australia; Tel: + 61 8 9266 7265; Email: sophie.cross@postgrad.curtin.edu.au). *The project arose from knowledge gaps identified through a literature review of fauna responses to mine site restoration, and an interest in stimulating further discussion on this topic at a time where such discussion will contribute to the international debate of what constitutes effective restoration..*

Summary Despite the evidence that fauna play complex and critical roles in ecosystems (e.g. pollination and nutrient cycling) and the knowledge that they need to be considered in restoration, fauna often remain poorly represented in restoration goal setting, monitoring and assessments of restoration success. Fauna clearly are integral to the aspirations of achieving full ecosystem recovery. However, over-reaching assumptions about the unassisted return of fauna to restored sites, low investment in fauna monitoring, and minimal consideration of the requirements for fauna monitoring in regulatory guidance and standards appear to have led to the historically vegetation-centric approaches to rehabilitation and ecological restoration. We argue that ecological complexities render assumptions of unassisted fauna return inappropriate in many situations and may represent a missed opportunity to enhance ecological outcomes and improve restoration trajectories. We advocate for greater consideration of fauna as facilitators of ecological restoration and, particularly for well-funded projects, for monitoring to place greater emphasis on examining the behaviour and resilience of restored fauna communities. There is a clear need for both industry and regulators to recognise that fauna can be crucial facilitators of restoration and appreciate that the return and monitoring of functional faunal communities can be costly, challenging and may require detailed study across a wide range of taxonomic groups. Failure to advance from business as usual models may risk leaving a legacy of ostensibly functional, but biodiversity-depauperate, restored ecosystems.

Key words: *ecological restoration, ecological monitoring, ecosystem engineers, ecosystem functioning, restoration policy, rehabilitation.*

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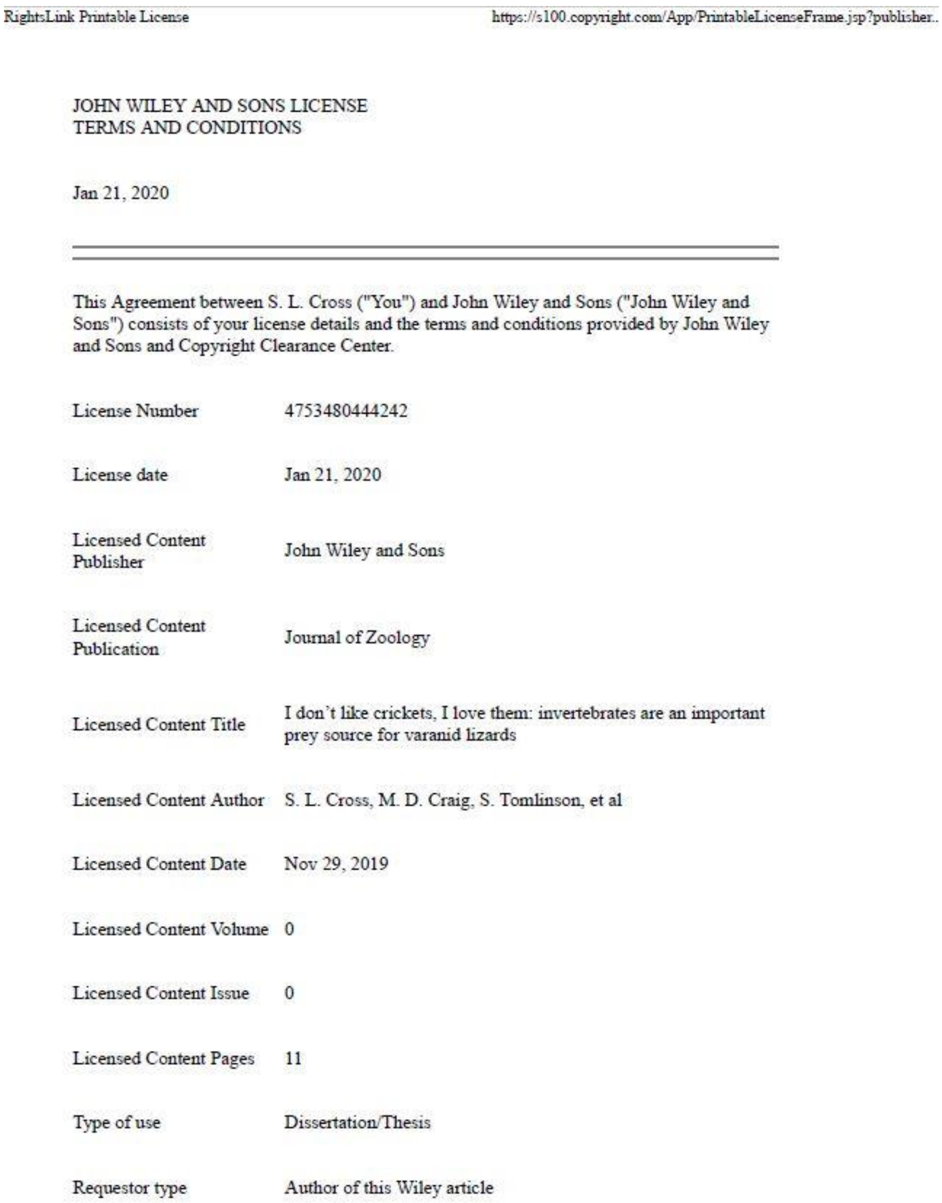

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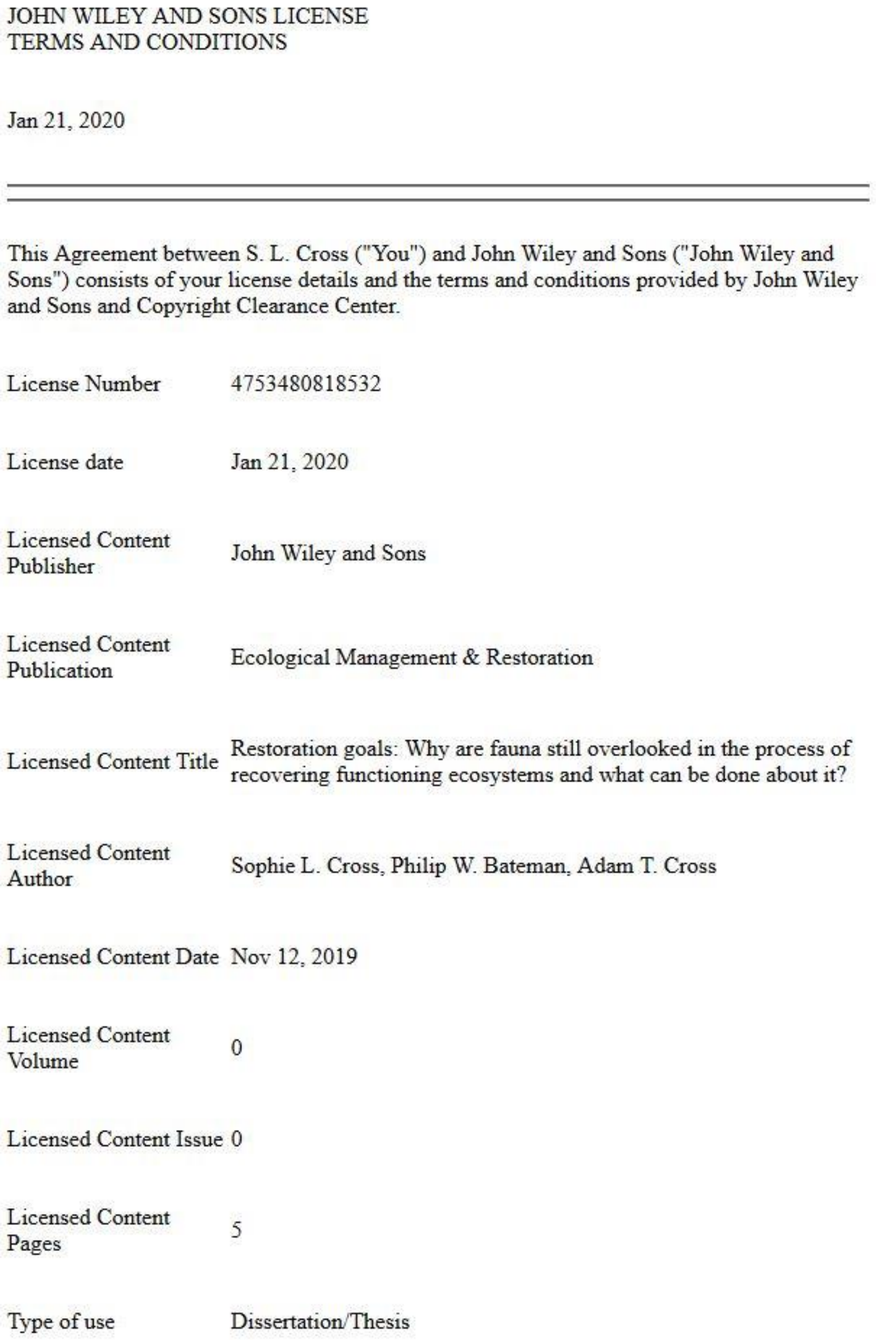

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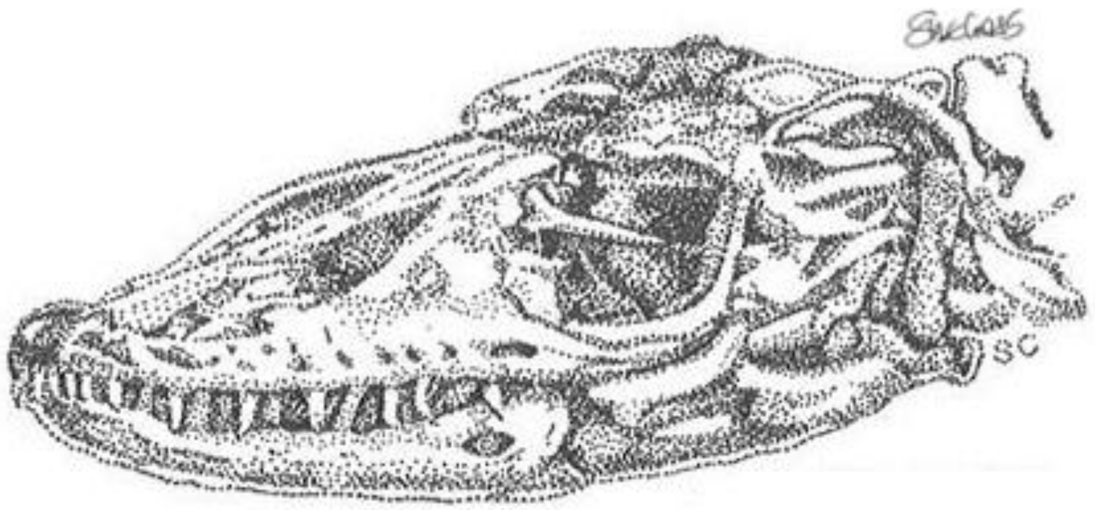

Varanus giganteus

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