Translocation site selection for the nationally endangered grasshopper species *Brachaspis robustus*

A thesis
submitted in partial fulfilment
of the requirements for the Degree of
Master of Science

at
Lincoln University
By
Rebecca Clements

Lincoln University
2021

Abstract of a thesis submitted in partial fulfilment of the
requirements for the Degree of Master of Science.

Abstract

Translocation site selection for the nationally endangered grasshopper species *Brachaspis robustus*

by

Rebecca Clements

The robust grasshopper (*Brachaspis robustus*) is one of the most endangered invertebrates in New Zealand. This species is at risk of extinction due to the compounding pressures of mammalian predation, habitat modification, and climate change. Protection of this species and its habitat is vital to its survival. One tool that has been identified to help manage *B. robustus* is translocation to suitable environments. In my research, I aimed to understand factors critical to developing an effective conservation translocation plan for *B. robustus*. Five potential translocation sites were selected closer to the Southern Alps than existing *B. robustus* populations, as it is thought this may be necessary to protect against future temperature increases that could affect successful egg diapause, a necessary part of the grasshoppers' development. These sites were evaluated for key threats to the species and how to mitigate them, as well as habitat suitability.

A capture-mark-recapture study was done to determine the area individual grasshoppers occupy to inform us on how much space is required at a translocation site to support a *B. robustus* population. Results showed that over 17 days *B. robustus* were usually found within 16 m of the original capture location, however, one individual was found almost 50 m away from the original capture spot after one day.

Microhabitat preference as investigated by assessing ground cover within 1 m of individual grasshoppers at two locations (Snowy River and Ōhau River) and comparing that to ground cover in the wider environment. Ground cover surveys were then replicated at the potential translocation sites. *Brachaspis robustus* were most commonly found on small rocks (1 – 10 cm diameter) with very minimal vegetation cover. Only 3/143 (2.1%) grasshoppers were found within 1 m of vegetation over 2 cm tall with no difference between male and females. In contrast, 46% of quadrats surveyed in the wider habitat contained vegetation with a mean top height of >2 cm, including 31.1% >20 cm.

Presence of native and introduced predators and level of predator control at currently occupied sites compared to potential translocation sites was investigated. Native and introduced predators were found to be present at all sites, but mammalian predator control is currently being undertaken at Cass and Tasman Rivers. The risk of hybridisation was evaluated by assessing maps for suitable *B. nivalis* habitat. Searches were undertaken to determine *B. nivalis* presence and access to potential *B. robustus* translocation sites. The risk of hybridisation is high on the Tasman River due to proximity of *B. nivalis*.

The collated results showed that the most suitable translocation site for this species is the Tasman River. This site is most similar in rock size composition to Snowy River and is large enough for translocation. It has the lowest cover of tall vegetation and is the most protected against mammalian predators. This site is closest to the Southern Alps, mitigating against future global warming. However, the risk of hybridisation may be increased there as *B. nivalis* occupy nearby habitats. Further work must be done to understand this risk.

Keywords: Conservation, invertebrate, translocation, habitat suitability.

Acknowledgements

I would like to extend a huge thank you to my supervisors James Ross, Jon Sullivan and Tara Murray for your valuable guidance and support. I appreciate everything you did to make this research happen. To James, thank you for taking on this project and learning about grasshoppers with me. I would have been completely lost in the world of R if it weren't for you and all your work on the stats for this project. Thank you also to David Norton for your invaluable help on designing the methods for the vegetation analysis and taking time to help me in the field. Most of all, thank you for your encouragement and words of wisdom throughout the years. I am grateful to the Department of Conservation in collaboration with the Environment Canterbury Braided River Research Initiative for providing the funding for this research. Thank you to Simone Cleland from the DOC Twizel office for the logistical support. Thank you to Richard Maloney for your enthusiasm and many great ideas that helped get this project going. Thank you to Jen Schori for all your help with this project and letting me build on the great work you had already done for the robust grasshopper. Shout out to all my friends who provided the most amazing and supportive community. Thank you for all the walks, potlucks and hugs. A special thank you to my office partner, bike buddy, and confidant, Katie Coster, you made coming into the office every day that much more enjoyable. Thank you Luke for sticking by me through this journey and all your very helpful advice. Most of all, thank you to my parents Philip and Leone for your support throughout my university journey, but especially these last couple of years. I simply would not be where I am today without you and I feel so blessed to have you as my parents.

Table of Contents

List of Tables

List of Figures

Chapter 1

Introduction

1.1 General introduction

There is growing consensus that the Earth is entering a sixth mass extinction event (Barnosky et al., 2011). The International Union for Conservation of Nature (IUCN) (2021) estimates that 900 species have gone extinct since 1500. However, this is most likely an understatement, as many species are yet to be assessed for their threat status or are too data deficient (IUCN, 2021). What is often overlooked in such estimates is that most extinctions that have already occurred, and are predicted to happen in the future, are of insects, and most of these extinctions go undocumented (Dunn, 2005). Compared to vertebrates, there is a disproportionately small amount of conservation effort towards insects and other invertebrate species worldwide (Bajomi, Pullin, Stewart, & Takács-Sánta, 2010). Even though, invertebrates are thought to compose well over 80% of all known species and are vital components of ecosystems (Naskrecki, 2013).

New Zealand is known as a biodiversity 'hotspot' due to the high concentration of species-level endemism and the imminent threat of extinction to those species (Myers, Mittermeier, Mittermeier, Da Fonseca, & Kent, 2000). New Zealand was one of the last large landmasses to be colonised by humans, with Polynesians arriving in c. 1280 AD (Wilmshurst, Anderson, Higham, & Worthy, 2008). The devastating consequences for endemic biota brought about by habitat modification and fragmentation since human colonisation, as well as the direct impacts of invasive species, has been well documented in New Zealand (Allen & Lee, 2006). Significant effort has gone into preserving remaining native forests and fauna, especially bird species (Miskelly & Powlesland, 2013). It is estimated that over 80% of invertebrate species in New Zealand are endemic (McGuinness, 2001), yet for most species, there is a considerable lack of ecological knowledge.

The most recent statistics show that 14,255 species (across all taxonomic groups) have been evaluated under the New Zealand Threat Classification System (NZTCS) (NZTCS, 2021). The NZTCS is a processes to assess native taxa for their threat of extinction (Hitchmough, Bull, & Cromarty, 2007). Biota are evaluated by an expert panel and classified into the following umbrella categories: Extinct, Data Deficient, Threatened, At Risk, Not Threatened, Non-resident Native, Introduced and Naturalised, Not assessed and Taxonomically indistinct (Townsend et al., 2008). Of the 14,255 species assessed, 4,961 species are too data deficient to determine their risk of extinction, 1,037 are classified as Threatened, and 3,165 are At Risk (NZTCS, 2021). About 3,838 (29%) of >13,000 known invertebrate taxa have been assessed within the NZTCS (Stringer & Hitchmough, 2012). Around 1,247 are considered to be at risk of extinction in the short to medium term, and 1,208 species are too data deficient to assess accurately (Stringer & Hitchmough, 2012).

New Zealand is unique because it was isolated from any other large landmass for more than 85 million years (Gibbs, 2016). As a result of this isolation, there is high endemism within the biota (Taylor-Smith, Morgan-Richards, & Trewick, 2020). Most of the endemic plant and animal species exhibit characteristics from evolving with avian and reptilian predators and the absence of many mammalian predators (Tennyson, 2010). The only terrestrial mammals endemic to New Zealand are several bat species. The lesser short-tailed bats, *Mystacina tuberculata*, are described as three sub-species (Hill & Daniel, 1985). These species are omnivorous and spend most of their time climbing on trees and the forest floor. One dietary study showed that insects made up about 50% of *M. tuberculata* diet (Arkins, Winnington, Anderson, & Clout, 1999). The long-tailed bat *Chalinolobus tuberculatus* predominantly feeds on flying insects (Dwyer, 1960). The greater short-tailed bat (*Mystacina robusta*) is thought to be extinct as there have been no sightings of the species since 1965 (Hill & Daniel, 1985). It is believed that mystacinid bats and many bird, reptile and invertebrate species, have evolved to fill the ecological role of small terrestrial mammals (Arkins et al., 1999) (Griffin, Trewick, Wehi, & Morgan-Richards, 2011).

With the arrival of Polynesians to New Zealand about 800 years ago came the first introduced predator of invertebrates, the Pacific rat, or Kiore (*Rattus exulans*) (Allen & Lee, 2006) (Wilmshurst et al., 2008). Europeans first arrived in New Zealand in 1769 and brought a suite of mammals and exotic plants equipped to exploit the habitat and naïve biota found here (Clout & Russell, 2006). A further 52 mammal species were introduced when Europeans settled here (King, 2005). After Polynesians settled in New Zealand, fires cleared around half of the lowland montane forests in New Zealand (McGlone, 1989). This directly led to widespread, severe landscape modification and loss of native biota (Perry, Wilmshurst, McGlone, McWethy, & Whitlock, 2012). The arrival of Europeans further increased deforestation as native forests were felled for timber and converted to pasture for animal grazing (McWethy et al., 2010). Terrestrial invertebrates in New Zealand had evolved to avoid avian and reptilian predators, resulting in traits that left them especially vulnerable to introduced mammals that arrived with settlers (Gibbs, 2010). Kiore wiped out many large-bodied terrestrial invertebrate species before Europeans arrived (Ramsay, 1978), and house mice and rats further impacted the terrestrial invertebrate fauna (King, 2005).

Many New Zealand terrestrial invertebrates exhibit gigantism, crypsis, flight loss, ground-dwelling behaviour, odour release and specific predator escape reactions that are not effective against introduced mammalian predators that rely more on their sense of smell (Gibbs, 2010). Invertebrates that often spend time stationery on exposed surfaces frequently have a cryptic appearance regarding their colour, shape, and decoration (Thery & Gomez, 2010). Crypsis is a beneficial trait for insects in an avian dominated landscape as birds rely on their excellent vision to locate prey (Walton & Stevens, 2018). Many grasshopper (Orthoptera: Acrididae) species are cryptic in appearance, matching the background colouration of their microhabitat (Eterovick, Figueira, & Vasconcellos-Neto, 1997). Examples of cryptic colouring in New Zealand invertebrates are seen in the underwings of the red admiral (*Vanessa gonerilla*) and yellow admiral (*Vanessa itea)* butterflies (Laidlaw, 1970). The concealer moth (*Hierodoris extensilis*) has cryptic wing colourings against its habitat of mosses and cushion plants (Hoare et al., 2017). Carnivorous land snails (*Powelliphanta*) and giant wētā (Orthoptera: Anostostomatidae) are well known invertebrate examples of gigantism in New Zealand (Daugherty, Gibbs, & Hitchmough, 1993). The giant wētā (*Deinacrida*) is renowned as one of the heaviest insects in the world (McIntyre, 2001).

Most birds, except for kiwi (*Apteryx* spp.), hunt very differently from mammals. They usually rely on highly developed visual skills rather than olfactory senses that mammals predominantly use (Martin et al., 2007). Many New Zealand endemic bird and invertebrate species release strong odours (Worthy & Holdaway, 2002). Pheromone communication is used for mating and communication purposes in bird and invertebrate species (Field & Jarman, 2001). Tree wētā and giant wētā species are known to have especially pungent smells, detectable to the human nose. These strong pheromones would not have posed a risk in a predominantly avian ecosystem (Hughes, Price, & Banks, 2010), and kiwi feed mostly on soft-bodied invertebrates found in the soil such as earth worms (Sales, 2005).

The extinction or decline of large and flightless invertebrates in New Zealand after the introduction of mammalian predators may not be entirely due to the physical characteristics described above but also to behavioural attributes (Gibbs, 1998). Loss of flight makes species vulnerable to habitat fragmentation and modification, as well as predation, as species cannot disperse or recruit quickly (Norbury et al., 2013). In addition to flight loss, many New Zealand invertebrates also often exhibit a freeze-response to predators (Gibbs, 2010). The key to avoiding rodent and hedgehog predation is early detection and rapid escape (Anson & Dickman, 2013). A study completed in Fiordland, New Zealand, showed that invertebrate response to predators was different between individuals who had always been in predator-free environments and those who lived with mammalian predators (Bremner, Barratt, Butcher, & Patterson, 1989). This observation demonstrates that invertebrates may change their behaviour as a tactic of predator avoidance. They may synchronise their activity to the sleeping cycles of their predators to enable them to be active whilst minimising their risk of predation (Bremner et al., 1989).

One New Zealand endemic invertebrate species that exhibits all the above characteristics (crypsis, pungent odour, large body size, loss of flight, freeze-response), is the Robust Grasshopper, *Brachaspis robustus*.

1.2 Study species

1.2.1 Distribution and ecology

Brachaspis robustus Bigelow, 1967 (Orthoptera: Acrididae) is a Nationally Endangered[1] grasshopper endemic to the Mackenzie Basin (~7339 km^2) in the centre of the South Island, New Zealand (White, 1994), and is considered to be one of the two most endangered Orthopteran species in New Zealand (Stringer & Hitchmough, 2012). *Brachaspis robustus* is a large grasshopper (Figure 1.1), with the females reaching up to 38 mm in length and males up to 17 mm (Schori, Steeves, & Murray, 2020). They are flightless, and the residual wings are very small, less than 2 mm long and almost as broad as long (Bigelow, 1967). *Brachaspis robustus* is a non-stridulating species, meaning it does not sing. It is assumed they communicate and locate mates through the release of pheromones, as well as using visual signals over shorter distances. Both males and females are cryptic in their appearance; they exhibit colour morphs that reflect the dominant substrate colour in their environment, including greys and browns of braided river gravels and silts (Figure 1.1) (White, 1994).

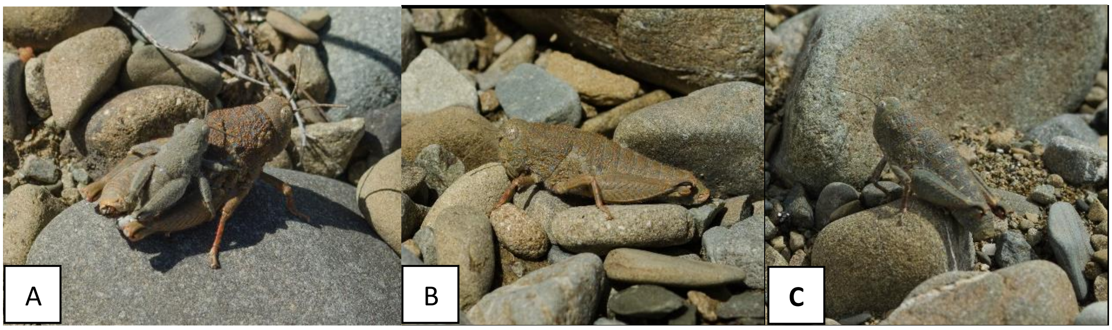


Figure 1.1. (a) Mating *B. robustus* pair. (b) & (c) the two main colour morphs seen in *B. robustus*. Photos: Rebecca Clements (2020).

Brachaspis robustus are considered generalist herbivores, feeding on a combination of lichens, mosses, and herbaceous plant species (White, 1994). They are found in rocky braided riverbeds and river outwash plains within the lower Mackenzie Basin (White, 1994). Braided rivers are highly dynamic landscapes characterised by many alluvial channels that vary in size and water flow. High

---

[1] Nationally Endangered is the second highest threat ranking within the NZTCS, behind Nationally Critical.

levels of disturbance caused by frequent flooding events and constantly changing channels, bars, and islands are associated with braided rivers (Caruso, Edmondson, & Pithie, 2013).

The current distribution of *B. robustus* is considerably smaller than historical records suggest (Trewick, 2001). This species are currently found in five different locations throughout the Mackenzie Basin; lower Ōhau River, Snowy River, a small area of Pūkaki River, most of Tekapō River, Fork stream gravel heaps and Patersons Terrace (Figure 1.2) (Schori, 2020). The holotype of this species was taken from near the Ahuriri River, at the Southern limits of the Mackenzie Basin. The two paratypes have no further information, such as collection date or collector, other than "Kurow". Kurow is a town located near the Waitaki River that drains from the Mackenzie Basin (Bigelow, 1967). Heavy habitat modification by creating Lake Benmore, planting of willows and rampant spread of lupins has resulted in no suitable habitat remaining along the Ahuriri River, nor in the Waitaki River by Kurow (White, 1994) and there have been no further sightings of *B. robustus* in these locations.

Brachaspis robustus is known to be patchy within its range (White, 1994), but the reasoning for this is not well understood. Current knowledge of the distribution of this species is based on observations made during Project River Recovery bird surveys >25 years ago (Maloney, 1992) and sporadic surveys at a few of the known sites, and *ad hoc* observations since. Searches for *B. robustus* in the landscape are time-consuming. It involves the observer slowly walking through all habitats of interest to visually detect grasshoppers moving or jumping in response to the observers' presence (Schori, Steeves, & Murray, 2020a). As *B. robustus* are cryptic both visually and acoustically, obtaining detailed information about the species is difficult. Poor detectability of this species may result in false absences (Schori, 2020). Therefore, the entire distribution is most likely not known for this species.

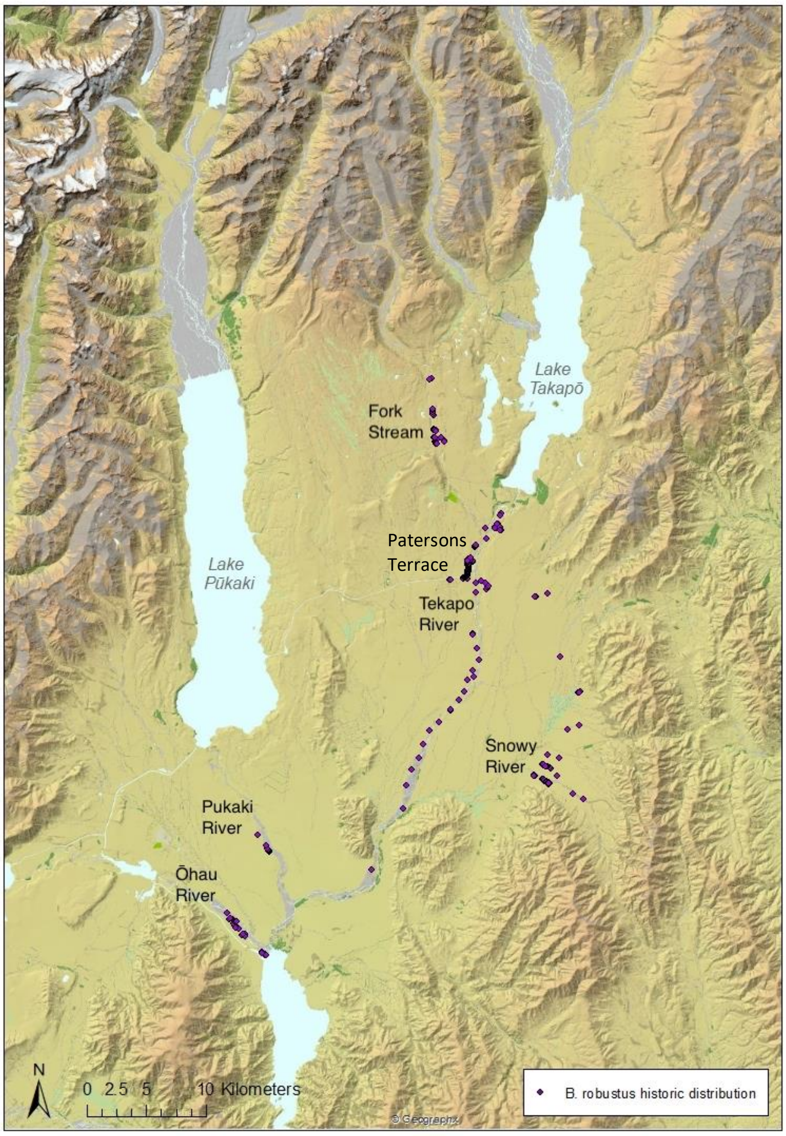


Figure 1.2 Recent distribution of the robust grasshopper. Purple dots represent all observations of *B. robustus* since ~1995. Present day distribution of *B. robustus* is much patchier than shown in this figure. The Ahuriri River and Kurow are not shown on this map but are found southwest of Ōhau River.

Recent progress has been made in understanding the life cycle of *B. robustus* (Schori, 2020). *Brachaspis robustus* can survive winter at any life stage (White, 1994) and are thought to have a lifecycle spanning over two years (Schori, 2020). The eggs of *B. robustus* undergo obligate diapause as part of their development. It is assumed they require a period of below 0°C to break the diapause and hatch, most likely to ensure eggs don't hatch during the harsh winter of Mackenzie Basin (Schori, 2020). This is a characteristic seen in other New Zealand Acrididae (Batcheler, 1967; Mason, 1971; Northcroft, 1967) most of which are considered cold-adapted species (Bigelow, 1967). Cold-adapted grasshoppers such as *B. robustus* can tolerate freezing at all life stages, resulting in relatively long life-spans and overlapping generations (Koot, Morgan-Richards, & Trewick, 2020). Endemic insects found in alpine and sub-alpine areas in New Zealand are thought to be adapted to tolerate freezing temperatures due to the reduction of alpine habitat during the Pleistocene, and New Zealand's erratic climate where snow may occur in alpine areas any time of the year (Koot et al., 2020).

1.2.2 Threats and protection

Although ecological knowledge of this species is limited, it is understood that the current populations are in serious decline (Trewick, Morris, Johns, Hitchmough, & Stringer, 2012). The biggest threat to the survival of *B. robustus* is thought to be predation (White, 1994). Cryptic colouring and the tendency to freeze or seek refuge under rocks is an effective defence against native birds (Schori, 2020), however, these characteristics, along with the grasshoppers pungent odour, leave them extremely vulnerable to introduced mammal predators such as feral cats (*Felis catus*), mustelids (stoats [*Mustela erminea*], ferrets [*M. furo*] and weasels [*M. nivalis vulgaris*]), hedgehogs (*Erinaceus europaeus*) rats (*Rattus rattus*, *R. norvegicus*) and mice (*Mus musculus*) that hunt using both visual and olfactory senses (Schori, Maloney, Steeves, & Murray, 2019). A major issue facing the conservation of *B. robustus* is the lack of predator control currently occurring within its natural range. There are extensive predator control measures being taken by Te Manahuna Aoraki in other areas of the Mackenzie Basin that would immensely benefit this species (Figure 1.3). The Te Manahuna Aoraki project is focused on restoring the natural landscapes and threatened species pf the upper Mackenzie Basin.

Due to the compounded risks towards this species, translocation is being considered as a conservation management strategy (Schori, 2020). There remains a knowledge gap surrounding exactly what factors *B. robustus* require in a habitat. Sporadic monitoring of the species, led by the Department of Conservation, has been occurring since 1992. Although inconsistent methods have been used, all observations indicate a decline in population numbers and increasing patchy distribution. The population that was recorded near the Ōhau River delta (Fraser, 1999) can no longer be found there, nor can the population that was once found below the Tekapō dam (T. Murray, personal communication, December 14, 2021). There has been limited management action in the past, but recent work towards understanding the ecological requirements of *B. robustus* may allow for successful translocation of the species in the future.

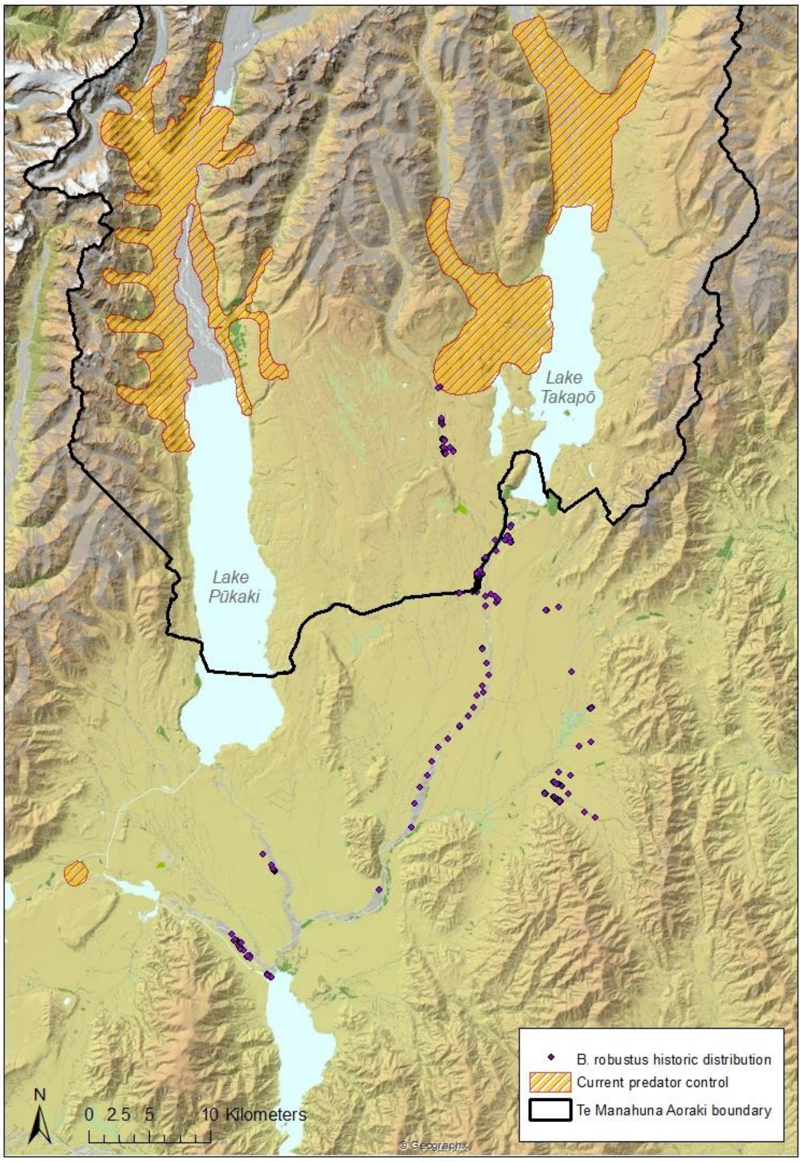


Figure 1.3 Areas where mammalian predator control is being carried out by Te Manahuna Aoraki, and historical *B. robustus* distribution.

1.3 Study Aims

Considering the declining trend and recognition that translocation should be explored as one of several conservation management strategies to protect the robust grasshopper, my research aims were:

1) Identify all major threats to *B. robustus* that need to be considered in developing an effective translocation plan (Chapter 2)

2) Use monitoring methods to determine the size of an area used by individual *B. robustus* to inform us of how much space a population would require at a translocation site, and to gather information on population demographics (Chapter 3).

3) Determine what ground cover composition is preferred by *B. robustus* (Chapter 4).

4) Assess if, or the degree to which, some of the threats to *B. robustus* survival can be eliminated or mitigated in a selection of proposed translocation sites (Chapter 5).

5) Use the knowledge obtained from reviewing relevant literature and data collected during this study to recommend the suitability of a selection of potential translocation sites within the Mackenzie Basin to better manage and secure this endangered grasshopper species (Chapter 5 & 6).

Chapter 2

Translocation as a conservation tool

2.1 Introduction

The objectives of this chapter were to review the relevant literature to identify key factors that need to be considered in developing an effective translocation protocol for *B. robustus*. Some of these key factors are assessed in the subsequent chapters.

2.2 Invertebrate translocations in New Zealand

Invertebrates make up the majority of all biodiversity, yet they are rarely the focus of conservation science or management (Seddon, Soorae, & Launay, 2005). Healthy ecosystems rely on the processes invertebrates, fungi and plants carry out, but still, these organisms are frequently overlooked in conservation strategies (Hafernik, 1992). Using extinction rates for well-known taxa, such as birds, it is estimated that c. 44,000 invertebrate species have gone extinct in the last 600 years (Dunn, 2005). As scientists tackle the global biodiversity crisis, translocation is viewed as a valuable conservation tool for some endangered species (Norbury, Van Den Munckhof, et al., 2014). Due to habitat modification and loss, paired with other anthropogenic pressures, species often cannot naturally restore their population by dispersal or recruitment (Seddon, 2010). Translocation, sometimes called reintroduction, is a human-facilitated process through which wild or captive-bred individuals of endangered species are moved from a habitat where they are vulnerable to extinction to a place where threats are minimised or eradicated (IUCN, 2013). It is important to acknowledge that translocation is not the best tool for all endangered species. Many factors need to be carefully considered to ensure any translocation attempt is appropriate and done well to ensure a successful conservation outcome (Bubac, Johnson, Fox, & Cullingham, 2019). The IUCN first published the 1998 guidelines for re-introductions (IUCN, 1998), which has been replaced by the updated 2013 version (IUCN, 2013). These guidelines are respected and followed globally by conservation managers conducting translocations (Armstrong, Hayward, Moro, & Seddon, 2015).

There are different types of translocation, all within the scientific field of reintroduction biology (Seddon, 2010). They are; conservation translocation, reinforcement, reintroduction, assisted migration and ecological replacement (Morris, Brook, Moseby, & Johnson, 2021). The majority of translocations in New Zealand have been conservation translocation of species to off-shore islands (Cromarty & Alderson, 2013). For the purpose of this study, I will focus on conservation translocation, hereafter translocation, as this is the type of translocation that is being considered for *B. robustus*. Only in the last ten years (Seddon, Moro, Mitchell, Chauvenet, & Mawson, 2015) has translocation been acknowledged in scientific literature as a viable option for species that must be moved *outside* of their natural range, often due to the imminent threat of climate change (Hoegh-Guldberg et al., 2008).

The topic of translocation sparks warranted controversy due to the potential ecological risks it poses to the receiving habitat, as well as to the translocated species itself (Ricciardi & Simberloff, 2009). It is important to establish the requirements of a species and the associated risks that come with translocating it. Threats towards a species need to be removed in order for a translocation to be successful. These factors underpin the research of this thesis. Most literature opposing the use of translocation deals with mammals (Bajomi, Pullin, Stewart, & Takács-Sánta, 2010), and animal behavioural issues are the most important reason translocations fail or are opposed (Berger-Tal, Blumstein, & Swaisgood, 2020). In a review of 293 animal translocation case studies, invertebrates and amphibians had the least reported difficulties and minimal animal behaviour issues (Berger-Tal et al., 2020). A global animal translocation literature review showed almost 50% of translocations were mammals, 27% of birds, and only 3% were invertebrate translocations (Bajomi et al., 2010). Criticism towards animal translocations often comes from the very low success rate of translocations. Only 52% of terrestrial invertebrate translocations globally have been successful (Bellis, Bourke, Williams, & Dalrymple, 2019), and (Sherley, Stringer, & Parrish, 2010) reported that 21.2% of invertebrate translocation in New Zealand had an unknown outcome.

Translocation has been carried out as an accepted tool for the conservation of endangered species in New Zealand since the 1990s (Lloyd & Powlesland, 1994; Seddon et al., 2015). Between 2002 and 2010, the number of translocation applications submitted to the Department of Conservation (DOC) in New Zealand for approval increased by 60%. Birds were still the most frequently translocated group, making up 74% of all applications submitted in 2002-2010. Reptile translocations contributed 15%, followed by plants (6%) and invertebrates (5%) (Cromarty & Alderson, 2013). Invertebrates can be translocated for different reasons (Bowie, 2001); as they underpin many ecological processes, invertebrates are often translocated for ecological restoration (Keesing & Wratten, 1998), as well as species conservation (Griffith, Scott, Carpenter, & Reed, 1989).

There are many well-documented instances of bird translocations, whereas reptile and invertebrate conservation is deficient in comparison (Bajomi et al., 2010). The first invertebrate translocations in New Zealand were of the Cook Straight wētā (*Deinacrida rugosa*), Mahoenui giant wētā (*D. mahoenui*) and the flax snail (*Placostylus hongii*) (McHalick, 1998). Wētā (Orthoptera families Anostostomatidae and Rhaphidophoridae) remain the most frequently translocated insect group in New Zealand. As of 2010, 71% of invertebrate translocations were of wētā (Sherley et al., 2010); this is partly due to the endangered status of many of the species (McGuinness, 2001). Wētā being large-bodied and iconic New Zealand insects results in more public support for their conservation (Watts & Thornburrow, 2009), a trend more commonly associated with charismatic birds and mammals (Bajomi et al., 2010). Currently, there have been six different wētā species translocated; the Mahoenui giant wētā (*D. mahoenui*), Cook Strait Giant wētā (*D. rugosa*), Middle Island Tusked wētā (*Motuwētā isolata*), Auckland tree wētā (*Hemideina thoracica*), Wellington tree wētā (*H. crassidens*) (Watts, Stringer, Sherley, Gibbs, & Green, 2008) and Banks Peninsula tree wētā (*H. ricta*) (Bowie, 2008).

The Mahoenui giant wētā was translocated to various mainland, and offshore island sites between 1989 and 2002, as the habitat they were occupying in the Waitomo District was small and at risk from fire (Sherley, 1998). The 200 ha area of pasture land and gorse (*Ulex europaeus*) was made a reserve for the conservation of species as it was the only area it was still found (Watts & Thornburrow, 2009). The Mahoenui giant wētā have become reliant on this gorse habitat as it is thought that the dense, prickly form of gorse bushes provides protection from mammalian predators and shelter and food (Sherley & Hayes, 1993). As the wētā now rely on gorse in the presence of introduced mammalian predators, they had to be translocated to habitats containing the plant species. However, gorse in New Zealand is eventually shaded out by native forest species regenerating as it is a light-demanding species. Mahoenui giant wētā on the mainland were released onto gorse habitat that had native forest adjacent in hopes that the wētā would eventually establish in the forest (Sherley, 1998). Habitat suitability of the release site is essential. Immediate dispersal and mortality can be reduced by releasing insects onto a substrate that they are known to be associated with (Richardson, Doerr, Ebrahimi, Lovegrove, & Parker, 2015). A total of 2,050 wētā from this species have been translocated to 7 sites over 32 different release events between 1989 and 2009 (Watts & Thornburrow, 2009). Wētā have been resighted in five of the translocation sites, however, the researchers concluded that only two of the translocated populations could be considered a success, as they were the only populations whose numbers had increased to a stable, self-sustaining level (Watts & Thornburrow, 2009). The translocation efforts made for the Mahoenui giant wētā highlighted the importance of removing the key threats when undertaking a translocation (Watts et al., 2008). The only two sites that Mahoenui giant wētā were thriving at and have successfully established a population were free from rats (Watts & Thornburrow, 2009).

The flax weevil (*Anagotus fairburni*) and knobbled weevil (*Hadramphus stilbocarpae*) were two of the first invertebrate species to be granted absolute legal protection in New Zealand under the Wildlife act 1953 (Miskelly, Tennyson, & Bishop, 2018). Both weevils were once widely distributed but are now restricted to predator-free islands in southern Fiordland in what is estimated to be 10% of their former range(Miskelly et al., 2018). Both species were translocated to Breaksea Island in 1991 after Norway rats were eradicated from the island (Sherley et al., 2010). Flax weevils were found on 56 islands after a recent survey, whereas only one knobbled weevil was found. Researchers note that further surveys, especially at night, would likely reveal other knobbled weevils (Miskelly et al., 2018). The flax weevil was also translocated from Maud Island to Titi Island in 2001 and Mana Island in 2004. Eighty-two individuals were released on Titi Island and 70 on Mana Island (Sherley et al., 2010).

The ecological restoration of Quail Island (Ōtamahua), an 85 ha island located in Banks Peninsula, has involved translocations of the ground beetle *Megadromus guerinii*, Banks Peninsula tree wētā (*Hemideina ricta*) and leaf-vein slugs (*Pseudaneitea maculata*) (Bowie, 2001). Mice and deer are the only remaining mammals on Quail Island as rabbits (*Oryctolagus cuniculus*), common brushtail possums (*Trichosurus vulpecula*), feral cats, mustelids, hedgehogs and rats have been eradicated through the use of toxins and an extensive trap network over the past 20 years (Bowie, Kavermann, & Ross, 2011). Mice have persisted despite two attempts at eradication and remain a threat to the invertebrate fauna there (Bowie et al., 2018). Quail Island is ecologically important as it provides a habitat for many endemic plant and insect species (Bowie, 2008). Habitat destruction from agriculture and farming has deteriorated native fauna assemblages there, but some insect species have been reintroduced through dedicated efforts of volunteers. Banks Peninsula tree wētā and leaf vein slugs were absent on the island during the first surveys (1998-2000), and Carabidae species were rare (Bowie, 2001). Recommendations from early surveys led to these endemic species being translocated onto Quail Island (Bowie, 2008). Wooden discs cut from felled trees were used in the translocation of leaf veined slugs and ground beetles. These provided refuges for invertebrates as there was a lack of large woody debris as a result of historical deforestation. The wooden discs also enabled non-destructive monitoring of the species post-release (Bowie, 2008). Banks Peninsula tree wētā were translocated from Banks Peninsula in artificial wooden cavities, referred to as wētā motels. These are hollow wooden boxes with an entrance hole in the bottom (Bowie, Hodge, Banks, & Vink, 2006). Wētā readily occupy these on their own accord as good quality refuges are usually limited. Wētā motels also allowed for easy, non-destructive monitoring post-release (Bowie, Allen, McCaw, & van Heugten, 2013). The successful of invertebrate translocations to Quail Island show the importance of understanding species ecology. The use of wooden discs and wētā motels undoubtedly made the translocations much more successful than if no refugia had been provided (Bowie, 2008).

Numbers of Middle Island tusked wētā (*Motuweta isolata*) were concerningly low in the early 1990s, with less than 200 individuals recorded (McIntyre, 2001). Numbers continued to decline, and conservation managers translocated captive-reared animals to nearby predator-free islands in an attempt to save the species from extinction (Stringer & Chappell, 2008). According to IUCN (2013) terminology, the release of Middle Island tusked wētā to nearby islands should be classified as relocation as this species were once found on all Mercury Islands, as they were one landmass ~6,500 years ago when sea levels were lower (Towns, 1994). However, evidence of many insect distributions disappeared with the arrival of humans to New Zealand as the landscape was modified to such a severe degree. The authors, therefore, continue to use the term 'translocation' in their article (Stringer & Chappell, 2008). It is thought that the invasion of kiore to the Mercury Islands is the reason why Middle Island tusked wētā were only found on Middle Island, as it remains rat-free (Stringer, Watts, Thornburrow, Chappell, & Price, 2014).

Other invertebrates that have been translocated or relocated for conservation purposes in New Zealand include the speargrass weevil (*Lyperobius huttoni*), turbott's weevil (*Anagotis turbotti*), large darkling beetle (*Mimopeus opaculus*), leaf-vein slug (*Pseudaneitea maculata*), giant centipede (*Cormocephalus rubriceps*), katipo spider (*Latrodectus katipo*), Kauri snail (*Paryphanta busbyi*), and several *Powelliphanta* and flax snail (*Placostylus*) species (Sherley et al., 2010).

2.3 Factors to consider in selecting translocation sites for *B. robustus*

To enable a successful translocation, suitable sites must first be selected. This entails the identification of attributes that the species requires in an environment (IUCN, 2013). IUCN guidelines specify that to implement a translocation effectively, the habitat in the selected site should provide all ecological needs to the species. Biological requirements throughout all life stages of the translocated species must be met at the proposed translocation site. In addition, there should be a high level of confidence that known threats to the survival of the species being translocated are absent or sufficiently mitigated at the translocation site (IUCN, 2013).

2.3.1 The importance of habitat suitability for translocation

Before conducting a translocation, a thorough survey of vegetation and other animal species present in the location where the founder population will be released must be conducted (IUCN, 2013). First and foremost, conservation managers must be certain that the same species being translocated does not already exist at the site. Unless population enhancement is the intention, releasing individuals from a separate stock into another population of the same species will destroy the genetic differences between the two populations if they have been geographically separate for long enough and have become genetically distinct (Seddon, Griffiths, Soorae, & Armstrong, 2014). The risk of corrupting the genetic integrity of the various *B. robustus* populations throughout the Mackenzie basin is not known as we have little genetic information on them (Koot, Morgan-Richards, & Trewick, 2020). However, progress is being made towards assessing the population genetics of the species, but the results are still pending (T. Murray, personal communication, July 14, 2021). Although some invertebrate species are tightly associated with the host plants which they rely on for food, breeding or shelter (Sherley & Hayes, 1993), *B. robustus* is thought to be a generalist herbivore (White, 1994), and feed on a range of mosses, lichens, native and exotic plant species (Schori, 2020). For invertebrates such as *B. robutsus* that are directly affected by mammalian predation, predator control at the release site is crucial (Armstrong et al., 2015). It is also important to understand what other species, such as competitors, are present at the translocation site that could reduce the translocated species' survival rate (IUCN, 2013). Much of the braided riverbed habitat that *B. robustus* is found in has been modified by invasive plant species (Caruso, 2006). This reduces suitable ground available for egg laying of female grasshoppers and basking. It also amplifies the risk to *B. robustus* of mammalian predation as vegetation provides habitat for predatory mammals (White, 1994).

The IUCN (2013) also describes the major risks to the host environment that founder populations are being translocated to. There is a risk of unknown ecological outcomes from introducing a species where it previously was not present, as well as the risk of spreading disease between populations (Ricciardi & Simberloff, 2009). Hybridisation with resident species in the release site and consequential loss genetic diversity is an essential factor to consider, especially when dealing with species that have been historically isolated (IUCN, 2013).

2.3.2 Threats to *Brachaspis robustus*

Predation

Recent work has shown that predation from introduced mammals is one of the biggest threats to *B. robustsus* survival (Schori, Maloney, Steeves, & Murray, 2019). Extensive efforts to eradicate mammalian predators, such as rodents, from off-shore islands have resulted in many islands acting as refuges for endangered invertebrates (St Clair, 2011). However, *B. robustus* habitat is not available on off-shore islands, so they must be managed in-situ or translocated to other mainland areas where predators can be managed.

Although mice and rats are relatively scarce with in the habitat of *B. robustus*, data from gut analysis show that hedgehogs (Jones, Moss, & Sanders, 2005), stoats (McAulay, Seddon, Wilson, & Monks, 2020), and feral cats (Murphy, Keedwell, Brown, & Westbrooke, 2004) present in the Mackenzie basin all prey heavily on insects. No studies have yet identified *B. robustus* specifically as part of these predators diets (Jones et al., 2005; Murphy et al., 2004; White, 1994), but this is not surprising given the rarity of the grasshoppers. Of the six remaining populations of *B. robustus*, only the Paterson's Terrace population receives predator control in the form of a recently installed 1.1 m high predator-exclusion fence (Schori et al., 2019). This fenced area is only 6,000 m^2 and was developed to research the effects of predation on *B. robustus* by assessing survival and population growth inside the fence verse outside where there is no predator control (Schori et al., 2019).

Brachaspis robustus is particularly susceptible to mammalian predation as it relies on its cryptic appearance rather than escape (White, 1994), (F. Thorsen, 2010, unpublished data). Freezing and camouflaging into surroundings is an effective method used on predators who rely on sight to hunt (Gibbs, 2010). However, introduced mammals such as rodents, feral cats, mustelids and hedgehogs use olfactory senses to find prey (Hoare, 2006). This makes *B. robustus* particularly vulnerable as they exude scent as a way of communicating with members of the same species (Gibbs, 2009). *Brachaspis robustus*, along with many grasshopper species, are ectothermic (Field, 2001): as temperatures cool with the sun's setting, their mobility becomes restricted, further increasing their vulnerability to nocturnal predators (Chapman & Joern, 1990). Observations of *B. robustus* during the night-time have confirmed that the species is not active at night (Schori, 2020)

Brachaspis robustus are also prey to native lizard species, with which they co-evolved (Daugherty, Patterson, & Hitchmough, 1994) and possibly predatory spiders (Schori et al., 2019). A wolf spider, genus *Anoteropis*, possibly *A. senica*, was seen and photographed (Figure 2.1) close to a female *B. robustus*. *Anosteropis*, like most spiders, are predators mainly of insects (Stratton, 1984). Redback spiders (*Latrodectus hasseltii*) are a highly venous spider species known from Australia (Whyte & Anderson, 2017). They are likely to become a threat to *B. robustus* in the future as they are currently found inland in Otago (Vink et al., 2011). Redback spiders usually prey on insects, but can catch and eat small lizards (Metcalfe & Ridgeway, 2013). *Brachaspis robustus* at all life stages, but especially nymphs, are susceptible to predation from web-building spiders as they can easily become stuck in a web whilst jumping (T. Murray personal communication, December 14, 2021).

There are several skink species present in the Mackenzie basin that inhabit similar habitats to *B. robustus* (O’neill, Chapple, Daugherty, & Ritchie, 2008). Although it has not been proven that skinks prey on *B. robustus*, they are known to consume other flightless insect species (Gibbs, 2010). As a likely pressure on the survival of *B. robustus*, the presence/absence of skinks in the current locations of *B. robustus,* as well as selected translocation sites, must be determined. Even low levels of predation can exert a large amount of pressure on endangered native species, and incursions to a predator-free habitat can quickly degrade the native fauna there (Norton, 2009). Schori (2019) showed that predator suppression is not sufficient for the conservation of this species, and complete predator exclusion is required (Schori et al., 2019).

Figure 2.1 An adult female *B. robustus* close to a wolf spider (left) and a wolf spider (right). Although this grasshopper is much too large to become prey to this spider, nymphs of *B. robustus* could certainly be small enough. Photos: Rebecca Clements (2020).

Te Manahuna Aoraki has an extensive mammalian predator control regime for a large area of the upper Mackenzie Basin Figure 1.3. Current predator control surrounds the Tasman River, Cass River, Godly River and Macaulay River. There is also a small area of intensive predator control surrounding a black-fronted tern breeding colony on the upper Ōhau River (Schori et al., 2019; Te Manahuna Aoraki, 2020). The mammalian predators that present the biggest threat to *B. robustus* are hedgehogs, stoats, rats and mice (Schori et al., 2019). Large bodied invertebrates have been found to make up a large part of hedgehog diet in the upper Waitaki Basin (White, 1994) (Jones et al., 2005).

Native predators

Research has shown that mammalian predator control benefits *Brachaspis robustus* (Schori et al., 2019). However, reduced mammalian predator pressure may induce mesopredator release of native predators (McIver, 2020). Many bird, lizard and some spider species are also predators of *B. robustus,* and their populations may benefit from reduced mammalian predation (White, 1994). Native mesopredators likely cause less predation pressure on native invertebrates as many have adapted to avoid native predators through defence mechanisms such as visual crypsis (Gibbs, 1998). Native skink species from the *Oligosoma* genus are present in Mackenzie Basin and may prey upon *B. robustus* as they primarily prey upon arthropods (Hare, Chapple, Towns, & van Winkel, 2016).

Habitat modification

Habitat fragmentation restricts the dispersal of *B. robustus* to other suitable habitats. The Mackenzie Basin has been heavily modified by infrastructure and agriculture (Caruso, 2006). Manufactured structures such as roads and water canals all pose physical obstacles to the potential movement of *B. robustus*. There have been eight hydroelectric power stations constructed in the Mackenzie Basin since 1935, and these now provide between 25% and 30% of New Zealand's electricity (Caruso, 2006). Water abstraction and damming for hydroelectricity schemes has altered the flow regime and shape of the Ōhau, Waitaki, and Tekapō Rivers (Caruso, Ross, Shuker, & Davies, 2013). *Brachaspis robustus* is currently found on the Ōhau and Tekapō Rivers but is presumed to no longer exist on the Waitaki River due to habitat modification. Three artificial lakes have been constructed for hydroelectricity schemes. These are Lake Ruataniwha, Benmore and Aviemore (Young, Smart, & Harding, 2004)

Exotic plant species have a large impact on braided river systems. These weeds often smother out native plant species and clog waterways. Their roots stabilize river gravels, altering channel shape (Caruso, Pithie, & Edmondson, 2013). Weeds reduce the amount of open gravel areas needed for many specialist braided river birds, lizards and invertebrate species (Schori, 2020). Female *B. robustus* require a substrate for depositing their eggs. Field experiments reported in Schori (2020) showed that *B. robustus* preferred to lay their eggs in gravel that lined the cages they were in rather than the 2 L ice-cream container of sand provided. Captive-rearing experiments in a laboratory complimented these field experiments. In the laboratory, *B. robustus* readily laid eggs in the sand provided (Schori, 2020). Schori (2020) hypothesizes that *B. robustus* prefer to lay their eggs in gravel because the small stones protect against flooding events. Invasive plant species, as well as native, may reduce suitable egg-laying sites for this species. However, if *B. robustus* search for stable gravel or sand to lay their eggs in, plant roots would benefit. One *B. robustus* egg sack was found in the roots of a plant within the field experiment cages in Schori (2020). I agree with Schori (2020) that further studies in a natural environment need to be conducted to understand what substrate this species requires for egg laying.

Plants that have an upright structure may also inhibit the thermoregulation, movement and dispersal of *B. robustus.* Dense vegetation is seen to inhibit the mobility of a flightless wetland grasshopper, *Chorthippus montanus*, in Germany (Weyer, Weinberger, & Hochkirch, 2012). Results from Schori (2020) showed that *B. robustus* were found in open, exposed areas much more frequently than shaded or sheltered positions and on rocky substrate rather than vegetation (Schori, 2020). This grasshopper species is a basker, meaning they bask in the sun to warm their internal temperature and increase activity which is needed for digestion, reproduction and other biological processes (Forsman, 1999). Increased vegetation cover shades out potential basking sites for *B. robustus*. Rocky substrate holds heat energy from the sun more efficiently than vegetation, making rocks and stones the preferred basking surface rather than on vegetation (Huey, Peterson, Arnold, & Porter, 1989).

Invasive weeds such as willow stabilise river channels. Disturbance events may be necessary for *B. robustus* as it has been suggested that they prefer younger, more disturbed river gravels. This hypothesis is supported by my survey of ground cover preferences of this species in Chapter 4.

Hybridisation

Small or fragmented populations are at risk of a genetic bottleneck, leaving an endangered species even more vulnerable to extinction (Nunney & Campbell, 1993). Loss of genetic diversity means a population is not safeguarded against diseases and the need to adapt to environmental changes (Toro & Caballero, 2005). There are three species in the *Brachaspis* genus; *Brachaspis collinus, B. nivalis, and B. robustus* (Bigelow, 1967). *Brachaspis collinus* and *B. nivalis* occupy alpine environments, and *B. collinus* only occurs in the top of the South Island and Arthurs Pass (Staples, 1967). Genetic information suggests that *Brachaspis* radiated during the Pliocene (3–5 mya), however, natural isolation of the alpine species, *B. collinus* and *B. nivalis* and the lowland *B. robustus* may have only occurred at the end of the last glacial period (~10 kya) (Trewick, 2001). Trewick (2001) suggests that gene flow between the alpine species and *B. robustus* may have only been significantly diminished with the arrival of humans some 800 years ago due to changes in vegetation and introduced predators.

There is evidence for hybridisation between *B. collinus and B. nivalis,* but it has not been confirmed if the hybrid offspring were sterile (Trewick, 2001). *Brachaspis robustus* are very genetically similar to *B. nivalis* (Koot et al., 2020). Although the two species occupy quite different habitats, they are both present in the Mackenzie Basin. *Brachaspis nivalis* typically are found on scree slopes above 1500 m whereas *B. robustus* remain on braided riverbeds. There is a risk the two species would hybridise if they came in contact (Koot, 2018), which would be detrimental to the genetic diversity of *B. robustus*. There is work being done to understand the likelihood of the two species mating and producing viable offspring (T. Murray, personal communication, November 16, 2019). *Brachaspis robustus* are only weakly genetically differentiated from *B. collinus* and *B. nivalis*. Even so, scientists agree that the conservation of *B. robustus* is important for preserving biodiversity (Trewick, 2001). Another associated risk of *B. nivalis* and *B. robustus* occupying the same habitat is that even if they do not hybridize, they may compete for the same resources. This could disadvantage either of the species.

Climate change

For many species, translocation sites must be selected to mitigate against future impacts of global warming (Rout et al., 2013). The Earth is the warmest it has ever been and is predicted to continue warming rapidly for the next 50–100 years (Easterling et al., 1997). We have already seen many consequences of climate change as mean temperatures increase. Global warming in this century has caused climatic isotherms to shift northwards in the Northern Hemisphere. Many plant and animal species have distinctive temperature ranges that they can grow and reproduce within. The ranges of many of these species are becoming restricted and modified (Parmesan & Yohe, 2003). A comprehensive study 35 of non-migratory European butterflies shows a definite shift in ranges towards the north pole, away from the equator (Parmesan et al., 1999). Global warming also alters the length of the growing season and plants are flowering earlier (Menzel & Fabian, 1999). Changes in phenology have also been observed for many insect species. Warmer temperatures affect the life-cycle of some cold-adapted invertebrate species (Gitay, Suárez, Watson, & Dokken, 2002). Changes in bird and insect migration cause a mismatch with food resources and breeding (Knudsen et al., 2011). Species have shown changes in morphology, physiology and behaviour in response to climate changes. Global warming has also resulted in an increased frequency and severity of disease outbreaks in insects (Gitay et al., 2002).

With the little biological information we have on *B. robustus*, it is predicted that as alpine and subalpine temperatures in New Zealand rise, *B. robustus* will be severely impacted, along with many other alpine and subalpine insect species adapted to freezing winter temperatures. It is thought that the eggs of *B. robustus* require a period of very cold temperatures to break diapause before hatching (Koot, 2018). Because *B. robustus* relies on sufficiently cold temperatures to complete its life cycle, there is a significant concern for the future of this species (Schori, 2020), as the climate change projections for the Mackenzie Basin show that annual mean temperatures are expected to increase by 0.5–1.5°C by 2040 (Macara et al., 2019). If the number of days below 0°C declines below the required threshold within the current distribution, this species may be at risk. This is because their dispersal to colder locations (e.g. closer to the Southern Alps) is restricted by flightlessness and natural and man-made physical barriers to movement (e.g. dams, roads, vegetated outwash plains and hill country) (Schori, 2020). The Ahuriri River where the holotype of *B. robustus* was collected occurs between 500 m and 1000 m elevation, its highest point being at the base of the Southern Alps. Paterson's terrace is at 700 m (NZ Topo Maps, accessed 13/08/2021). Snowy River, where possibly the largest population of *B. robustus* occurs, is at 600 m. The Lower Ōhau River is at 400 m, Depending on where exactly on the Ahuriri River *B. robustus* was collected from. While anecdotal evidence may suggest that *B. robustus* has had a slight range shift as they are no longer extant at the lowest elevation of their historical range, there are other important factors that influence *B. robustus* distribution. For example, earlier arrival of weeds, mammalian predators and different anthropogenic influences all play a role.

Based on this review of the literature, a number of key factors important to the development of a successful translocation plan for *B. robustus* have been identified. In the following chapters I will assess habitat suitability, individual *B. robustus* home range, microhabitat requirements, mammalian predation and hybridisation with *B. nivalis*.

Chapter 3

What sized area is required to support a translocated population of *B. robustus*?

3.1 Introduction

3.1.1 The importance of movement in the flightless grasshopper

When considering translocation for animals, it is important to know the range of the species and how big a habitat they require (IUCN, 2013). Habitat suitability is important for all organisms, especially for an endemic species such as *B. robustus*. An appropriately sized area of suitable habitat must be available to the translocated species (IUCN, 2013). A radio-tracking study by Schori (2020) estimated the home range of adult female *B. robustus* is likely between 250 m^2 and 300 m^2 of habitat, based on relocation distances of the same individuals. Males were not tracked as they are too small for a transmitter to be attached (Schori, 2020). Because there is still no information on the maximum population density of *B. robustus*, it is unclear what sized area of suitable habitat is required to support a of population of *B. robustus*, nor what density is required to enable mate finding (Schori, 2020). To better understand the area of habitat required for a successful translocation of *B. robustus* further data on the movement of individuals, including male grasshoppers, is needed.

Methods for assessing range size for insects

For short-term results, the most effective way of determining an animals range is to track it remotely (Lövei, Stringer, Devine, & Cartellieri, 1997). Radio telemetry tracking devices have been used on various invertebrate species (Kissling, Pattemore, & Hagen, 2014), including wētā and carnivorous land snails in New Zealand to learn more about their movement (McCartney, Armstrong, Gwynne, Kelly, & Barker, 2006; Stringer & Chappell, 2008; Stringer, Parrish, & Sherley, 2018; Watts, Empson, Thornburrow, & Rohan, 2012).

Radio telemetry involves a battery-powered transmitter that is attached to the animal, the transmitter regularly emits a radio signal at a fixed radio frequency, at a programmed pulse rate. The signal is recorded by radio receiver. The main issue with using radio telemetry is that battery life and signal range are proportional to battery size. Consequently, radio telemetry has historically been used only for large bodied (>300 g) vertebrates (Wikelski et al., 2007). Only recently have radio telemetric devices been made small enough to attach to invertebrates (Kissling, Pattemore, & Hagen, 2014). For example, Mormon crickets (*Anabrus simplex*) were fixed with radio transmitters to track individual movement to assess differences in outbreak and non-outbreak populations (Lorch, Sword, Gwynne, & Anderson, 2005). This species creates migratory bands that move through the landscape and are destructive to vegetation. Radiotelemetry showed that, on average, non-outbreak population individuals travelled 0.66 m per day, while outbreak population individuals travelled 331 m per day (Lorch, Sword, Gwynne, & Anderson, 2005).

Harmonic radar transponders are also passive and use a diode to reflect a continuous harmonic radar wave. The diode is attached to the insect that is being tracked, these weigh very little and do not rely on an attached energy source (Psychoudakis, Moulder, Chen, Zhu, & Volakis, 2008). A portable transmitter/receiver emits a continuous frequency, the diode tag on the organism reflects this signal and the transmitter/receiver transforms it into an audible signal (Lövei, Stringer, Devine, & Cartellieri, 1997). However, in practicality, the transmitter/receiver is very large and not easily carried. Harmonic radar transponders were used to monitor translocated populations of Middle Island tusked wētā. Devices were attached to adult wētā that led researchers to burrows where their un-marked mate was. An infrared time-lapse video was also used to observe the wētā within a predator-free enclosure to determine if they were returning to the artificial burrows that they were released into. This increased the effectiveness of monitoring post-release (Stringer & Chappell, 2008). Harmonic radar transponders were also used to monitor the movement of the giant land snails *Placostylus ambagiosus* and *P. hongi.* This gave the researchers insight into the behaviour and dispersal of these species post-release. They found that many of the snails returned to the location they were originally collected from, using homing techniques (Stringer et al., 2018). The movements of thirty-six Cook Strait giant wētā (*Deinacrida rugosa*) were monitored using radio transmitters immediately after their release at Zealandia, an eco-sanctuary in Wellington. Radio-tracking tags were glued to the pronotum of wētā. The purpose of this translocation was to educate the general public on this species, as well as achieving conservation outcomes for the species and the ecological restoration of the ecosanctuary (Watts et al., 2012).

Radio frequency identification (RFID) is a type of passive radio tracking (Kissling et al., 2014). The RFID tags do not contain batteries or a power source, rather, radio signals emitted by a hand held scanner are used to activate the tag. One of the major constraints of RFID is that the tags can only be detected within a few meters (Van Geystelen, Benaets, de Graaf, Larmuseau, & Wenseleers, 2016) or even centimetres if the tags are very small.

Mark-recapture methods can be helpful in post-translocation monitoring of invertebrates and enable long-term monitoring if marks are not lost due to moulting(Haglrt & Jackson, 2001; Jamieson, Forbes, & McKnight, 200; Janks & Barker, 2013). Like transmitters, mark-recapture is an invasive and sometimes destructive method of monitoring as it requires handling and marking of the invertebrates (Besnard, Piry, Berthier, Lebreton, & Streiff, 2007). There are many ways to mark an invertebrate for mark-recapture. These include externally fixed tags, mutilation (e.g. toe clipping), paint and ink, dust/powder marking. Methods such as internal dye and protein marking are not practical for endangered insect species due to heightened risk of harming individuals (Hagler & Jackson, 2001). Mark-recapture methods involve capturing and marking insects, sometimes with uniquely identifiable codes, and releasing them. The same methods are repeated to search for and capture individuals from the species after a set intervals. Individuals that were recaptured with marks are distinguished from those who were not caught the first time. This process can continue for as long as is required (Hagler & Jackson, 2001). Mark-recapture methods were used to monitor Raukumara tusked wētā (*Motuweta riparia*) after the species was recently discovered. Most marked wētā were recaptured within 10 m of their original capture site, but some individuals were recorded travelling 80 m in one night (McCartney et al., 2006).

Regardless of the technology used, tracking comes multiple challenges. For example, many invertebrates moult throughout their development. If a tag is attached to an insect, such as a grasshopper, before reaching adulthood, the device will be lost during moulting. Therefore, only adults are used to track. Additionally, all marking methods for invertebrate tracking involve handling the animal and inevitably causing the individual stress (Teixeira, De Azevedo, Mendl, Cipreste, & Young, 2007). Stress can have detrimental effects on animals causing mortality, dispersal, and inhibiting reproduction success (Armstrong & Seddon, 2008). To minimise stress, handling of the animal should only be done when absolutely necessary and for as little time as possible. Handling of animals should not be conducted in conditions that increase stress e.g. extreme heat (Letty, Marchandeau, & Aubineau, 2007). Mark-recapture is cheaper than using radio-tracking devices and removes the obstacle of technical difficulties that sometimes accompanies radio devices. Therefore, if financial resources are a constraint of the project, more individuals are able to be tracked using mark-recapture. Attaching radio-tracking tags may affect insect behaviour (Kissling et al., 2014). One study showed that tracking-tag weight significantly compromised the movement of a flightless cricket species (*Gryllus locorojo*) in low temperatures. Like *B. robustus*, *G. locorojo* is an ectotherm and is; therefore, temperature dependant (Kaláb et al., 2021). In contrast, Watts (2011) reported that transmitter weight (4.8–5.2% of female body weight and 8.1–8.9% of male body weight) did not affect that distance travelled by wētā at night (Watts, Stringer, Thornburrow, & MacKenzie, 2011).

Brachaspis robustus are difficult to detect due to their cryptic appearance and patchy distribution within their range. Visual observation methods used to date rely on the grasshoppers moving. These methods also only work in fine, sunny weather conditions (Schori, Steeves, & Murray, 2020a). Low detectability combined with the lack of targeted monitoring may have resulted in inaccurate distribution estimates throughout the landscape as false absences are recorded (Schori, Steeves, & Murray, 2020b). Low detectability makes understanding how individuals move throughout the habitat and therefore how big an area is needed for translocation difficult. Miniaturisation of radio telemetry technology allowed Schori (2020) to obtain some data on female *B. robustus* movements, but not for very long due to battery life and weight of tags. Schori (2020) previously used 0.22 g radio transmitters to attach to female *B. robustus*, but males of this species do not reach a size large enough to carry a radio transmitter without movement being hindered. This study also found that the distance travelled by the female grasshoppers decreased as the transmitter to bodyweight proportion increased (Schori, 2020). Due to these issues, mark-recapture or lightweight RFID tags provide a solution for monitoring *B. robustus* movement and therefore allow us to inform translocation of this species.

3.2 Objective

The objective of this chapter was to gather information on the range *B. robustus* travel within their habitat and therefore estimate how much space they would require in a translocation site. Mark-recapture was used as a lower-risk, practical method but RFID tagging was also trialled. RFID tagging presented technical difficulties that were not able to be resolved within the time frame of the project.

3.3 Methods

3.3.1 Site description

The Snowy River (-44.2234º, 170.5029º) (Figure 3.1) is a river outwash stream bed that is usually dry but has been known to flood during heavy rainfall (White, 1994) (Schori, 2020). This site is on private farmland; cattle and sheep are often run on the surrounding grassland and can roam the riverbed. There is also a farm road that follows beneath the power lines that cut diagonally across Snowy River, and this is used occasionally by farm staff. There are distinct edges where the grassland stops and the rocky riverbed begins. A steep bank drops away from grassland into a riverbed on the side closest to Hakataramea Road and the boundary fence line (Figure 3.2).

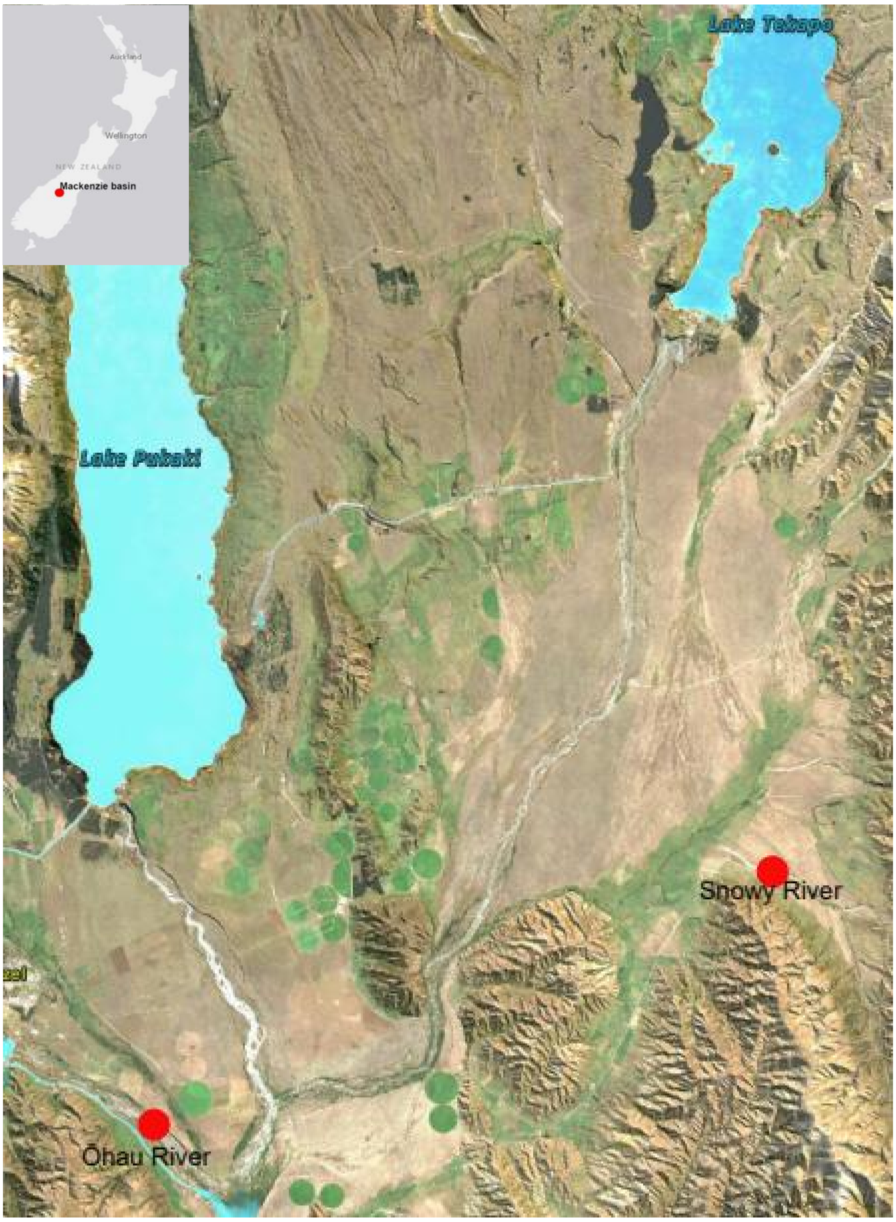


Figure 3.1 Location of Snowy River and Ōhau River where populations of *B. robustus* have been detected since ~1995. Satellite imagery source: canterburymaps.govt.nz

Figure 3.2 Steep bank dividing grassland and rocky riverbed at Snowy River.

Snowy River was chosen from the six known populations of *B. robustus* because it has the highest density population that occurs on a natural braided riverbed. There may be other dense clusters on the larger braided rivers, such as the Tekapō River, but because of their patchy distribution and cryptic nature these are not readily detectable (Schori, 2020). Grasshoppers are more accessible on Snowy River than the dispersed populations on the larger rivers such as Ōhau and Tekapō Rivers. While Patersons Terrace possibly supports a larger population of *B. robustus,* the habitat (a disused gravel road constructed in the 1970s using stones from the Tekapō River) is not natural. It is thought that the population of *B. robustus* was established at Patersons terrace either after individuals were accidentally transported from the Tekapō River with stones used to build the road, or they naturally migrated downstream from the Forks stream population (McKay, Anderson, Light, & Hancock, 1978). There are two contrasting types of habitat at Snowy River: rocky riverbed and vegetated land surrounding the riverbed. From previous research on the biology of *B. robustus* and sightings of this species throughout the Mackenzie Basin, we assumed that they would mostly be confined to the rocky riverbed (White, 1994). However, it is important to have a high level of assurance that they do not also occupy vegetated habitats that are thought to be unsuitable for *B. robustus*.

The *B. robustus* population on the lower Ōhau River (-44.3310º, 170.1797º) was included in this part of the study, but far fewer days were spent searching that area. Consequently, less data was collected for home range and habitat preferences of *B. robustus* at the Ōhau site. Less search effort was put into the Ōhau River site because the site is significantly larger than Snowy River, making it more difficult to find the grasshoppers. Additionally, time constraints of this study did not allow for frequent travel between the two sites. Because of the small dataset collected from the Ōhau River, data from both Snowy and Ōhau Rivers were combined for the analysis. At Ōhau River, the riverbed has a fairly uniform ground cover and does not have similar vegetated banks to Snowy River.

3.3.2 Grasshopper detection

For a thermophilic species such as *B. robustus*, monitoring must be carried out on warm days to increase the likelihood of sighting the grasshoppers. The technique used to monitor *B. robustus* involves an observer slowly walking through the habitat and systematically searching for grasshoppers, utilising the slow-walk technique described above. The grasshoppers are disturbed by the observer's presence and will jump, making it easier to capture them or to see their mark-recapture code (Schori et al., 2019). Once a grasshopper was caught, its hind femur length and body length were measured, and sex and age (adult or nymph) recorded. Previous studies on *B. robustus* used hind femur length to determine adulthood as wing development is very difficult to see. A female is considered an adult when her hind femur is ≥15 mm long, and a male is an adult when the hind femur measures ≥9 mm (Schori, 2020).

3.3.3 Radio Frequency Identification

Permission to handle the threatened grasshopper *Brachaspis robustus* was granted by DOC (DOC project reference DOCDM-1528162) and training provided to ensure best practice was followed.

I searched for grasshoppers to tag by walking slowly, with my front leg held low to the ground and sweeping to and fro systematically across the habitat. The motion of the sweeping leg is to elicit a jump response from the grasshoppers which made them easier to find (Schori, 2020). The RFID tags used in this experiment weighed 0.10 g. Only larger (body length >25 mm) female grasshoppers were targeted for this experiment (Figure 3.1 a) as males of this sexually dimorphic species are too small to carry an RFID tag without being inhibited.

The process of attaching the RFID transmitters was conducted in the shade to reduce stress on the grasshopper. A small dab of super glue was placed in the centre of the pronotum, a couple of minutes was allowed to let the glue semi-dry so that it was tacky. The RFID tag was then placed on the glue (Figure 3.1 a). The grasshopper was held for a further 3 minutes to ensure the glue was dried. The grasshopper was then released in the same location it was captured. The unique code of the RFID tag was recorded using a hand-held RFID reader (not pictured). The GPS coordinates of the release location were recorded using a Garmin (Garmin GPSMAP 64), along with body length and hind femur length.

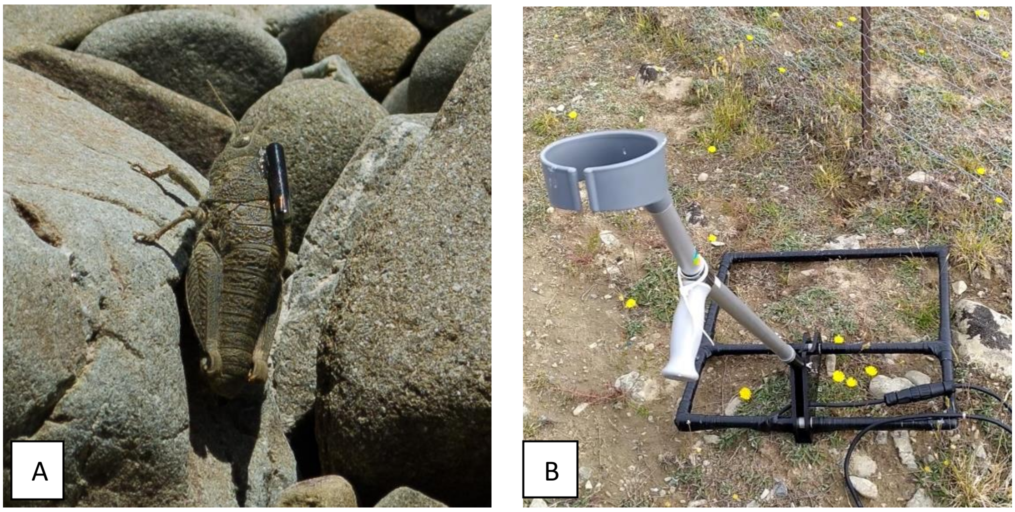


Figure 3.1 (A) Adult female fitted with RFID transmitter. (B) RFID transceiver, the ‘hopper wand’. Photos: Rebecca Clements (2020).

Two adult female grasshoppers were fitted with RFID tags. In the days following their release I attempted to locate the grasshoppers again using a purpose built RFID transceiver, the ‘hopper wand’, provided by DOC (Figure 3.1 b). This is a device held by the observer, who slowly waves it back and forth over the ground whilst walking across the habitat. The transceiver must be held within 30 cm of the ground to ensure RFID tags on grasshoppers are picked up. The device beeps clearly when an RFID signal is received. When a tagged grasshopper is located, a small hand-held RFID reader detects the unique code of the RFID tag. All of this can be done without handling the grasshoppers. However, technical difficulties were encountered when using the purpose-built RFID equipment. The RFID receiver had several malfunctions that could not be immediately repaired. We consequently decided to forgo further RFID tracking and use a capture-mark-recapture method with paint marking instead.

3.3.4 Paint marking with visual searches

Grasshoppers were found using the slow-walk methods described above. Capture-mark-recapture (CMR) involves marking each grasshopper caught with a unique code of painted dots using non-toxic paint pens (Edding® 780 gloss paint marker: https://www.edding.com/products/edding-780-gloss-paint-marker/). I utilised the 1-2-4-7 numbering method created by Buchvveitz and Walter (1992) (Figure 3.1 a) that has been extensively tested for *B. robustus* (Murray and Schori personal communication, November 4, 2019). This method is efficient and straightforward to use in the field, reducing handling time and thus stress on the grasshoppers (Dickens, Delehanty, & Romero, 2010). I used a modified version of this method and included an additional dot on the pronotum (Figure 3.3

a). This dot indicates which abdominal segment the paint markings begin, simplifying mark reading upon recapture and often allowing marks to be read from a distance without handling the insect. If the paint dots commence on the fourth segment, no pronotum dot was added. The numbering system was used in conjunction with different coloured paint pens, for example; "Pink 15" and "Green 49" (Fig. 3.3 b and c respectively).

A major drawback of this method is that there is no way to estimate marks lost due to individuals moulting. Although the objective here was not to estimate population size, which is the usual purpose of CMR (Besnard et al., 2007), mark loss was still a problem as adult *B. robustus* grasshoppers do not moult, they were targeted for CMR, but larger nymphs were also marked to maximise the amount of data gathered on grasshopper movements. To maximise chances of resighting marked juveniles, many individuals needed to be marked and searches needed to be scheduled regularly to minimise the likelihood moulting would occur before I was able to attempt to resight the individual.

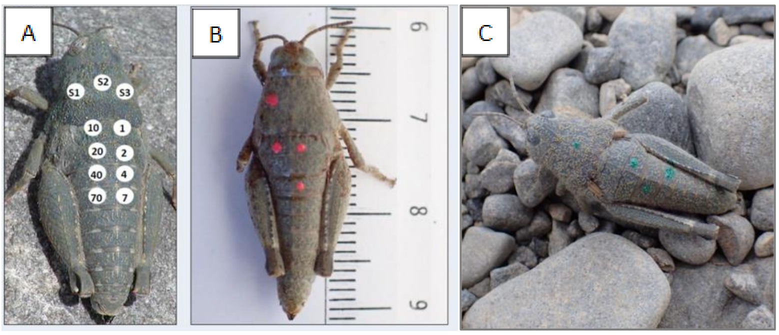


Figure 3.3. ***Brachaspis robustus*** **with (A) The locations of dots utilising the 1-2-4-7 method with the addition "S-"dots on the pronotum; (B) application of 1-2-4-7 method with non-toxic paint marker to individual "Pink 15" (10 + 1 + 4 = 15) with application of "S1" dot to denote marking begins on the first abdominal segment; (C) Application of 1-2-4-7 method with non-toxic paint marker to individual "Green 49" (40 + 2 + 7 = 49) with application of "S2" dot to denote marking begins on the second abdominal segment. Reused with Permission from (Murray & Maloney. (2015). 64th Annual Conference of the Entomological Society of New Zealand, Auckland, 7-10 April 2015).**

Using the 1-2-4-7 method described above, each grasshopper was marked with a unique number. The location the grasshopper was first sighted was GPS located. There was often a short period of pursuing a grasshopper after it was first seen. Therefore, I placed a coloured rock at the location I saw the grasshopper jump from so that I could come back to the exact place to GPS record. The time of day was recorded each time a grasshopper was caught. Ground surface temperature, air temperature at 1 m, relative humidity and average wind speed were measured once every hour using a Kestrel 3500 weather meter throughout each search effort. Although *B. robustus* are more active during warmer temperatures, searching was conducted in various weather conditions and temperatures to monitor behaviour changes.

Once a grasshopper was captured or sighted, an area approximately 30 m in diameter was searched for other grasshoppers. The area was explored by starting at the capture or sight location and spiralling out in circles, using the slow-walk technique. If another grasshopper was captured, the methods were repeated from that location. Once an area of about 30 m had been searched and no further grasshoppers were found, I continued my search in the direction I was originally walking. This resulted in search efforts being concentrated where patches of *B. robustus* occurred within the habitat. When searching the riverbed for *B. robustus*, I usually travelled in straight lines back and forth across the riverbed or long transects up and down the length of the riverbed (Figure 3.6, 3.7). I usually tried to walk so that I was not looking at my shadow, as constantly adjusting between my shadow and the bright surface of river stones made it difficult to see.

The surrounding vegetated riverbank was also systematically searched for the presence of *B. robustus*. An area of vegetated habitat 50 m wide on either side of the Snow River, for the length of the riverbed, was examined using the same slow walk technique. Distance to nearest open gravel was measured each time a grasshopper was caught not in the riverbed.

3.3.5 Data analysis

Distances between grasshopper recapture points were calculated using the GPS coordinates marked during the data collection. Data was used to estimate how far grasshoppers travel within their habitat. GPS points of grasshopper observations and GPS search tracks were inserted onto ArcMap version 10.8.1. Area searched polygons were created by clipping a polygon around the outermost track of all combined search days.

3.4 Results

Two days were spent searching with the RFID receiver in the area where the two female *B. robustus* were tagged at Snowy River. Neither of the tagged grasshoppers were able to be located again and this marking method was subsequently abandoned.

3.4.1 Capture-mark-recapture

A total of 20 days were spent conducting CMR at Snowy River and four days at Ōhau River. This field work was conducted between 24th of January and 20th of March, 2020. The only day that I searched for *B. robustus* and found none was on the 19th of March, 2020, at Ōhau River. The approximate area searched for *B. robustus* over the duration of this study at Snowy River was 23.6 hectares (Figure 3.6). About 18.3 hectares was searched at Ōhau River (Figure 3.7).

One hundred and nine *B. robustus* individuals were captured and marked at Snowy River (Table 3.1), and ten individuals were marked at Ōhau River, however, only 14 grasshoppers were recaptured and therefore useful for assessing range. As such, I present as basic description and analysis of grasshopper movement as my sample size was not large enough to run statistical tests e.g. between sites, sexes, or life stages. The combined recapture rate of male and female *B. robustus* at Snowy River was 9.2 %, with 10 out of the 109 marked grasshoppers recaptured once, one grasshopper recaptured twice and another recaptured three times. Significantly fewer *B. robustus* individuals were marked at Ōhau. This was due to the fewer study days spent at that site and the apparent smaller population of *B. robustus* present there.

Table 3.1 Demographics of *B. robustus* found and marked at Snowy (20 days searching) and Ōhau (4 days searching) Rivers between 24 January and 20 March 2020. 'Total caught' is the sum of all male, female, sub-adult and immature individuals caught over the entire period excluding any recaptures.

Site	Sex	Adult	Sub-adult	Immature	Recapture instances	Total caught (excluding recaptures)
Snowy River	Male	2	13	25	3 (7.4%)	40
	Female	5	4	60	7 (10.1%)	69
	Total	7	17	85	10 (9.2%)	109
Ōhau River	Male	0	0	2	0 (0%)	2
	Female	0	0	10	4 (40%)	10
	Total	0	0	12	4 (33.3%)	12

The minimum distance a grasshopper travelled in a day after marking was 2.5 m, while one individual (P22) travelled considerably further than all the other grasshoppers and was recaptured 49.8 m from the original capture location one day after marking (Table 3.2). All other individuals were recaptured within 16 m of the original capture location (Table 3.2), even after 17 days from marking. The mean of the maximum distances moved by the 10 grasshoppers that were recaptured at Snowy River after marking was 12.0 m. Individual P33, a juvenile female, was recaptured three times throughout the study (Table 3.2). The three recaptures of P33 were made over 17 days. The time between capture and application of mark, and the first recapture, was 15 days. The next two recaptures were made in the next two consecutive days.

Table 3.2 Distances between capture and recapture locations of *B. robustus* at Snowy River, 0 = first capture and mark, A = first recapture, B = second recapture, C = third recapture.

Grasshopper code	Sex and Life stage	Dist. 0-A.	Time elapsed	Dist. 0-B	Time elapsed	Dist. 0-C	Time elapsed
Br17	Female sub-adult	8.3 m	1 day				
G16	Female juvenile	3.8 m	1 day				
W5	Female juvenile	5.4 m	1 day				
P09	Female juvenile	15.5 m	1 day				
P10	Male juvenile	2.5 m	1 day	5.6 m	2 days		
P12	Female juvenile	2.9 m	1 day				
P22	Female juvenile	49.8 m	1 day				
P07	Male juvenile	7.6 m	2 days				
P33	Female juvenile	13.6 m	15 days	15.3 m	16 days	12.2 m	17 days
G22	Male juvenile	5.7 m	18 days				

The recapture rate of *B. robustus* at Ōhau River was 33.3%. The mean distance travelled by the four grasshoppers recaptured at Ōhau was 5.6 m. The minimum distance travelled after marking was 0.9 m and 13 days had elapsed between marking and recapture of individual OW10. The furthest distance between capture and recapture points at Ōhau was 11.4 m (Table 3.3).

Table 3.3 Distances between capture and recapture locations of *B. robustus* at the Ōhau River site, 0 = first capture and mark, A = first recapture.

Grasshopper code	Sex and life stage	Dist. 0-A.	Time elapsed
OW1	Female juvenile	4.4 m	1 day
OW7	Female juvenile	11.4 m	13 days
OW9	Female juvenile	5.6 m	13 days
OW10	Female juvenile	0.9 m	13 days

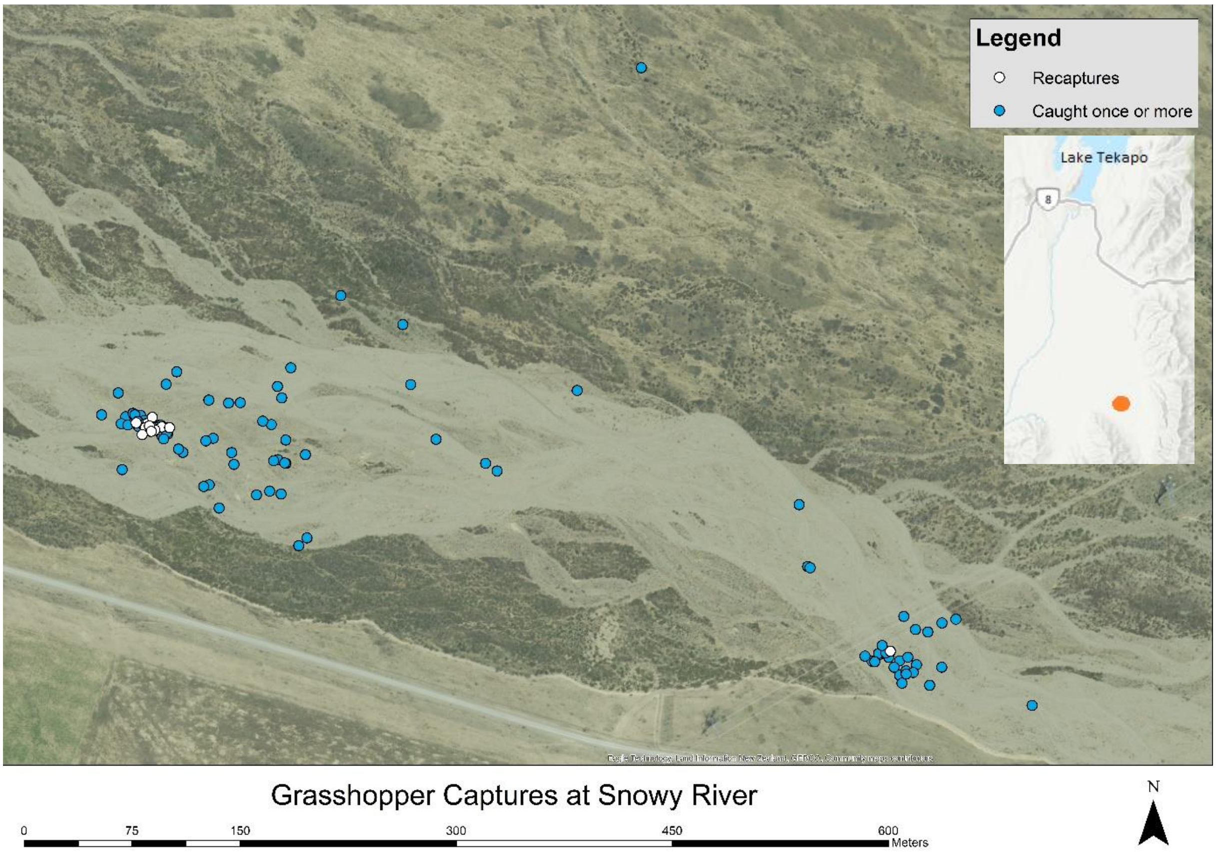


Figure 3.4 Grasshopper capture (Blue) and recapture (White) GPS points at Snowy River. There were two main clusters of *B. robustus* within the habitat. Assuming all blue dots are individual grasshoppers and not recaptures after mark loss due to moulting, this map displays the total density across the entire search zone. The majority of density in grasshoppers is in two main clusters to the far left and far right of the image.

Figure 3.5 ***Brachaspis robustus*** **capture (Blue) and recapture (White) GPS points at Ōhau River. Individual OW10 (juvenile female) was found less than 1 m away from its original capture spot after 13 days. Only the white recapture symbol is visible on the map due to the close proximity of the GPS points.**

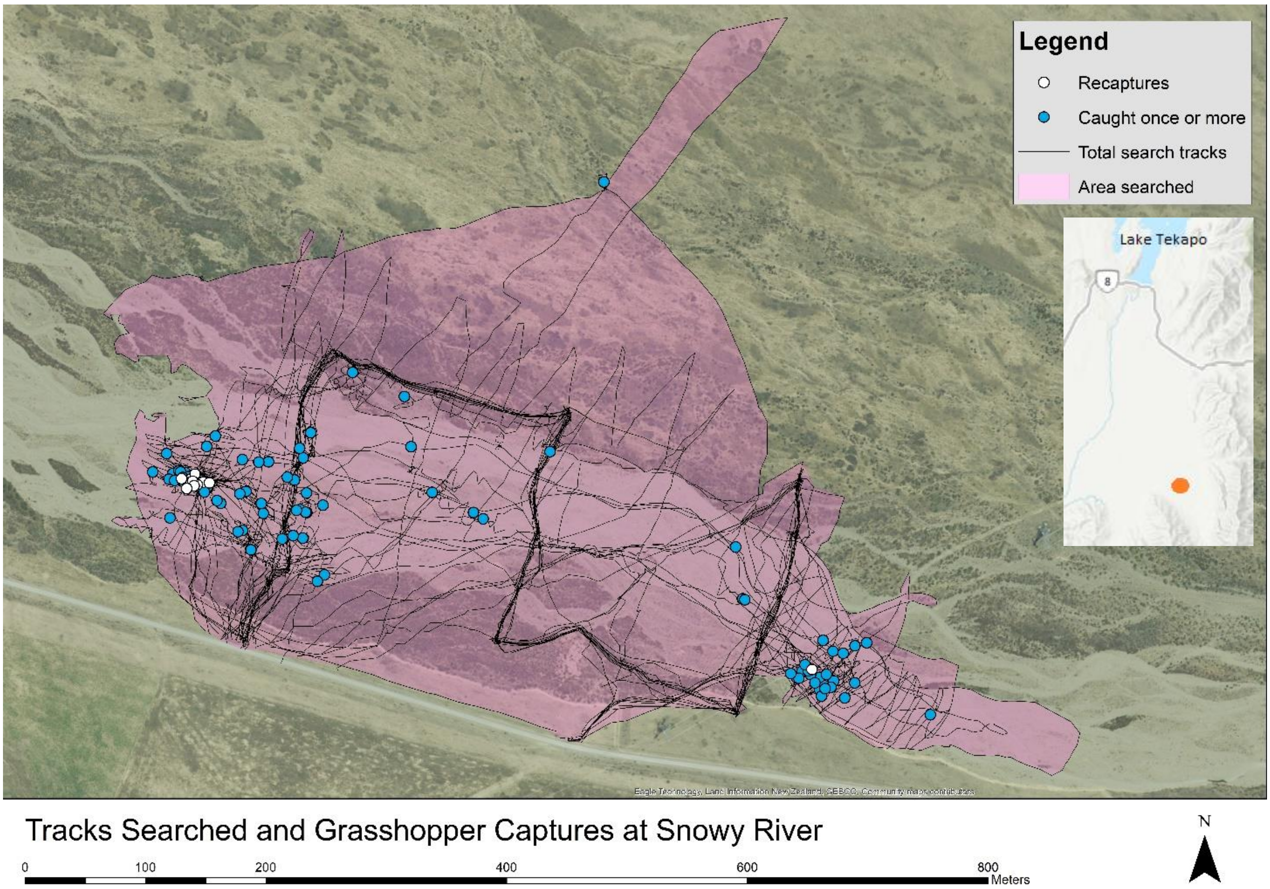


Figure 3.6 GPS tracks and area searched from all 20 search days at Snowy River. GPS unit was carried on me, the observer, whilst searching for grasshoppers and servicing tracking tunnels.

When walking along tracking tunnels transects, I was simultaneously searching for *B. robustus* grasshoppers. The total length of tracks searched over 18 days at Snowy River was 60.62 km. A width of 1 m either side of the search track was used to calculate the total area searched (121,240 m^2, 12.1 ha). The pink polygon is snapped the outermost track of the area searched. The area of this polygon is 21.3 ha.

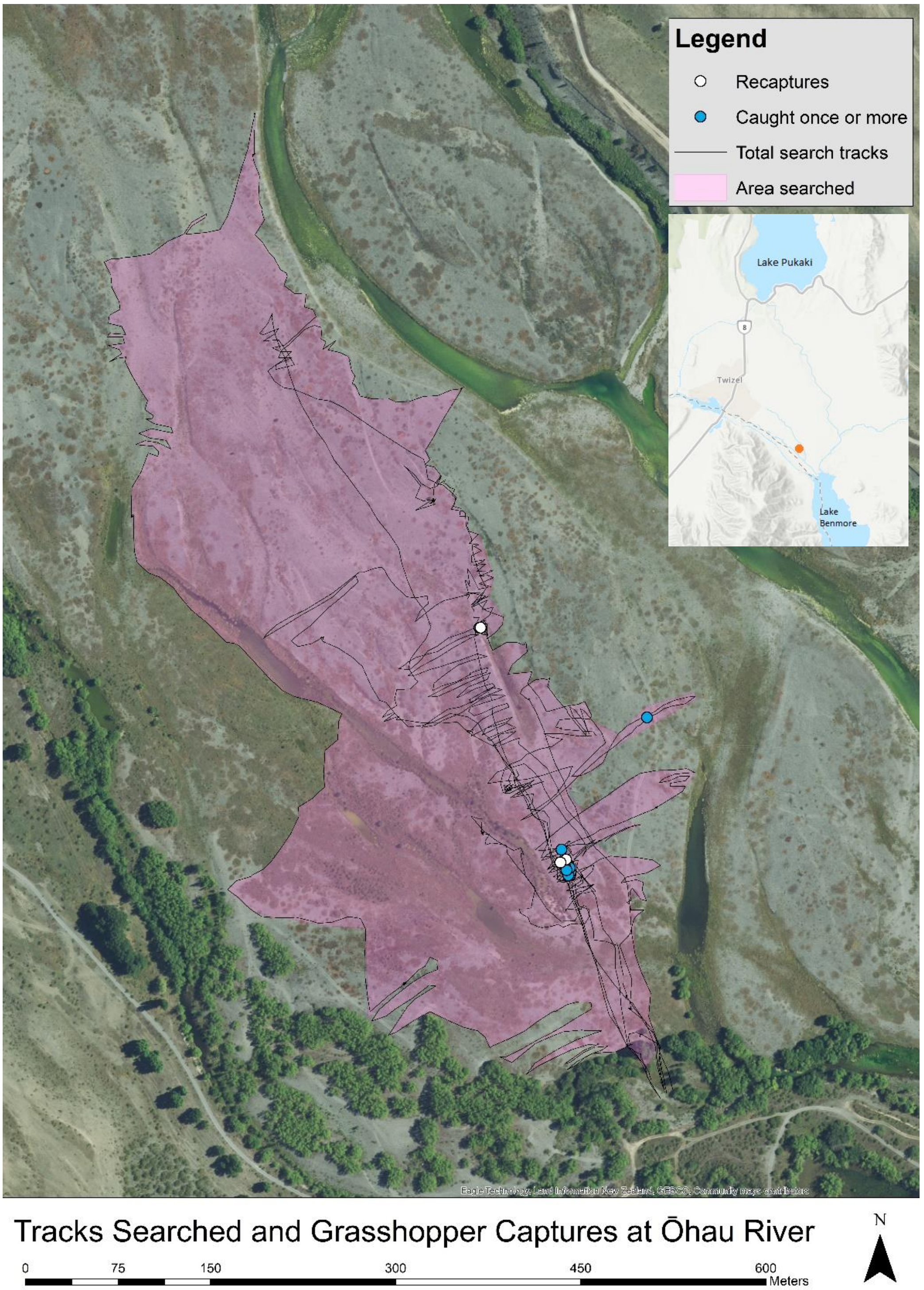


Figure 3.7 GPS tracks and area searched from all four search days at Ōhau River.

The total length of the tracks was 9.22 km. The total area searched was calculated using 1 m either side of the search track. Therefore, the total area searched at Ōhau River was 18,440 m^2, 1.8 ha. The polygon that encompasses the area searched is 15.5 ha.

3.5 Discussion

The objective of this chapter was to gather information on the range *B. robustus* travel within their habitat and therefore estimate how much space they would require in a translocation site. Due to the small sample size, this question could not be fully answered. However, from the 10 grasshoppers that were recaptured, it appears that *B. robustus* do not require large areas when in a juvenile life stage as most were recaptured within 16 m of the original capture sites after 17 days. One female juvenile at Ōhau River was recaptured less than 1 m away from the original captured spot after 13 days. All grasshoppers that were recaptured were juveniles and most were females (11 out of 14). One female juvenile grasshopper was found almost 50 m away from the original capture spot after one day. If this distance is representative of how far *B. robustus* travel within their habitat, an individual may require at least 2,500 m^2 of suitable habitat. This is because a 50 m by 50 m square equals 2,500 m^2. This is far larger than was suggested by Schori (2020) who found that an adult female grasshopper may require 300 m^2 of suitable habitat. It is notable that this is the first time males of this species have been tracked. Three male grasshoppers were recaptured, all were found within 8 m of the original capture spot after 18 days.

If it was feasible to track *B. robustus* movement through their entire life cycle, I believe that they would not actively range around much when not necessary. They might increase their movement when mates are needed to be found, or food resources. Females of this species require fine gravels or sand to lay their eggs in, so they would need to travel to find an appropriate spot to lay their eggs.

I attempted to use RFID tracking to monitor the movement of *B. robustus* at Snowy River as radio transmitters have successfully been used to monitor this species in the past (Schori, 2020). Unfortunately, I encountered several technical difficulties with the equipment, which would have taken far too long to fix given the scope and time pressure of this study. There was a limited window for monitoring adults of this species as adult *B. robustus* are only present from November–early December. Therefore, I decided to use capture-mark-recapture methods using non-toxic paint dots on the pronotum of the grasshoppers to gather movement data. This was carried out at both Snowy River and Ōhau River. Eventually, this too proved difficult and highlights the constraints of working with a very rare species. Fourteen out of 121 individuals marked were recaptured, a recapture rate of 11.6%. This resulted in minimal range data for this species. This may have been because most of the individuals marked were mid- to late- instar nymphs, not adults, and they may have shed off the marks with their exoskeleton in subsequent moulting or died before reaching adulthood. Orthoptera in general have high juvenile mortality rates (Dempster, 1963). That, along with the rarity of the species makes monitoring this species difficult. It is not known exactly how many instars *B. robustus* have, but it is thought they may go through 6 or 7 instar stages before becoming adults (Batcheler, 1967; White, 1994). The recapture rate reported here is not unusually low when compared to other invertebrate CMR studies. One study collated CMR data from 19 different studies on damselflies. The recaptured proportion of marked individuals ranged between 0.015 % and 0.902 % (Beirinckx, Van Gossum, Lajeunesse, & Forbes, 2006). Capture-mark-recapture initially resulted in low recapture rates (0–11%) for an endangered grasshopper, that is very similar to *B. robustus*, in rocky open grassland plains in France until researchers optimized their search methods and higher recapture rates were obtained (10–65 %) (Bröder et al., 2019). A study using fluorescent paint to mark grasshoppers in Wyoming (USA) native mixed-grass prairie resulted in 64% of grasshoppers being re-sighted at night using UV light, 28% being re-sighted during the day by visual searching, and 9% were recaptured with a sweep net (Narisu, Lockwood, & Schell, 1999). Another CMR study using fluorescent paint markings resulted in a 23.66 % of marked grasshoppers being recaptured. This study did however have a much higher density of marked individuals being studied (n = 1,179) as it is a common species. These methods are acceptable for common species but are inherently difficult for rare species like *B. robustus*. Ultra violet light was not used for re-sighting of marked grasshoppers as the sampling was conducted in the afternoon (Mussgnug, 1972).

A recent study by Schori (2020b) used a power analysis to detect population density trends in *B. robustus*. This study found that 100 m x 1 m transects, searched on days with no cloud, present the highest probability ($p_g > 0.6$) of detecting *B. robustus* (adult or nymph, male or female) at Snowy River, compared to overcast days with low temperature. The author recommends that if the objective of the study is to monitor *B. robustus* distribution at a landscape scale, a minimum of 3 search efforts should occur in February, under the above conditions. This is because nymph emergence is highest in February and thus species detection probability is highest. However, if the objective is to obtain population density information, monitoring should occur in November and December as this is when adult females of the species are most common (Schori et al., 2020b). The field work for this current study was conducted from the end of January to end of March out of necessity after initial marking methods attempted in December failed. This time coincided with the highest *B. robustus* densities but also the lowest adult densities, consequently, mark-loss due to moulting is highly likely. The data I collected is, therefore, more of an indicator of population distribution, rather than adult population density.

I hypothesise that another reason why very few individuals were re-sighted as capture and handling was undoubtedly a stressful experience, even though efforts were made to reduce handling time. The grasshoppers that were caught may have moved away from the study area as a result of the stressful encounter (Dickens, Delehanty, & Romero, 2010). Grasshoppers that were caught and handled for CMR often regurgitated their gut contents. This is a common predator defence mechanism seen in other Orthopteran species (Eisner, 1970). Grasshopper regurgitation is distasteful and sometimes toxic to predators, depending on the grasshopper diet. This can result in grasshoppers surviving predator attacks as the regurgitate leads to rejection before ingestion (Sword, 2001). The *B. robustus* that were handled during this study and regurgitated were therefore responding to what they saw as a predator attack (Eisner, 1970). This may have increased stress levels and also led to dispersal from the site (Dickens et al., 2010). However, I searched the riverbed extensively after releasing marked grasshoppers. If *B. robustus* disperse away from the capture site due to stress, we would not have expected to find most of the marked individuals within such close proximity as we did (most within 16 m after 17 days). Furthermore, a mark-recapture study of mountain stone wētā (*H. maori*) on rock tors in the South Island of New Zealand did not report any instances of immediate dispersal after release. About half of the wētā marked for the study were resighted (Jamieson, Forbes, & McKnight, 2000). Results from another wētā mark-recapture study showed little to no dispersal immediately after handling and release. The researchers used tracking tunnels to record *M. isolata* movement post-release on the Mercury Islands (Stringer et al., 2014). Little is known about how the stress of handling affects post-release dispersal in invertebrates as monitoring movements and survival is difficult (Parker et al., 2015). Therefore, stress may not have been the main reason why few grasshoppers were recaptured.

Another potential reason for why few marked individuals were recaptured is that the colourful paint markings may have made them more visible to predators. *Brachaspis robustus* usually relies on visual crypsis to hide from native predators such as birds (White, 1994). It is therefore plausible that colourful dots on the grasshoppers pronotum made them more visually conspicuous to birds and were predated upon. Bird eyesight is commonly known as superior in the animal kingdom. Most birds rely on sight to hunt and can see colours (Meyer, 1977). One study found that red coloured frogs were more conspicuous and easier to find for chickens, than green coloured frogs. The researchers also found that the pattern on the frogs determined detectability for to the chickens. Fewer and larger black dots, compared to small and many dots, made it easier for the chickens to find, independent of frog colour (Edström, 2012). The paint pens used to mark grasshoppers in this study are marketed as 'Low-odour ink'. However, it is unknown whether olfactory predators such as hedgehogs would still be able to smell the paint on the grasshoppers. If they can, this is another possible explanation for not recapturing marked individuals

Schori (2020) conducted a radio transmitter monitoring study that was published after my initial work. The results showed that an adult female *B. robustus* might have a range of >300 m^2 (Schori, 2020). The furthest distance a marked grasshopper was found from a previous capture in my study was 49.6 m. Whereas Schori (2020) reports that the furthest distance a female *B. robustus* was recorded (using a radio transmitter) from the original release site was 65 m after 10 days.

Furthermore, the maximum cumulative distance travelled from all GPS fixes was 148 m over 11 days (Schori, 2020). My researched built on the work done by Schori (2020) by increasing the sample size of female *B. robustus* movements tracked, and tracking males for the first time.

I hypothesise that the grasshopper at Snowy River were found in two main patches (Figure 3.4) because of the life stage of the demographic. Most of the grasshoppers found were early instar. It is thought that *B. robustus* have seven instars (Batcheler, 1967). Perhaps *B. robustus* do not disperse from where they hatched until later in their life cycle when they need different, or more resources. For example, adult female *B. robustus* need sand or fine gravel to lay their eggs in (Schori, 2020). Therefore, they may need to disperse further into the habitat to find suitable substrate for oviposition.

When searching for *B. robustus*, I thoroughly searched the riverbed in between the two main clusters at Snowy River, but did not go further upstream or downstream as often. The area searched at Snowy River was 12.1 ha and 20 days were spent searching. Coinciding monitoring with peak adult female emergence in the population (November–December) would have resulted in higher recapture rates as marks would not have been shed during moulting (Schori et al., 2020a). My search effort for marked individuals extended beyond the estimated range of adult females (17 m x 17 m) (Schori, 2020). Although it is possible the captured and marked individuals moved away from the area due to stress, or that they were predated upon. It is also plausible that because this species is hard to detect, more search effort and a higher number of marked individuals was needed to gain more data, using these methods.

3.5.1 **Conclusion**

In conclusion, *B. robustus* may move further throughout their habitat than previously thought. This has implications for selecting an appropriately sized habitat for translocation as the receiving habitat must be larger than the minimum area required by the founder population (IUCN, 2013). Although, home range of animals in source environments is not always indicative of their true home range. Source habitats are often degraded and of low quality, therefore, the animal may have to use a wider area of habitat to obtain its required resources (Schori, 2020).

Capture-mark-recapture proved to be a practical and inexpensive method of monitoring *B. robustus*. Although the recapture rate was low (14.29 %), it was not unexpected given the constraints of working with a rare, cryptic species that moults. Significant effort was put in to maximise the number of grasshoppers marked and frequency of recapture searches. To increase the data set of recaptures, a higher density of individuals needs to be marked and an increase in search efforts. Having multiple observers to cover the area being searched would have greatly improved the chances of re-sighting marked individuals.

RFID tagging is a promising method for monitoring this species if the technical difficulties can be overcome. Attaching an RFID tag to the pronotum of a grasshopper did involve a longer handling period and was more cumbersome to achieve in the field than CMR due to the various equipment needed (glue, RFID tag, receiver). With the current model of RFID receiver, the 'hopper wand' must pass within 20–30 cm of the tagged grasshopper to detect it. This elicits a jump response from the grasshopper, unless it is hiding underneath rocks. Searching for *B. robustus* using the slow-walk technique and CMR would therefore result in a similar chance of finding *B. robustus* in the habitat, if searches are conducted in optimal conditions (no cloud cover). We have no way of knowing if marked individuals were not re-sighted because they were hiding beneath rocks, or because of their cryptic appearance and therefore if using RFID would have resulted in higher recapture rates as RFID tags can be detected beneath rocks and in poor weather. Although the 'hopper wand' beeps when a tag is detected, the observer must still pursue the grasshopper and use the small handheld receiver the record which numbered tag has been located. Therefore, this part of the methods is not made faster by using RFID technology.

Chapter 4

Determining the habitat requirements for *Brachaspis robustus*

4. 1 Introduction

The IUCN (2013) translocation guidelines emphasise the importance of understanding biotic and abiotic habitat requirements for all life stages of a species (IUCN, 2013). Ensuring habitat suitability is fundamental to successful fauna translocations (Kemp, Norbury, Groenewegen, & Comer, 2015). Habitat suitability includes all resources and conditions that impact a species survival and reproduction (Hall, Krausman, & Morrison, 1997).

The objective of this chapter was to gather information on the microhabitat preferences of *B. robustus* to inform what physical characteristics should be looked for when selecting suitable translocation sites. This is particularly important because *B. robustus* is a habitat specialist. There is limited detailed knowledge on what characteristics *B. robustus* requires or prefers within the braided river habitat to which it is restricted. Vegetation coverage is thought to be a major influence *on B. robustus* presence as it reduces open areas for grasshopper basking and may provide habitat for introduced predators. Rock size and composition may also affect *B. robustus* survival as it has been suggested that sand/small pebbles are more suitable for egg laying. Rock size also influences the amount of refuges from adverse environmental conditions and predators (Schori, 2020).

A good example of the complexities of habitat matching when translocating invertebrates is the mahoenui giant wētā (*Deinacrida mahoenui*). Habitat destruction and pressure from introduced predators have greatly restricted the mahoenui giant wētā (*Deinacrida mahoenui*). This species was first discovered in remnant patches of tawa (*Beilschmiedia tawa*) forest in the North Island of New Zealand, however, only a few individuals were found in the forest (Sherley & Hayes, 1993). The wētā were eventually no longer found in the remnant forest patches but were found to be inhabiting gorse that was gradually taking over the farmland. The majority of the mahoenui wētā habitat in the tawa forest was destroyed through clearing of the land, and what was left of the understory was trampled by stock. The nearby gorse stands provided habitat and protection from predators and sheltered ground where wētā could lay their eggs (Sherley & Hayes, 1993). Since its initial discovery, the mahoenui giant wētā has been translocated many times. However, the receiving habitat must either include gorse, to provide protection to the wētā from mammalian predators, or be free of mammalian predators to ensure its survival (Watts & Thornburrow, 2009).

An extensive survey of braided river vegetation composition in the Upper Waitaki Basin, part of the Mackenzie Basin, has shown a wide range of variation. The variation results from differing responses of vegetation communities to a combination of anthropogenic influence, environmental factors, and natural disturbance history (Woolmore, 2011). Therefore, it is also important to understand if any specific vegetation communities are characteristic of exists at the sites currently occupied by *B. robustus*. This information could then be used to find a similar habitat as translocation sites for this species.

While Snowy River was not included in the vegetation survey by Woolmore (2011), the Ōhau River site was. Woolmore (2011) classified the vegetation communities in the Upper Waitaki braided rivers into 11 different community types. By their definitions, communities 4 and 5 are found on the Ōhau River. Community 5 is comprised of 214 plant species; 109 (51%) of the species are exotic. The vegetative cover makes up 40% of total ground cover, and the majority of the plants are <10 cm tall. Sweet-briar (*Rosa rubiginosa*), and introduced woody weed, often over tops the ground vegetation. Community type 4 comprises primarily bare rock, with sparse vegetation making up 2% of ground cover. Images of the most common plant species found and both Snowy River and Ōhau River are attached in Appendix 1. Snowy River is a relatively highly modified community, similar to Ōhau River in having a high proportion of introduced weeds amongst the vegetated areas (e.g. Viper's bugloss (*Echium vulgare*) and sweet briar. These two plants species grow relatively tall and shade the understory seasonally when leaf growth increases or flowering. Some native mosses, lichens and herbaceous plants occur, but there are no old, stable surfaces with a higher cover of native plants seen on the undisturbed rivers like the Tasman.

4.1 Objectives

The research objectives of this study were as follows:

1) Determine vegetation composition at sites currently occupied by *B. robustus* and if the grasshoppers are limited by any vegetation.
2) Determine the amount and particle size of bare ground at *B. robustus* sites.
3) Assess microhabitat preferences of *B. robustus*.
4) Quantify any differences in habitat occupied by males and females.

In each case, I attempted to assess how distinctive the conditions where *B. robustus* are found were from vegetation composition in the wider habitat where *B. robustus* were not found.

4.2 Methods

4.1.1 Quantifying microhabitat used by *B. robustus*

To determine the microhabitat used by *B. robustus*, individual grasshoppers were first located by systematically searching the habitat at a slow walk. I walked with my front leg held low to the ground and swept it back and forwards. The motion of the sweeping leg is to elicit a jump response from the grasshoppers to enable visual detection (Schori, 2020). When a grasshopper was sighted, a coloured rock was placed on the ground where it was first seen. The grasshopper was captured, measured, and marked with a unique code to be identified if observed again later. Body length, hind femur length and sex were recorded for each grasshopper caught. Nymphs with body length <9 mm were not caught in this study due to the risk of injury to the young grasshoppers during handling. A 1 m^2 quadrat was placed on the ground centred on the coloured rock where the grasshopper was first found. The coloured rock was removed and the quadrat was photographed. A small white label with the grasshoppers' unique code was placed inside the quadrat so that each photograph could be linked to a particular grasshopper. The photograph number on the camera was also recorded with the measurements taken of the grasshopper. This data was collected between the 25th of January and the 20th of March, 2020. A total of 24 days were spent searching for *B. robustus* at both sites. One hundred thirty-nine photographs of habitat occupied by *B. robustus* were taken.

A modified reconnaissance plot description (RECCE), described by Allen (1992), was used to quantify ground cover in each 1 m^2 quadrat photograph after cropping the images so that each photograph was the same size and placing a grid over the image to make categorising easier (Figure 4.1). The REECE description created for this project included the percentage of rock, moss, lichen, sand, and vascular plant cover, such that the sum of all categories present added up to 100%. For the rock category, there were three different size classes; 1–10 cm (small), 11–30 cm (medium) , >30 cm (large). Included within the sand category were any other fine-grained substrates such as soil. The vascular plant category was further split into woody plants, mat-forming plants, grasses, and herbaceous plants. These subcategories added up to equal the percentage of 'Rock' or 'Vascular' recorded for that quadrat. The mean top height of woody plants was recorded if the average was over 15 cm tall. If the average height of plants within a quadrat was higher than 2 cm but lower than 15 cm, the mean top height was recorded as >15 cm.

As *B. robustus* are understood to be generalist herbivores, identifying plants such as grasses and small herbs to species level was not considered imperative for this study. To enable the efficient collection of data within the time restrictions of this project, plant species IDs were therefore not routinely recorded. However, brief notes were made of what species the woody plants were and if native or invasive. When collecting the data in the field, I recorded if vegetation had a mean height of below 15 cm but higher than 2 cm. To be able to analyse this data statistically, the value <15 was changed to the mean of 7.5.

Figure 4.1. Example of the digital grid laid over an image of a 1 m^2 quadrat photographed in the field. The metal quadrat is just visible around the edges of the image.

Determining the wider habitat available to *B. robustus*

To determine the composition of the ground cover available to *B. robustus* at Snowy and Ōhau Rivers, a randomised transect survey was conducted. A 600 m long-baseline was first established at Snowy River, this ran parallel to the Hakataramea PassFigure 4.2 Layout of ground cover transects at Snowy River. Road (Figure 4.2), and at the lower Ōhau River, it was run between a small pond and patch of sweet-briar shrub (Figure 4.3). The baseline was then divided into six 100 m segments at both sites. Within each segment, a 200m transect was laid out perpendicular to the baseline. The starting point for each transect was determined using a random number generator to produce a number between 1 and 100, corresponding to the distance along the segment. Each transect included eight 5 x 5 m (25 m^2) quadrats in which ground cover composition was quantified using the REECE categories described above. The first quadrat on each transect was located at a random starting point between 1 and 15 m from the baseline (0 m). Subsequent quadrats were spaced out at 25 m intervals from the first. A compass bearing was used to ensure transects were straight. Quadrats were established from the bottom right-hand corner. The data was collected in November 2020 over five days. A combined total of 96 quadrats were surveyed at the Snowy (n = 48) and Ōhau (n = 48) Rivers.

Transects intersected areas of the river beds where grasshoppers had been observed, but none of the quadrats overlapped with exact locations of *B. robustus* recordings and subsequent 1 m^2 ground cover sampling. This was confirmed by loading all GPS points of observed grasshoppers and randomly placed quadrats into QGIS.

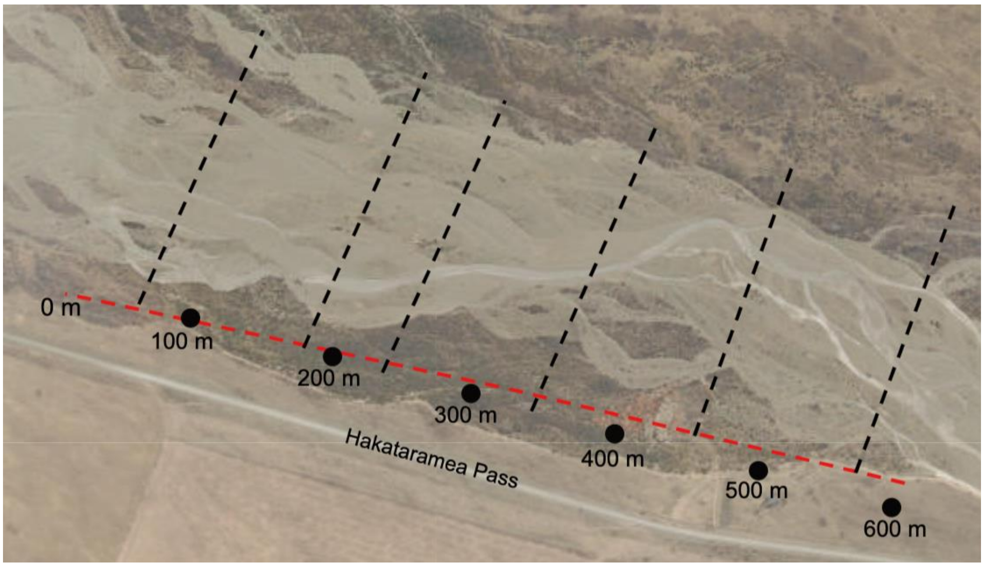


Figure 4.2 Layout of ground cover transects at Snowy River. Ground cover was assessed in eight 5 x 5 m quadrats randomly placed along six transects (black lines) running perpendicular to a 600 m baseline (red line).

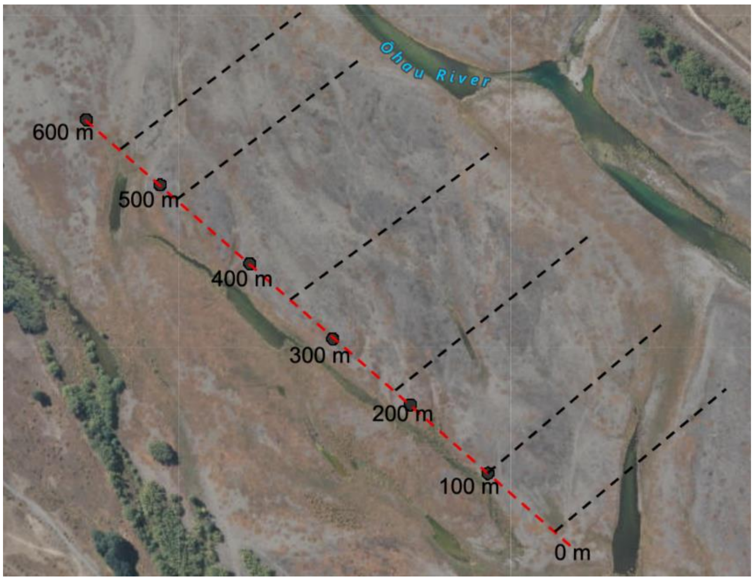


Figure 4.3 Layout of ground cover transects at Ōhau River. Ground cover was assessed in eight 5 x 5 m quadrats randomly placed along six transects (black lines) running perpendicular to a 600 m baseline (red line).

4.2.3 Data analysis

Grasshopper presence and ground cover

All data were analysed in R version 4.0.5 (R Core Team, 2021). Because this is a multivariate dataset, all 13 habitat variables were analysed by Principal Component Analysis (PCA) using packages *ggfortify* version 0.4.12 (Horikoshi & Tang, 2016) and *factoextra* version 1.0.7 (Kassambara & Mundt, 2020). Before conducting the PCA, correlation coefficients between all habitat variables were inspected with a correlation matrix constructed using package *corrplot* version 0.90 (Wei & Simko, 2021).

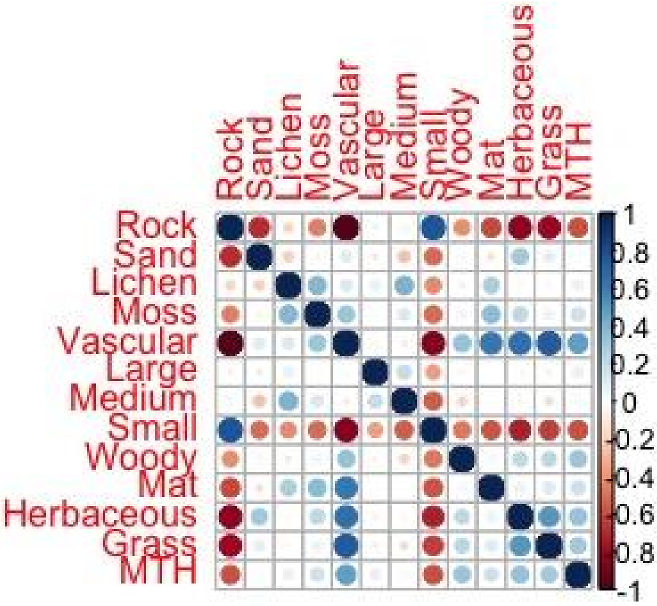


Figure 4.4 Correlation matrix plot of grasshopper habitat ground cover categories including the umbrella categories 'vascular' plants and 'rock' as well as the subcategories within (rock = 'small' rocks, 'medium' rocks, 'large' rocks; 'vascular' = 'woody' plants, 'mat' forming plants, 'grass'). MTH = mean top height of vegetation. Positive correlations (0 to 1) between ground cover variables are displayed in blue colours, and negative correlations (0 to -1) are displayed in red colours. Additionally, the larger the dot size, the strong the correlation between variables.

Based on the correlation matrix (Figure 4.4), we removed the 'rock', 'vascular', and 'mean top height' (MTH) variables, and the final PCA analysis was conducted with ten habitat variables. The rock size classes 'small', 'medium' and 'large' combined equate to' rock' 's total % cover value. Similarly, 'woody', 'mat', 'herbaceous' and 'grass' all added up to equal the value of 'vascular'. This created relationships between 'rock' and 'vascular' and the subcategories within (Figure 4.4) and as such the decision was taken to remove those variables from further analysis. Mean top height was also removed as it related to the variable 'woody'. Woody plant species such as sweet-briar and some coprosmas were usually the only plants to contribute to a MTH value as only plants >15 cm tall on average were included.

The statistical significance of the habitat variables on grasshopper presence was then investigated by MANOVA followed by univariate ANOVA for each habitat variable using the package stats version 4.0.5. Where ANOVA showed a significant effect on grasshopper presence, pairwise comparisons were undertaken using package *emmeans* version 1.6.2-1 (Lenth, 2021). Boxplots of habitat variable values were constructed using packages *ggsci* version 2.9 (Xiao, 2018) and *ggpubr* version 0.4.0 (Kassambara, 2020).

Male vs Female habitat preferences

Variances in male and female *B. robustus* habitat use were also analysed using the 1 m quadrat ground cover data from Snowy and Ōhau Rivers, where individual *B. robustus* were caught (or re-sighted if paint code could be seen without handling) and recorded. The main relationship I was interested in was between adult females, sand, and small rocks. Again, a PCA followed by a MANOVA was completed for this data set with each grasshopper identified as male or female.

4.3 Results

Microhabitat occupied by *B. robustus*

Microhabitat data was collected for a total of 139 *B. robustus* observations at Snowy and Ōhau Rivers (**Table 4.1**), which including multiple sightings of some individuals on different days (see Chapter 3). Most grasshoppers were found at Snowy River (n = 122) as the population at Ōhau River was sparse and considerably smaller (n = 16), and as a result less ground cover data within 1 m of a grasshopper was collected at Ōhau River, compared to Snowy River. The ground cover data collected from Snowy River will therefore dominate my results.

Only seven adult grasshoppers were found across both sites; five were females and two males. More females, both adult and sub-adult, were captured (n = 94) than males (n = 46). Adult female *B. robustus* have a hind femur measurement of >15 mm, and males of >9 mm. Female *B. robustus* are considered in the penultimate life stage (sub-adult) when their hind femur measures between 12 mm–14 mm and sub-adult males when the hind femur measures 7–8 mm (J. Schori, personal communication, November 7, 2021).

Table 4.1 Demographics of *B. robustus* for which microhabitat data was collected at Snowy and Ōhau Rivers. Recaptures refer to grasshoppers that were marked and recaptured at a later date. Microhabitat quadrats were analysed for both captures and recaptures.

Site	Sex	Adult	Sub-adult	Immature	Recapture	Total 1 m^2 quadrats (including recaptures)
Snowy River	Male	2	13	25	5	40
	Female	5	4	60	9	69
	Total	7	17	85	14	123
Ōhau River	Male	0	0	2	0	2
	Female	0	0	10	4	14
	Total	0	0	12	4	16

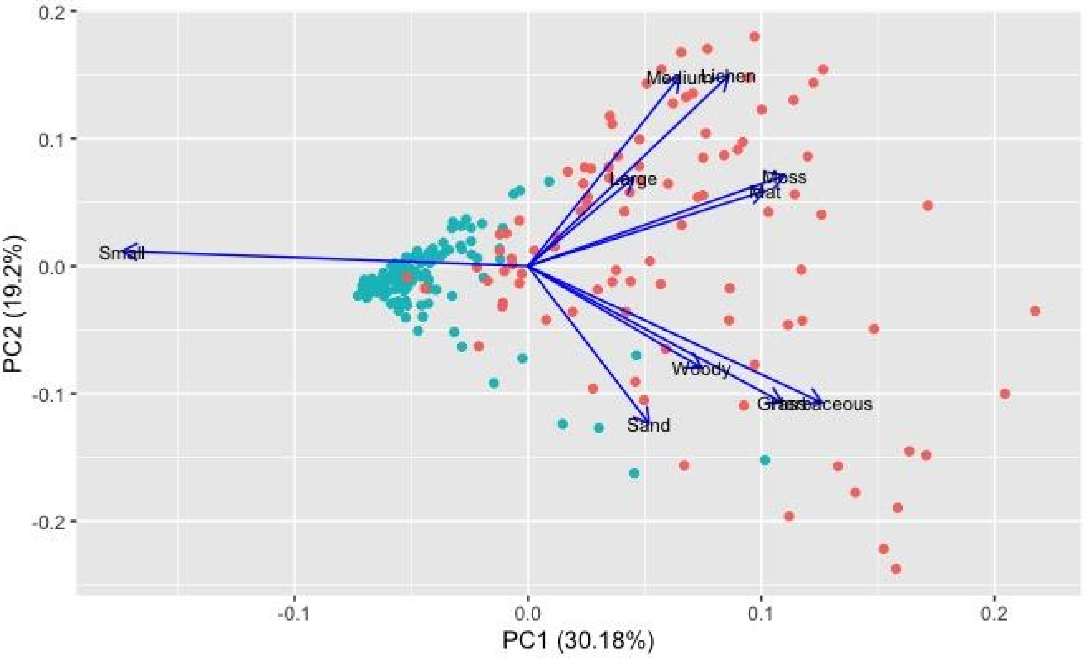
Table 4.2 Importance of components table from summary Principal component test.

	PC1	PC2	PC3	PC4
Standard deviation	1.7371	1.3856	1.0695	0.98397
Proportion of Variance	0.3018	0.192	0.1144	0.09682
Cumulative Proportion	0.3018	0.4937	0.6081	0.70493

The total proportion of variance explained by the first four principal components is 70.5%. PC1 (30.2%) and PC2 (19.2%) explained a cumulative total of 49.4% of the variation in grasshopper presence, PC3 explained 11.4%, and PC4 9% (Table 4.2). Small-sized rocks (1–10 cm) and herbaceous plants were highly important on PC1 as an indicator of where grasshoppers were found (red dots). Lichen, medium-sized rocks (11–30 cm), and sand are the most important on PC2 and are more strongly associated with where grasshoppers were not found (blue dots) (Figure 4.5, Table 4.3 Eigenvector matrix showing the correlation between the Principal Components (dependent variables or rows) and the input image bands (independent variables or columns). Values that are closer to 1 or -1 are more important on that PC. Values over 0.6 or -0.6 are highlighted as having a significant contribution towards the PCs.Table 4.3).

Table 4.3 Eigenvector matrix showing the correlation between the Principal Components (dependent variables or rows) and the input image bands (independent variables or columns). Values that are closer to 1 or -1 are more important on that PC. Values over 0.6 or -0.6 are highlighted as having a significant contribution towards the PCs.

Variable	PC1	PC2	PC3	PC4
Sand	0.2815549	-0.5342317	-0.3318785	**-0.585373858**
Lichen	0.4671724	**0.64709806**	0.12315178	-0.127180414
Moss	0.5974774	0.30747546	0.38584411	-0.189926324
Small rocks	**-0.9443487**	0.05051654	0.20918816	0.023843353
Medium rocks	0.3528337	**0.64430776**	-0.4213341	0.003506359
Large rocks	0.2460286	0.30038388	**-0.6277772**	0.325995233
Herbaceous plants	**0.6842817**	-0.4632492	-0.0785679	-0.09011351
Woody plants	0.4040267	-0.3452583	0.14934996	**0.654084272**
Mat forming plants	0.5521104	0.2555319	0.4702611	-0.043256037
Grass	0.5915089	-0.4627607	0.06731841	0.169159798



Blue = Habitat within
1 m of grasshopper
Red = Wider habitat

Figure 4.5 Biplot showing distribution of habitat ground cover variables. Blue dots = Habitat within 1 m of an observed grasshopper (grasshopper presence). Red dots = randomly sampled quadrats from the wider riverbed environment where grasshoppers were not found (Grasshopper habitat but no grasshoppers detected) on the PC1 and PC2.

A MANOVA (P-value <0.001), indicated that there were significant differences between the composition of ground cover variables found within 1 m of a grasshopper and ground cover present in the same habitat but where grasshoppers were not found. Univariate ANOVA was then subsequently carried out for each explanatory variable (Table 4.4). Results from the ANOVAs show that the median of small rocks within 1 m of grasshoppers (84%) was significantly higher than the median of small rock cover in the wider habitat (35%). The median of medium sized rocks in habitat within 1 m of grasshoppers (8%) was lower than that in the wider habitat (20%). The median of lichen in the wider habitat was ten, whereas the median of lichen within 1 m of grasshoppers was zero, indicating the grasshoppers were never found on lichen but on high proportions of small rocks.

Table 4.4 Lower and upper quartiles, median and P-values for each ground cover variable value from univariate ANOVA. Small rocks = 1 – 10 cm, medium rocks = 11 – 30 cm, large rocks = >30 cm. * indicates the significance of the P-value.**

	Wider habitat available			**Habitat within 1 m of an observed grasshopper**			
Variable	**Lower quartile**	**Upper quartile**	**Median**	**Lower quartile**	**Upper quartile**	**Median**	**P-value**
Small rocks	17.5	50.5	35	77	89	84	<0.001***
Herbaceous plants	2	8	4	0	2	1	<0.001***
Medium rocks	14	37	20	4	15	8	<0.001***
Lichen	0	35	10	0	0	0	<0.001***
Sand	1	10	3	0	2	1	0.0375 *
Moss	0	1	0	0	0	0	<0.001***
Large rocks	0	5	2	0	3	1	<0.001***
Woody plants	0	3	0	0	0	0	0.0179 *
Mat forming plants	2	15	7	0	1	0.5	<0.001***
Grass	0.75	8	2	0	0	0	<0.001***

The results from ground cover surveys of the wider habitat available to grasshoppers show that moss and woody vegetation were the least abundant, with medians of zero. Lichen was much more abundant at the Ōhau site than Snowy River; 30.77% of plots within 1 m of a grasshopper had lichen cover over 20%. Due to the limited number of grasshopper observations at Ōhau (n= 14), there was a low output value for the lichen cover within 1 m of a grasshopper. No lichen was found in any quadrat where a grasshopper was observed (n=123) at Snowy River. In comparison, there was lichen in every quadrat where a grasshopper was observed at Ōhau River. This demonstrates how the geophysical differences between the two sites has somewhat skewed the data.

Whilst the ANOVAs indicated significant differences for all habitat variables, the most important ground cover variable (derived from the PCA and ANOVA analysis) explaining the presence of *B. robustus* was small rocks (Fig. 4.6). In this study, 89.9% (125 out of 139) of *B. robustus* were found in quadrats with >50% cover of small rocks, while the mean cover of small rocks across the randomly placed quadrats in the wider habitat was only 34.9% (s.e.m. 1.94%). The mean cover of small rocks within 1 m of a grasshopper was 80.1% (s.e.m. 1.65%)

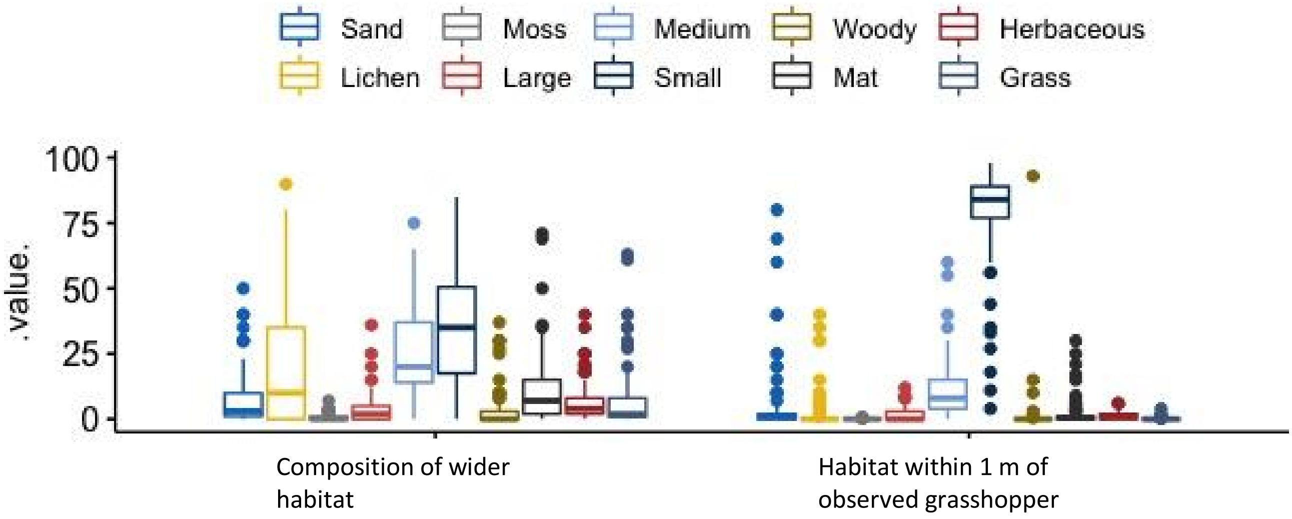


Figure 4.6 Box plot showing ground cover abundances (%) where *B. robustus* were found and the composition of ground cover available in the wider habitat at Snowy and Ōhau Rivers.

Grasshoppers were least likely to be found in microhabitats with moss, lichen, large rocks, or any kind of vegetation. The median for lichen, moss, large rocks, woody vegetation, and grass within 1 m of a grasshopper were all zero (Table 4.4). Common woody plant species found at both Snowy and Ōhau Rivers were sweet briar (*Rosa rubiginosa*) and matagouri (*Discaria toumatou*). *Raoulia* spp. were common mat-forming plants, *Echium vulgare* and *Hieracium* spp. were frequently observed herbaceous plants. Mat forming vegetation had a median of 0.5, and herbaceous vegetation had a median of one. Higher ground cover percentages of these variables were available in the wider habitat, but still significantly less than small and medium rocks. The above Table (Table 4.4) clearly demonstrates that *B. robustus* were found in habitats with a higher percentage of small rock (median 84%) than any other ground cover type.

Height of vegetation found within 1 m of a grasshopper vs in the wider habitat

Only three (2.2% of total) out of 139 grasshopper observations were found within 1 m of vegetation over 2 cm tall. The mean top height of vegetation within 1 m of two grasshoppers at Snowy River was 7.5 cm, and 35 cm respectively and was 7.5 cm within 1 m of one grasshopper at Ōhau River. The 25 m^2 quadrats sampled in the wider habitat had much taller vegetation on average compared to the $1m^2$ quadrats where grasshoppers were found. My data shows that 31.1% of quadrats from the wider habitat contained vegetation with a mean top height of >20 cm, and 14.9% had a mean top height of below 20 cm but higher than 2 cm. The mean value of vegetation mean top height in the wider habitat at Snowy and Ōhau Rivers was 26 and 14.6 respectively.

Male vs Female habitat preferences

When looking at adults and juveniles combined, there are some minor differences between where male and female *B. robustus* were found at Snowy and Ōhau Rivers, but in general, they were found in habitats with fairly similar ground cover compositions. There was no strong pattern dividing male and female ground cover preferences. Some females were slightly affiliated with sand on PC2; however, the P-value generated from the MANOVA was not significant (0.411), meaning the ground cover survey produced similar values for male and female *B. robustus;* this is also demonstrated in the following box plot (Figure 4.8 Box plot showing that small rocks are important for both male and female *B. robustus*. A few females have an affiliation with sand.Figure 4.8).

Overall, the total proportion of variation in male and female habitat preferences explained on PC1 and PC2 was 42.79% (Table 4.5). PC1 explained 22.34% of the variation, and PC2 explained 20.36%. The cumulative proportion of variation explained on the first four PCs was 67.21%.

Table 4.5 Importance of components Table from Principal Component Analysis output.

	PC1	PC2	PC3	PC4
Standard deviation	1.4975	1.4269	1.1533	1.0549
Proportion of Variance	0.2243	0.2036	0.133	0.1113
Cumulative Proportion	0.2243	0.4279	0.5609	0.6721

Results from the PCA test show that lichen, moss, and mat-forming plants are most important on PC1, with sand and small rocks most important on PC2 (Table 4.6).

Table 4.6 PC loadings table showing the explanatory variables contribution to the PCs. Values over 0.6 or -0.6 are highlighted as having a significant contribution towards the principal components.

Variable	PC1	PC2	PC3	PC4
Sand	0.10371260	**-0.766016382**	-0.17882928	0.14818223
Lichen	**-0.83197840**	-0.036787063	0.13416173	0.10220516
Moss	**-0.90151163**	-0.001346152	0.07680217	0.003108138
Small rocks	-0.01227518	**0.894723686**	-0.42770790	-0.047867971
Medium rocks	0.10939660	-0.041876838	**0.72175613**	0.588818655
Large rocks	0.21155729	-0.252153329	-0.04328369	-0.318495549
Herbaceous plants	0.23189668	-0.505772146	-0.40171591	0.265243341
Woody plants	0.04622142	-0.327974547	0.44238780	**-0.742201809**
Mat forming plants	**-0.75375857**	-0.211918324	-0.13671739	-0.077714235
Grass	-0.21412809	-0.416787772	-0.43903897	0.0510785

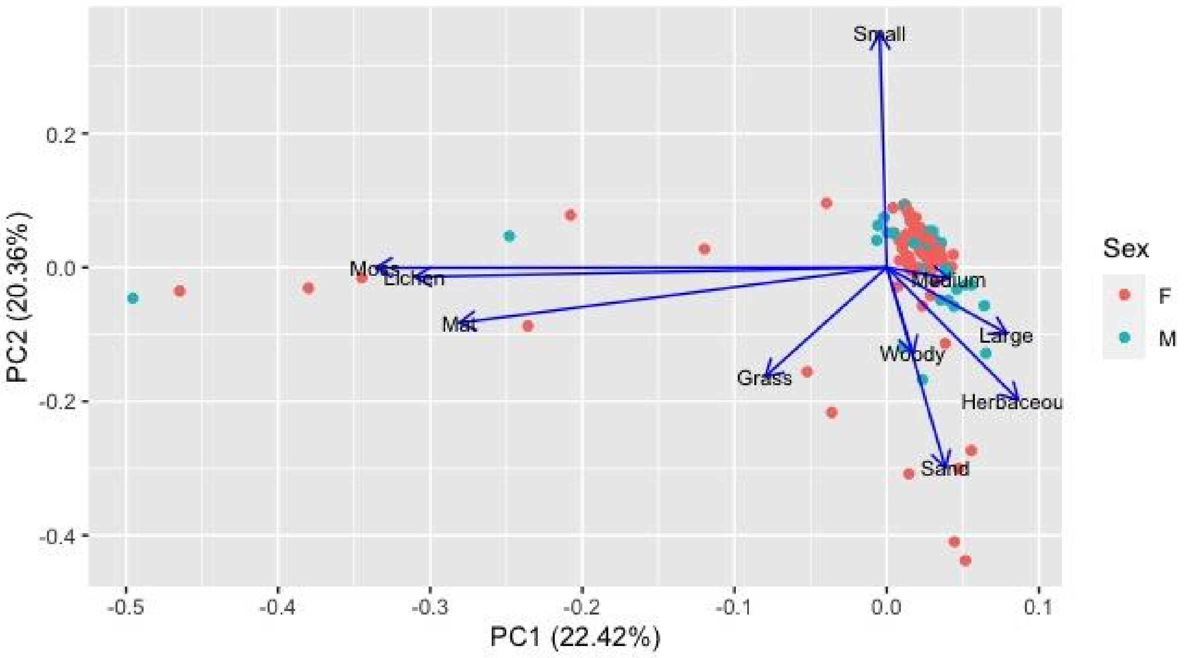


Figure 4.7 Biplot showing the distribution of explanatory variables between male and female *B. robustus*. This data set includes both adult and juvenile grasshoppers.

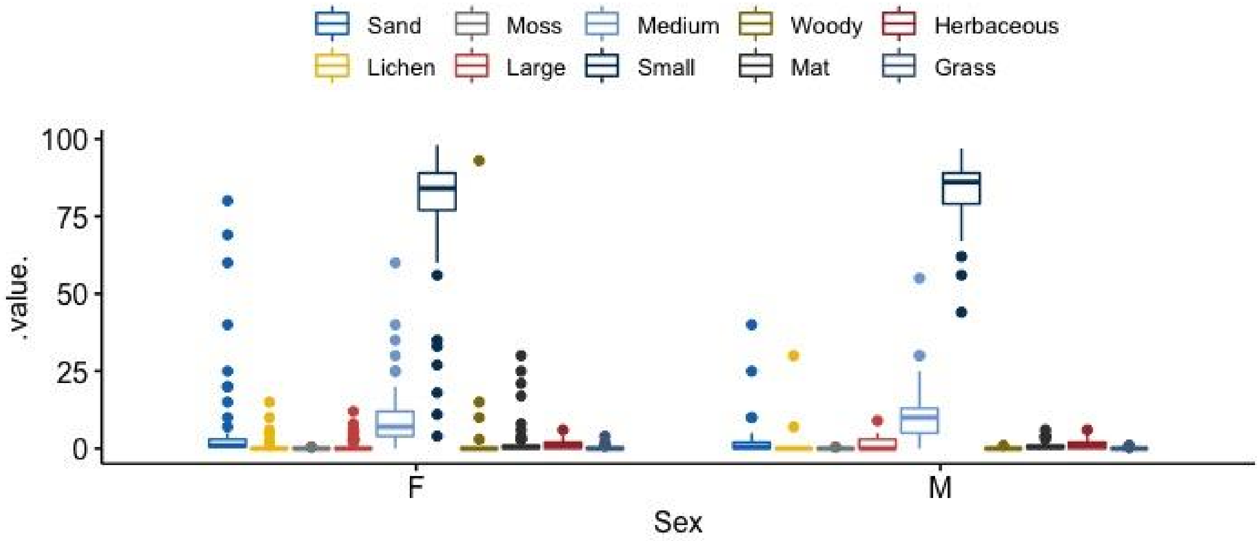


Figure 4.8 Box plot showing that small rocks are important for both male and female *B. robustus*. A few females have an affiliation with sand.

Adult female and sand relationship

In total, 92 female grasshoppers were recorded across Snowy and Ōhau Rivers (Table 4.1); 11 of these were recaptures of the same grasshopper. Out of the 79 individual female grasshoppers caught, five were adults, all found at Snowy River. Out of those five, the habitat within 1 m of three comprised >20% sand. In the wider habitat at Snowy River, the mean cover of sand was 11.5%. No adult female or male *B. robustus* were recorded at the Ōhau River site. To increase sample size sufficiently to test for a correlation between near-reproductive females and sand as a possible oviposition habitat, three sub-adult females were included in the data set (Table 4.1). A univariate ANOVA was completed for sand and small rocks to test for a relationship to sex (male or female grasshopper) (Table 4.8). Although this is not statistically significant, we can see sand has the third lowest P-value for all the habitat variables (Table 4.7).

Table 4.7 P-value represent the significance of the relationships between ground cover variables and grasshopper sex.

Variable	P-value
Sand	0.2506
Lichen	0.6762
Moss	0.9166
Large	0.602
Medium	0.5333
Small	0.2687
Woody	0.3534
Mat	0.1792
Herbaceous	0.1485
Grass	0.3624

Table 4.8 All adult and sub-adult female and male grasshoppers from Snowy and Ōhau Rivers. The percentage of sand and small rock the grasshoppers were found in, within 1 m^2 and the percentage of sand and small rocks in the wider habitat.

	Female	Male	Female and male	Wider habitat
Mean % sand	18.3	5.7	10.1	11.5
Mean % small rock	64.4	82.5	76.2	34.6
Mean % sand and small rock combined	41.3	44.1	43.1	21

4.2 Discussion

4.2.1 The importance of small rocks

I hypothesised that rock size would influence *B. robustus* microhabitat selection. This is because *B. robustus* are ectothermic and bask in sunlight to increase metabolic rate (Forsman, 1999), and small rocks heat up faster due to their small volume. In contrast, larger rocks take longer to reach maximum temperature but retain heat for longer (Huey et al., 1989). Therefore, the availability of basking sites is highly important. A behavioural study conducted on this species by Schori (2020) found that 78.8% of the grasshoppers observed were basking, 13.4% were found mating, 6.9% were observed moving (walking, jumping), and 1% were seen eating (Schori, 2020).

My results showed clearly that *B. robustus* prefer a microhabitat dominated by small rocks (1–10 cm) rather than any other ground cover variable tested. Small rocks were also the most prevalent in the wider habitat, which may account for why *B. robustus* were most likely to be found on them. Mean cover of small rocks across the randomly placed quadrats in the wider habitat was only 34.9% (s.e.m.

1.94%). The mean cover of small rocks within 1 m of a grasshopper was 80.1% (s.e.m. 1.65%). I hypothesis that *B. robustus* prefer small rocks over sand, soil, or larger rocks because small rocks enable more effortless mobility for this species. Female *B. robustus* can grow to ~38 mm long, and males of the species about half that length. It is thought that this species only jumps when it feels threatened, otherwise it typically crawls to move about the habitat. Therefore, rocks with a smaller mass would be easier to navigate. This species uses rocks of the braided rivers to bask upon and as shelter from predators. Observations made during the fieldwork on this study found that when attempting to capture a *B. robustus* individual, it was extremely difficult to find if it decided to hide underneath rocks. Often, grasshoppers would crawl down through the spaces of rocks where they could only be located by moving many rocks and intensive searching. This behaviour, combined with their cryptic colouring, would likely be a very effective predator avoidance technique. As mentioned above, spaces between and underneath rocks may provide shelter from extreme daytime heat and moisture in the form of condensation during the night (White, 1994). Cycles of condensation and evaporation in soil particles underneath rocks decreases temperatures beneath rocks (Huey et al., 1989). Considering this, larger rocks would provide more protection from predators, but they do not heat up as fast as small rocks. Perhaps the ‘small’ class of rocks used in this study provides a happy medium between protection and warmth, or that the grasshoppers were easier to detect on small rocks.

The exact thermal threshold for *B. robustus* is not known. Willott (1997) examined the thermal thresholds of four different grasshopper species in England. They observed that all four species displayed ‘shade-seeking behaviour’ above 40 °C and increased escaped reactions above 44 °C. Even going a few degrees above the temperature optimum can result in severe physiological harm or death (Willott, 1997). Results from a behavioural study by Schori (2020) showed that monitoring conditions optimal for locating *B. robustus* through jump response were fine, warm conditions above 14 °C. The fieldwork for my study was conducted in a range of temperature and weather conditions, including air temperature (at 1 m) of 14 °C to 30 °C, sunny, overcast, strong winds and rain. Grasshoppers were recorded in all these weather conditions.

4.2.2 Relationship between vegetation height and *B. robustus*

Vegetation cover and height is thought to be an important factor in microhabitat selection for *B. robustus* as plants that cover large areas of rock reduce basking sites and the ability for grasshoppers to hide under rocks to avoid predation. Vegetation may also reduce oviposition sites for egg-laying. Tall vegetation also provides cover and habitat for mammalian predators (Norbury, 2001). Tall or dense vegetation has anecdotally been seen to hinder *B. robustus* movement. When a grasshopper of this species jumps, it appears random, and they often land clumsily. If a grasshopper is jumping to avoid predation, it may get caught up in vegetation in its jump path. For these reasons, the mean top height of vegetation within 1 m of each grasshopper was recorded. The results confirmed the hypothesis that *B. robustus* would select against microhabitat sites with adjacent tall or dense vegetation as the average height of vegetation within 1 m of grasshoppers found in this study was only 38 mm compared with xx in the wider environment. Out of 139 grasshoppers observations, only three were found within 1 m of vegetation taller than 2 cm. There were two grasshoppers at Snowy River; one was found within 1 m of vegetation with a mean top height of 7.5 cm, the other 35 cm. The third grasshopper was at Ōhau River and was found within 1 m of vegetation with a mean top height of 7.5 cm.

4.2.3 Plant species and diet of *B. robustus*

It is accepted that *B. robustus* are generalist herbivores and may consume many native and introduced plant species. Faecal analysis of *B. robustus* in 1994 showed that the diet of this species was mainly comprised of unidentified herbaceous species, the grasses *Elymus rectisetus* and *Poa pratensis*, *Achillea millifolium*, and unidentified mosses and lichens (White, 1994). However, the prevalence of plant species in the habitat may not represent the diet of *B. robustus*. As White (1994) showed, the presence of *A. millifolium* in Mackenzie Basin was discovered through the faecal analysis of *B. robustus*, rather than vegetation analysis of the landscape (White, 1994). Therefore, the low frequencies of mosses, lichens, and other vegetation within my study sites do not indicate the frequency of these plant species in the diet of *B. robustus*.

The Mackenzie Basin is a dry, arid landscape with harsh weather conditions. The average annual rainfall for the basin is 600 ml, and there can be long periods of drought in the summer (Macara, 2016). White (1994) suggests that *B. robustus* may acquire the necessary moisture to survive the Mackenzie Basin extreme heat through the consumption of mosses and lichens as these plants absorb moisture during the night (White, 1994).

When held in captivity as part of a behavioural study by Schori (2020), *B. robustus* grasshoppers were fed a combination of dandelion and daisy flowers, and various seedlings of leafy vegetables (spinach, cos lettuce, carrot), exotic weeds (yarrow, borage, *Phacelia sp*., herbal-ley mix, forget-me-not, dandelion, clover, hawkweed). *Pinus radiata* pollen was also provided as a source of protein to the grasshoppers when dandelion flowers were not available (Schori, 2020).

The Ōhau River site had high proportions of lichen in the wider habitat, yet the grasshoppers found there were not located near areas of high lichen density. Very low levels of lichen were recorded in the wider habitat at Snowy River, so, understandably, grasshoppers found at Snowy River were not recorded near lichen. Low proportions of moss were recorded at both study sites within the wider habitat, and no grasshoppers were found within 1 m of moss.

As seen through the vegetation analysis of my study and other vegetation analyses of Mackenzie Basin, mosses and lichens are not particularly abundant in the main riverbed channels (White, 1994). Native lichens are seen to be associated with moisture-deficits in soil and atmosphere (Woolmore, 2011).

4.2.4 Male and female habitat preferences

Overall, male and female *B. robustus* of all ages combined were found occupying similar microhabitats. Female grasshoppers occupied areas with slightly more variable ground cover composition compared to males, but both sexes preferred to be on a substrate with high proportions of bare rock and minimal vegetation and were rarely found in association with high levels of moss, lichen and mat-forming plants. This supports the hypothesis that this species prefers younger, more disturbed riverbeds as the vegetation types mentioned above are often associated with more stable, undisturbed habitats (F. Thorsen, 2010, unpublished data).

Metamorphological species such as *B. robustus* usually have different habitat requirements at different life stages. When looking just at adult and sub-adult male and female grasshoppers, microhabitat occupied was also dominated by a high proportion of small rocks (mean 79.0% and 82.3%). Females of this species are thought to require some form of gravel or sand substrate to lay their egg pods. From previous studies, it was thought that they prefer sand or fine pebbles (Schori, 2020). I attempted to gather further information on the microhabitat preferences of adult female *B. robustus* that are at egg-laying age, however, the timing of my study did not coincide with peak adult female presence, which occurs in November–early December, because of technical problems that delayed sampling. The majority of *B. robustus* grasshoppers that I recorded for this study were therefore juveniles. I only recorded five adult females and two adult males. I did, however, record many more sub-adult males (n= 13) than sub-adult females (n= 3). If I had conducted my fieldwork earlier in the season, I might have collected more valuable data, as adult females are the most biologically informative demographic (Schori et al., 2020a). Across both study sites, adult and sub-adult female grasshoppers were found on a mean of 18.3% sand, whereas adult and sub-adult male *B. robustus* were found on a mean of 5.7% sand.

This small sample size does not have enough evidence to support the hypothesis that adult females require sand or small gravels to lay their eggs in. Conducting field work to coincide with peak adult female emergence (November–early December) would increase the sample size and provide the necessary data on adult female *B. robustus*.

4.2.5 Conclusions

My research at Snowy and Ōhau Rivers has demonstrated there is evidence for microhabitat preferences of *B. robustus*. These results are important as they can be used to inform translocation site selection. This species prefers to inhabit areas with a high proportion of small (1–10 cm) rocks and low levels of vegetation cover. Although moss and lichens have previously been highlighted as potentially important in the diet of *B. robustus*, the grasshoppers in my study often occupied habitats with low cover of these species. Adult females of this species most likely require a fine substrate for egg laying, but limited data collected on this demographic resulted in only a slight affiliation with adult females and sand/small rocks. Using the results from my study, suitable translocation site selection should therefore contain the abiotic factors of high levels of small stone cover and minimal vegetation, as well as considering other factors that contribute to suitable habitat for this species which are discussed in the following chapters.

Chapter 5

Identifying sites that have attributes that meet the criteria for a successful translocation of *B. robustus*.

5.1 Introduction

5.1.1 Habitat requirements of *Brachaspis robustus*

Ensuring habitat suitability and availability is an important factor when planning a translocation of any species (IUCN, 2013). Habitat suitability includes ensuring the biotic and abiotic requirements of the species is met in the receiving habitat of a translocation (IUCN, 2013). Fish and Wildlife Service Habitat Evaluation Procedures in the U.S defined habitat suitability as the potential of a habitat to support a specific species (Kellner, Brawn, & Karr, 1992).

The Chapter 2 literature review identified predation, habitat modification, hybridization and climate change as the key threats towards *B. robustus* that need to be considered in translocation site selection. Chapters 3 and 4 helped to identify and quantify some of the habitat requirements we think *B. robustus* have. This chapter now looks at several possible translocation sites to determine if they have those habitat requirements and also if the threats identified in Chapter 2 can be removed or mitigated.

Results presented in Chapter 4 of this thesis showed that *B. robustus* are most likely found in high proportions of small (1–10 cm) rocks with minimal vegetation. These results can be used to inform what site characteristics should be searched for at any proposed translocation sites. By replicating the methods from Chapter 4 at proposed translocation sites, we can compare the ground cover variables between sites where they are currently found and future translocation sites. We will then be able to recommend which locations will be most suitable for *B. robustus,* based on similarities in ground cover. There are many other factors that contribute to habitat suitability for a species other than ground cover (Richardson et al., 2015), and there are gaps in our knowledge of the detailed ecological and biological requirements for *B. robustus* (Schori, 2020), but some of the other important factors will be addressed herein.

In determining the habitat requirements of this species, conservation managers must also consider the threats to *B. robustus,* what threats the species may face in the future, and what can be done to mitigate them. Important previous research by Schori (2020) informs decisions on how much space, weed control, and predator control must be carried out in the receiving habitat for a successful translocation of *B. robustus* (Schori, 2020). A review of 54 translocations in Australia found that lack of adequate predator control and habitat suitability were the two main contributing factors to failed animal translocations (Sheean, Manning, & Lindenmayer, 2012). Much planning and consideration may go into translocation site selection. However, because many factors contribute to habitat suitability, the translocated species' entire needs may still not be met at the site (Richardson et al., 2015).

5.2 Objectives

The research objectives of this chapter were to:

1) To select a series of potential sites for consideration of suitability for translocating *B. robustus*.

2) To determine which of the potential translocation sites are most similar in habitat to the Snowy and Ōhau *B. robustus* populations, based on the ground cover composition of rocks and vegetation.

3) To determine the risk of predation at potential translocation sites

4) To determine the risk of hybridization with *B. nivalis* at translocation sites.

5.3 Methods

5.3.1 Translocation site selection

Through speaking with conservation managers and experts on *B. robustus* and the Mackenzie Basin braided river area, several potential translocation sites for *B. robustus* were identified. The list of potential translocation sites was then narrowed down based on some of the key biological and ecological requirements known for *B. robustus* using information gathered from the limited past studies on this species (McIver, 2020; Schori, 2020; Schori et al., 2019; White, 1994). Potential translocation sites that were considered at the beginning of this research were: Upper Ōhau River (-44.2655º, 170.0018º), Macaulay River (-43.71809º, 170.5767º), Cass River (-43.8723º, 170.4779º), upper Tasman River (-43.8050º, 170.1369º), and lower Tasman River (-43.8332º, 170.1345º) (Figure 5.5). I assessed each site for suitability against habitat suitability and risk of predation, hybridisation and climate change. Three sites were excluded early on (see results). Intensive habitat and predator surveys detailed below were only undertaken in the Tasman and Cass River sites.

5.3.2 Ground cover composition at proposed translocation sites

The ground cover was surveyed to assess the habitat suitability of potential translocation sites, repeating methods described in Chapter 4 of this thesis. This compares the ground cover composition where grasshoppers were most definitely found (Snowy and Ōhau Rivers), to that of the habitat where they were not found but could be translocated to. The ground cover survey was conducted using REECE plots at two sites on the Tasman River (Upper Tasman and lower Tasman) and one on Cass River. At each river site except Cass River, six transects at 100 m spacing with eight quadrats in each were set out. These methods are described in Chapter 4 of this thesis (Figures 4.2 and 4.3). Cass River only had four transects as the area of open gravels was much smaller (~10 ha) than the other sites. Quadrats were 5 x 5 m, 25 m^2. In total 128 REECE plots were surveyed across the three potential translocation sites and compared to the 96 REECE plots were surveyed within known grasshopper populations in Chapter 4

5.3.3 Predators: Tracking tunnels

Following the Department of Conservation protocol (Gillies & Williams, 2013), three transects, 200 m long and 200 m apart, were placed across the Snowy River and Cass River sites. Each transect included five tracking tunnels with ink cards at 50 m spacings (Figure 5.1). Transects intersected both rocky riverbed habitat and the surrounding vegetated land. Tracking tunnels were baited with peanut butter (targeting rodents) for one night then rabbit meat (targeting hedgehogs and mustelids) for the following four nights. The same methods were replicated in Paterson’s Terrace to compare mammalian predator activity to another site with a known *B. robustus* population. Cards were removed after the peanut butter night and replaced with fresh ones baited with rabbit meat. Footprints were examined and an index created for which predators were present at each of the sites. Two monitoring events were carried out at Snowy river, conducted on 10th–13th February and 14th–20th March 2020. One monitoring event was carried out at Paterson’s Terrace, and this occurred on 12th–18th of March 2020. No tracking tunnel monitoring was conducted at Cass River as DOC already had mammalian predator presence data available from their trapping network.

Figure 5.1 Tracking tunnel transects at Snowy River. Transects began and ended on the vegetated bank either side of the riverbed.

5.3.4 Mesopredators: Artificial retreats

Artificial retreats (ARs) made from Onduline (distributed by Composite, Christchurch NZ) were used to detect the presence of lizards at Snowy and Cass Rivers. The ARs measured 28 cm x 40 cm and were installed at Snowy River on the 21st of September 2020 and Cass River on the 22nd of September 2020. Both sites had six transects at 100 m spacings, and each transect included 10 ARs at 15 m spacings. The ARs were given a placement period of 4 months to let the Onduline weather in. At Snowy River, three transects were placed in the riverbed, and three transects were on one side of the vegetated bank because DOC conservation managers recommended monitoring for lizards on both vegetation and rocky riverbed as lizards often inhabit vegetated areas but may also move into the riverbed. At Cass River, the six transects ran in a series down the riverbed and transects intersected a range of habitat types. Some artificial retreats were placed in very stable stony ground with high coverage of moss and lichen. Others were in moist grassland close to channels of water feeding into the main river, and some ARs were in the disturbed riverbed with little vegetation. Following lizard monitoring best practice, ARs were placed on vegetation if possible without veering off the bearing of the transect too much. This is because lizards are more likely to be found in vegetated spots rather than bare stone (Department of Conservation, 2012).

The permanent placement of artificial retreats can alter lizard distribution and attract predators (Lettink & Monks, 2016). Therefore, the ARs specifically placed out for this study at Cass and Snowy Rivers were removed after inventory. The ARs at Paterson’s Terrace were already in place and were part of ongoing monitoring by the Department of Conservation and were not removed. These were used for comparison to a habitat where *B. robustus* already survives.

5.3.5 **Hybridisation with *Brachaspis nivalis***

Recordings made on iNaturalist NZ, a citizen science tool, were used to check for observations of *B. nivalis* in areas surrounding each of the potential translocation sites. Biological information known for *B. nivalis* was used to predict other similar habitats where *B. nivalis* would likely occur. We know that *B. nivalis* is an alpine species and usually inhabits scree slopes above 1500 m in the northern two-thirds of the South Island (Trewick, 2001). Using Google Earth Pro, scree slopes surrounding proposed translocation sites on the Tasman River above 1500 m elevation were marked on the map. The map also identified and marked clear paths that connected potential *B. nivalis* habitat to the riverbed by scree fan or stream (Figure 5.7). Two areas of scree slopes, Jacks Stream (-43.9667º, 170.0718º) and upper Forks Stream (-43.8528º, 170.3332º), adjacent to the Tasman and Forks potential translocation sites respectively, were physically searched for the presence of *B. nivalis* on a single day in November 2019.

5.3.6 **Data analysis for site comparisons**

All data were analysed in R, version 4.0.5 (R Core Team, 2021). The ground cover composition was compared across the five braided riverbed sites included in this study; Snowy, lower Ōhau, Cass, upper Tasman and lower Tasman Rivers. As this is a multivariate dataset, all ten ground cover variables within the five different sites were analysed by Principal Component Analysis (PCA) using packages *ggfortify* version 0.4.12 (Horikoshi & Tang, 2016) and *factoextra* version 1.0.7 (Kassambara & Mundt, 2020). Before conducting the PCA, correlations coefficients between all ground cover variables were inspected, with a correlation matrix constructed using package *corrplot* version 0.90 (Wei & Simko, 2021).

The statistical significance of the ground cover variables between the five different study sites was then investigated by MANOVA followed by ANOVA for each ground cover variable at each different site using the package *stats* version 4.0.5. Where ANOVA showed significance, pairwise comparisons were undertaken using package *emmeans* version 1.6.2-1 (Lenth, 2021). Boxplots of ground cover variable values were constructed using packages *ggsci* version 2.9 (Xiao, 2018) and *ggpubr* version 0.4.0 (Kassambara, 2020). Medians and interquartile ranges were produced for each site and ground cover variable and displayed in graphs.

5.4 Results

The upper Ōhau River supports a black-fronted tern (*Sterna albostriata*) breeding colony (Keedwell, 2005). Black-fronted terns are insectivores and would prey upon grasshoppers (Keedwell, 2002). They are also endemic to New Zealand and classified as Endangered (Miskelly et al., 2008); therefore, controlling them as a predator of *B. robustus* was not an option. Even though there is intensive mammalian predator control surrounding the area of the breeding colony (Anderson & Woolmore, 2009), the risk of predation by terns to *B. robustus* was too high. This eliminated upper Ōhau River as a translocation site. This site was also not much closer to the Southern Alps, or higher elevation, than the current populations of *B. robustus*. The proposed translocation site on the upper Ōhau River was only ~7.5 km from the current population on the lower Ōhau River and 51 m higher elevation. Therefore, it does not mitigate against the threat of climate change like some of the other proposed translocation sites do.

Macaulay River was excluded as the risk of flooding events and proximity to *B. nivalis* made this site not as suitable for translocation. Upper Fork Stream presented some obstacles in access. Permission from the New Zealand Defence Force was required to enter the land. Upon an initial survey, we found *B. nivalis* in very close proximity on a scree field above the potential translocation site. We hypothesised that it would not be difficult for a *B. nivalis* to accidentally travel down to the riverbed, subsequently increasing the risk of hybridisation between the two species. Due to restricted access and the high risk of hybridisation, Fork Stream was discounted as a translocation site in this study.

The remaining suggested translocation sites were two sites on the Tasman River (upper and lower Tasman) and Cass River. These three sites were surveyed for ground cover composition, risk of predation by mammals and lizards, risk of hybridisation with *B. nivalis*, and mitigation against climate change.

5.4.1 Ground cover composition at all sites.

This analysis compares ground cover across all five study sites: Snowy, Ōhau, Cass, upper Tasman and lower Tasman Rivers. By looking at the biplots (Figure 5.2 and Figure 5.3) and box plot (Figure 5.4) we can see which sites have a strong correlation with certain ground cover variables, and how they compare to Snowy and Ōhau Rivers.

The P-value produced from the MANOVA was $p < 0.001$, indicating that there was a significant difference in ground cover composition between sites. Because this was a significant result, we then conducted an ANOVA for each ground cover variable to test the relationship to the site.

Each of the five sites had a significant correlation to small rocks, as displayed in the box plot (Figure 5.4). This means that there was a high proportion of small rocks in the ground cover. Ōhau clearly had a higher percentage of lichen as ground cover than the other sites. The confidence intervals show that upper and lower Tasman River sites predominantly had high ratios of small rocks and very little vegetation cover. In comparison, the other three sites had higher values of vegetation than the Tasman sites, but were still dominated by small rock cover.

The proportion of variance in ground cover composition between the five sites explained by PC1 of the principal component analyses was 30.2%. The cumulative proportion explained by PC1 and PC2 was 49.4% (Table 5.1). Small rocks, herbaceous vegetation and grass had the highest contribution to PC1. Lichen and medium sized rocks had the highest contribution to PC2(

Table 5.2).

Table 5.1 Importance of components Table from summary PCA test

Variable	PC1	PC2	PC3	PC4
Standard deviation	1.7371	1.3856	1.0695	0.98397
Proportion of Variance	0.3018	0.192	0.1144	0.09682
Cumulative Proportion	0.3018	0.4937	0.6081	0.70493

Table 5.2 PC loadings Table showing the explanatory variables contribution to the PCs. Values over 0.6 are highlighted as having a significant contribution towards the PCs. This is an Eigenvector matrix that shows the correlation between the PCs (dependent variables or rows) and the input image bands (independent variables or columns). Values that are closer to 1 or -1 are more important on that PC.

Variable	PC1	PC2	PC3	PC4
Sand	0.15718738	-0.4529493	-0.53008014	0.57716846
Lichen	0.02404818	**0.7804102**	0.06838621	-0.06391703
Moss	0.4326467	0.143026	0.56556186	0.37707008
Small	**-0.8167401**	-0.3588568	0.29179683	-0.19514261
Medium	-0.16288601	**0.8057233**	-0.32236959	-0.07032722
Large	-0.11119352	0.4352836	-0.46068977	0.27365898
Herbaceous	**0.75757566**	-0.1475432	-0.19804849	0.03034838
Woody	0.5971435	-0.0141604	-0.14850812	-0.46543323

Mat	0.38022725	0.2491198	**0.61926865**	0.29296229

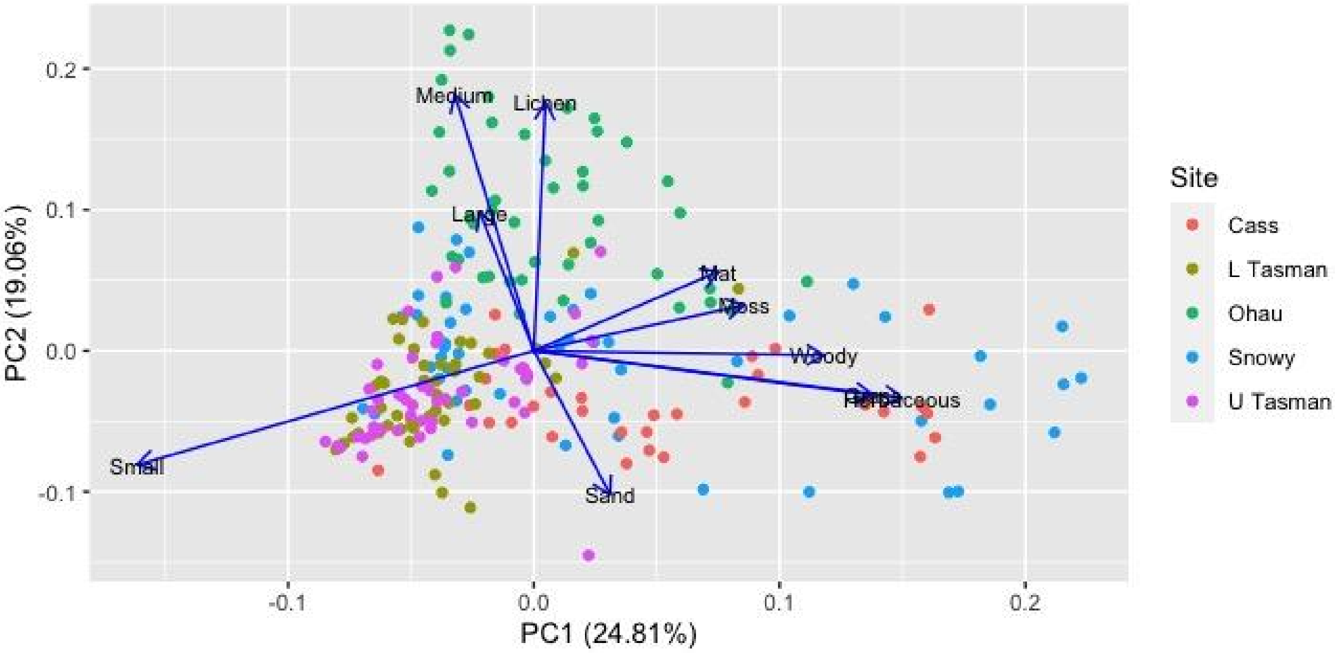


Figure 5.2 Biplot showing the spread of explanatory variables on PC1 and PC2. Snowy and Ōhau represents habitats where B. robustus is currently known to survive while the other three sites are being assessed as potential translocation sites

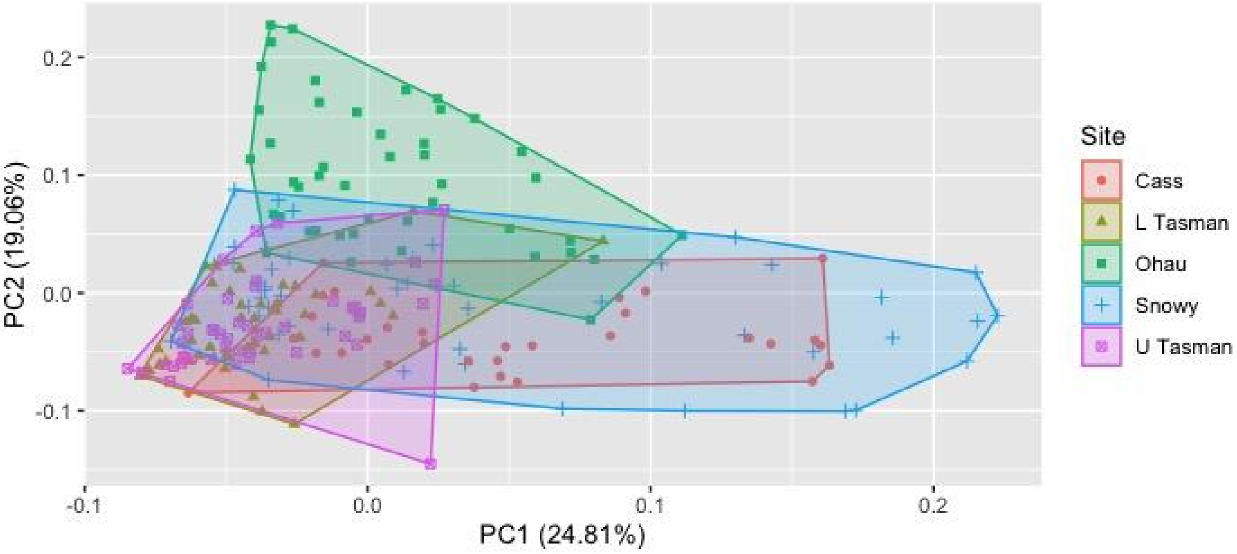


Figure 5.3 Biplot showing the overlapping ground cover values of each of the sites.

Ōhau River differs the most in ground cover to the other sites, including Snowy River (Figure 5.3 and 5.4); it has higher values for medium rocks and lichen. Snowy River was most similar to Cass River. Upper Tasman and lower Tasman are similar as expected due to their spatial proximity. A minimum of 3.164 kilometres separated the two sites.

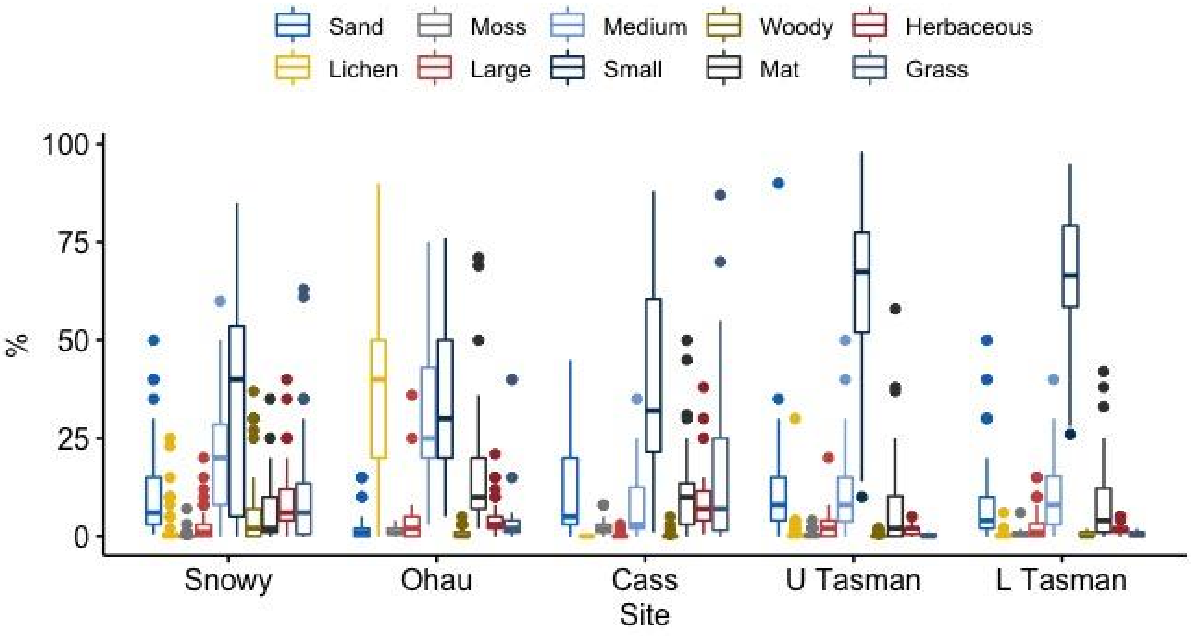


Figure 5.4 Ground cover variable percentages at each site. Small = rocks sized 1 – 10cm, Medium = rocks sized 10 – 30 cm, Large = rocks >30 cm. Mat = mat forming plant species. Herbaceous = leafy plant species that are not grass, woody, or mat forming. Woody = woody plant species (e.g. sweet briar, matagouri).

Snowy, Ōhau and Cass Rivers had fairly similar levels of small rock cover (Table 5.3). Upper and lower Tasman are similar to each other, with higher values than the other three sites. Small rocks were undoubtedly the most characteristic ground cover variable associated with current habitat of *B. robustus* and the most prevalent in the wider habitat of the riverbeds sampled. Snowy and Ōhau Rivers had the most similar results for the cover of small rocks. Cass River was the most similar translocation site to Snowy and Ōhau Rivers for ground cover characteristics. Lower Tasman had the highest percentage of small rocks out of all five sites. From the pairwise comparisons (Table 5.3), we can see that Snowy River is not significantly different to Cass River for small rocks but is significantly different to both Tasman River sites.

All sites except Ōhau had low values for lichen (median = 0). Ōhau had significantly higher values for lichen than any other site (median = 40). No lichen was recorded at Cass River. Upper Tasman is most similar to Snowy River for ground cover of lichen, closely followed by lower Tasman. No lichen was recorded at Cass. Snowy is not significantly different to either of the three translocation sites for lichen coverage.

Ōhau had the highest percentage of medium sized rocks (10–30 cm); the median value at that site was 25%. Snowy had the second highest proportion of medium sized rocks, with a median value of 20%. The proposed translocation sites all produced similar mean values for medium sized rocks. Snowy River (median = 6) is least similar to Cass River (medium = 3) for the cover of medium rocks. Cass River had the lowest value for medium rocks out of the five sites. Out of the three translocation sites, both Snowy and Ōhau (median = 25) were most similar to both Tasman Rivers sites (medians = 8) for medium rocks. Every site was significantly different to each other in the proportion of medium sized rocks when pairwise comparison was conducted.

The two Tasman River sites had very similar proportions of herbaceous vegetation (median = 2). This is to be expected as they are in close proximity to each other. Cass and Snowy Rivers had similar levels of herbaceous vegetation. The Ōhau River range of herbaceous cover lies in the middle, with a median value of 3%. Snowy River (median = 6) and Cass River (median = 7) had similar results for percentage of herbaceous vegetation cover. Snowy therefore was not significantly different to Cass for herbaceous plant cover, but it was significantly different to both Tasman sites.

To simplify the results, Ōhau River was removed from the summary table as most grasshopper microhabitat preference data was collected from Snowy River. Ōhau and Snowy Rivers were quite different in ground cover composition (Table 5.4), and there was a considerably larger population of *B. robustus* at Snowy, compared to Ōhau River. Therefore, to streamline the results, the decision was made to focus on the Snowy River.

Table 5.3 Summary Table of interquartile ranges and medians for each ground cover variable at each site. P-values are from the pairwise comparison between Snowy River and each of the three translocation sites. '*' indicates which of the translocations sites is most similar to Snowy River for that ground cover variable. '--' represents a statistically significant P-value (<0.05). Ōhau was removed due to the small data set collected (Chapter 3) and the difference from Snowy River.

	Grasshoppers present		Grasshoppers absent		
		Snowy	**Upper Tasman**	**Lower Tasman**	**Cass**
Small	**Median:**	40	67.5	66.5	32*
	IQR:	5 - 53.5	52 - 77.5	58.5 - 79.25	21.5 - 60.5
	P-value:		--	--	0.8558
Herbaceous	**Median:**	6	2	2	7*
	IQR:	4 - 12	0.5 - 2	1 - 2	4 - 11.5
	P-value:		--	--	1
Medium	**Median:**	20	8*	8	3
	IQR:	8 - 28.5	3.75 - 15	3 - 15.25	2 - 12.5
	P-value:		--	--	--
Lichen	**Median:**	0	0	0*	0
	IQR:	0 - 0.75	0 - 0.25	0 - 0.5	0 - 0
	P-value:	--	0.9832	0.91	0.8614
Sand	**Median:**	6	8*	4	5
	IQR:	3 - 15	4 - 15	2 - 10	3 - 20
	P-value:	--	0.9995	0.6087	0.967
Moss	**Median:**	0	0*	0.5	2
	IQR:	0 - 0	0 - 0.63	0 - 1	1 - 3
	P-value:		0.9485	0.743	--
Large	**Median:**	1	2	1*	0
	IQR:	0 - 3	0 - 4	0 - 3.25	0 - 0
	P-value:		1	0.9993	0.0705
Woody	**Median:**	2	0	0*	0
	IQR:	0 - 7	0 - 0	0 - 1	0 - 0
	P-value:		--	--	--
Mat	**Median:**	2	2*	4	10
	IQR:	1 - 10	0 - 10.25	1 - 12.25	3 - 13.5
	P-value:		0.9044	0.8734	0.1412
Grass	**Median:**	6	0	0.5	7*
	IQR:	0.5 - 13.5	0 - 0.5	0 - 1	1.5 - 25
	P-value:		--	--	0.1133

The asterisks (*) in Table 5.3 identify which of the translocations sites was most similar to Snowy River in the composition of each ground cover variable. Upper Tasman River had four ground cover variables that were most similar to Snowy River, lower Tasman River had three, and Cass River had three. Using the interquartile ranges and medians, the upper Tasman River is most similar in ground cover composition to Snowy River.

Table 5.4 Significant P-values of ground cover variables from pairwise comparisons between Snowy and Ōhau Rivers. These are the variables that Snowy River and Ōhau River significantly differ in. The two sites were not significantly different in their values for large rocks, small rocks and grass.

Ōhau - Snowy	**P-value**
Sand	0.0023
Lichen	<.0001
Moss	0.0005
Medium	<.0001
Herbaceous	0.0009
Mat	0.0004
Woody	<.0001

5.4.2 Predators: Tracking tunnels

Mammalian predator monitoring was not conducted at Cass River because Te Manahuna Aoraki already had data available on mammalian predator captures on their trapping network surrounding the Cass River. Trapping rates gave an indication of presence of mammalian predators. Table 5.5 shows the combined tracking tunnel rates from both Snowy River and Paterson's Terrace. Presence of non-targets, (possums, cats and invertebrates) was also recorded. Possum presence was recorded at both sites by the tracking tunnel cards being chewed or pulled out. Kitten footprints were recorded once in a tracking tunnel at Snowy River whilst baiting with rabbit meat. No rats were recorded as there are very few present in the Mackenzie Basin, and all rats that have ever been recorded in the area have been Norway rats (Sanders & Maloney, 2002).

Table 5.5 Combined tracking rates of mammalian predators at Snowy River and Patersons Terrace.

	Peanut butter	Rabbit meat
Species	**Tracking rate %**	
Hedgehog	30.435	36.23
Mouse	20	33.66
Mustelid	0	3.335
Skink	6.52	18.55

Results supplied by DOC where predator control occurs (Figure 5.5) and the Te Manahuna Aoraki annual report shows that hedgehogs, mice and mustelids are present on the Tasman and Cass River (Te Manahuna Aoraki, 2020).

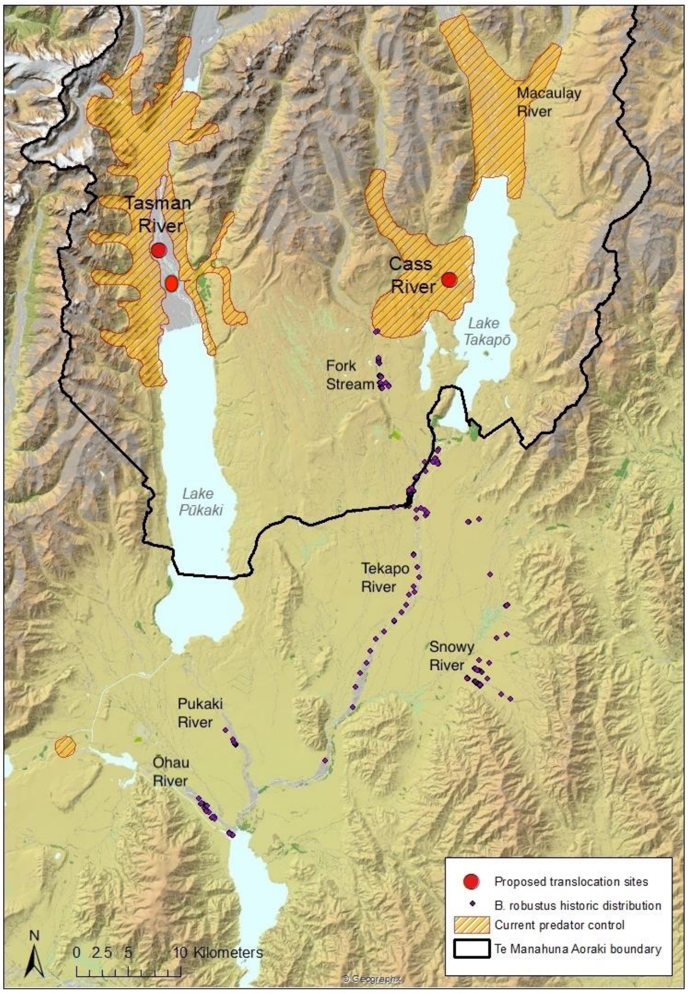


Figure 5.5 Map showing the historic distribution of *B. robustus*, the boundary of Te Manahuna Aoraki project, current predator control and the translocation sites that were assessed for this study.

5.4.3 Mesopredators: Artificial retreats

Out of the 60 artificial retreats installed at Snowy River, seven had one skink underneath them during monitoring inventory. This resulted in an occupancy rate of 11.7%. Photographs of two of the skinks recorded at Snowy River have been identified as McCann's skink (*Oligosoma maccanni*) on iNaturalist (inaturalist.nz/projects/vegetation-of-braided-rivers-mackenzie-basin). No skinks were found under any of the ARs at Cass River.

When I came to monitor the ARs at Cass River on January 22nd 2021, I found considerable disturbance to the riverbed as the surrounding pine forest was in the process of being felled. Logs were being stored on the riverbed, and it was evident that heavy machinery had been travelling through the area. The logs were placed on the first half of my first AR transect and destroyed six covers. This removed 10% of my data set and no lizards were detected under the remaining ARs.

The Department of Conservation in Twizel monitor 50 double layered ARs on the Tasman Riverbed that are checked once in each month (Leseberg, Wahlberg, Stevenson, & Maloney, 2005, unpublished internal report, DOC). Fourteen Southern Alps geckos (*Woodworthia* "Southern Alps") were found over two monitoring periods (February and November) in 2005, and 13 were found in March 2006 (Leseberg, Wahlberg, Stevenson, & Maloney, 2006, unpublished internal report, DOC). Three Southern Alps geckos, and one unidentified skink were found in November 2009, and four Southern Alps geckos were found in February 2010. Southern Alps gecko and an unidentified skink were found on the Tasman River during 2020/21 summer survey (Personal communication, Jennifer Schori, December 14, 2021). There have been no observations made of skinks or geckos on the Tasman River on iNaturalist NZ.

5.4.4 Hybridisation with *Brachaspis nivalis*.

On the mountain ranges bordering the Tasman River, 31 streams or scree fans that connect scree slopes and riverbed were located on the map (Figure 5.7), and 50 different scree slopes above 1500 m were identified as potential *B. nivalis* habitat. There were four iNaturalist observations of *B. nivalis* in the Mt Cook area at the time of this analysis, (red pins on the map Figure 5.7). There are no recordings of *B. nivalis* near the Cass River on iNaturalist and the nearest suitable *B. nivalis* habitat is about 8.8 km upriver from the potential *B. robustus* translocation site.

It is possible for *B. nivalis* to travel down from their alpine habitat and be found on braided riverbeds at much lower elevations. Multiple *B. nivalis* were found at the base of a scree fan on the upper Forks stream during the initial fieldwork for my thesis and Jacks Stream, located just below the Tasman River bed was found to have abundant *B. nivalis* at ~1500 m when searched in November 2019. An adult *B. nivalis* was also found in one of the REECE vegetation plots during ground cover surveys for this project (-43.8089º, 170.1370º) (Figure 5.6) and is marked with a red pin on the river bed (Figure 5.7)

Figure 5.6 The adult *B. nivalis* found in a REECE plot on the Tasman River. (Photo: Tara Murray, December 8, 2020).

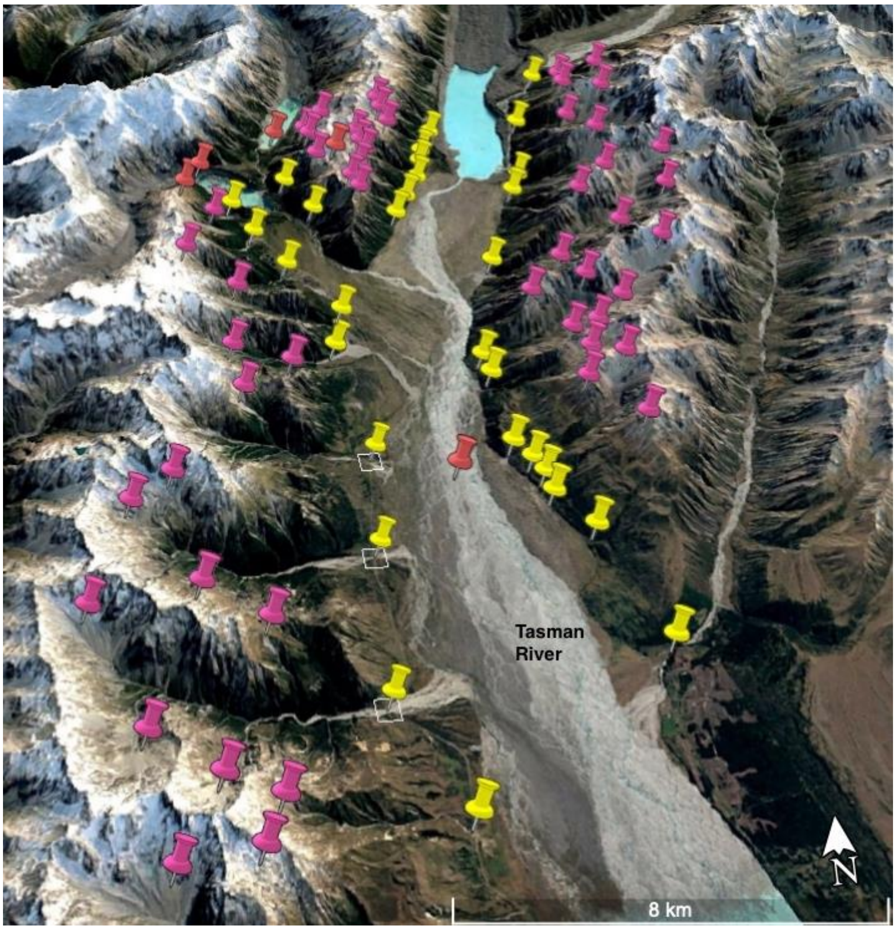


Figure 5.7 Pink pins represent scree slopes above 1500 m that is likely *B. nivalis* habitat. Yellow pins mark streams or scree fans that connect scree slope to the riverbed. Red pins show the location of the four iNaturalist observations made of *B. nivalis*.

5.5 Discussion

5.5.1 Habitat suitability at proposed translocation sites

The Macaulay River, upper Forks Stream and upper Ōhau River were all removed from the list of potential translocation sites early on as they were determined to be unsuitable for *B. robustus*. Below details the positives and negatives for the remaining potential translocation sites (Tasman and Cass Rivers), comparing them to the two sites of known *B. robustus* populations (Snowy and Ōhau Rivers).

5.5.2 Tasman

Ground cover characteristics

The Tasman River is estimated to be about ~6178 ha (O'Donnell, 2000). A study conducted by Schori et al (2020b) found that an adult female *B. robustus* utilise at least 300 m^2 (30 % of 1 ha) of suitable habitat. Therefore the size of the Tasman River bed should provide ample area for a population of *B. robustus*.

Woolmore (2011) presents a comprehensive and in-depth description of the plant community composition at three of the four rivers included in this current study. On the Tasman River, 152 plots were sampled for vegetation community type. Plots were boundless, occupying areas of uniform landform. The mean estimated plot size was 213 m^2, considerably larger than the survey methods used in this current study (25 m^2). The Tasman River was characterised by large *Raoulia* cushion fields and low frequencies of exotic plant species. The Tasman River's climatic variables (water-balance ratio, minimum-annual temperature) and isolation contribute to the low levels of exotic plant species. There are high numbers of threatened plants found on the Tasman River. There were nine different plant community types found on the Tasman River, according to Woolmore's classification. The main community types found there were *Raoulia haastii* and *R. australis* cushion fields. These areas had a vegetation cover of 27% of the ground cover in that particular community type. The majority of the vegetation was *Raoulia* cushion plants, lichens and mosses. I purposefully did not put my vegetation sampling transects in areas with these cushion fields as it is not suitable habitat for *B. robustus* due to the high percentage of vegetation cover. There are also large areas of stonefield on the Tasman River comprised mostly of bare rock (97% of ground cover in those areas sampled) and minimal vegetation cover. There were two other predominant plant community types found on the Tasman, both stone fields with low levels of vegetation ground cover (<10%) and short (0 - 10 cm) vascular plant species (Woolmore, 2011).

Predation

The Southern Alps gecko (*Woodworthia* “Southern Alps”*)* is present on the Tasman River (Leseberg, Wahlberg, Stevenson, & Maloney, 2006, unpublished internal report, DOC). Geckos in the *Woodworthia* genus are omnivorous and often prey on invertebrates are other small animals (Cree & Hare, 2016). Likely, geckos would also prey on *B. robustus.*

A big positive for the Tasman River is that there is intensive mammalian predator trapping occurring on and surround the riverbed. However, the Tasman River is considerably larger than Snowy or Ōhau Rivers. Therefore there is more land to ensure predators are suppressed on, and a large effort is required to minimise predators. Mammalian predator control requires significant resource and financial input. The most cost-effective methods of controlling mammalian predators in areas as large as Tasman and Cass River is trapping with 0.2 traps per hectare, compared to using predator exclusion fencing at smaller sites (Norbury, Hutcheon, Reardon, & Daigneault, 2014).

Hybridisation

The fact that a *B. nivalis* individual was found amongst the ground cover survey plots during the field work on the Tasman River (Figure 5.7) may greatly decrease the Tasman Rivers suitability as a translocation site. Laboratory experiments need to be conducted to confirm if *B. robustus* and *B. nivalis* will copulate and produce viable offspring. If it is found that the two species do successfully hybridise, this may rule out the Tasman River as a translocation site. However, it depends on how important conservation managers deem the risk of hybridisation. It was likely an accident that the *B. nivalis* individual made its way down the riverbed. It was possibly washed down a stream from the scree slope it was inhabiting. *Brachaspis* surveys of the Tasman River have been conducted since ~1995, and this is the first *B. nivalis* to have been recorded on the river bed. Further monitoring of the Tasman River should be conducted to assess if there are resident *B. nivalis* living there and if there is a breeding population there. More information is needing on the egg physiology of both species to understand if they can hatch successfully at the altitude and environment of the Tasman River which has an altitudinal range of about 520–700 m. It is possible that *B. nivalis* may not be able to hatch, seeing as they usually are found above 1500 m and may require more frost days than *B. robustus* to break diapause and hatch. However, we do not know this and it was outside of the scope of this project to investigate. Further research needs to be done to determine if *B. nivalis* can reproduce and hatch on the Tasman River.

5.5.3 **Cass River**

Ground cover characteristics

Like most braided river systems, Cass River is characterised by a highly variable flow and changeable gravel formations (Caruso, Edmondson, & Pithie, 2013). The area of suitable habitat for *B. robustus* to be translocated to was roughly 20 ha. It may be possible for the grasshoppers to disperse down the river, towards Lake Tekapo. Cass River has one of the highest levels of flood disturbance, and is characterised by large areas of bare rock and low levels of vegetation cover (Woolmore, 2011).

Predation

Mammalian predator monitoring was not conducted at Cass River as part of this study because data was available from the predator control programe led by Te Manahuna Aoraki. Trapping rates from the area showed that hedgehogs, mice and mustelids have been captured near Cass River (Te Manahuna Aoraki, 2020). The river is surrounded by farmland, and immediately boardering the potentially suitable habitat for *B. robustus* are dense swaths of grass. Dense vegetation provides habitat and cover for mammalian predators (Hoare, Adams, Bull, & Towns, 2007).

When conducting fieldwork for this study, it was clear that Cass River had abundant bird life, more so than the other four sites. During the time I spent there in November 2020, there was a black-fronted tern colony, many banded dotterels, and a black-billed gull breeding colony (Te Manahuna Aoraki, 2020). Many native bird species found on braided rivers in the Mackenzie Basin feed on aquatic and terrestrial invertebrates and would most likely include *B. robustus* in their diet (Hughey, 1997; Lalas, 1977). This may be an indication of reduced mammalian predators as a result from the predator control undertaken by Te Manahuna Aoraki. Although *B. robustus* are adapted to avoiding avain predators, it is unlikely that they would be able to sustain the pressure of living next to a bird colony, especially in initial low densities after translocation.

Hybridisation

The risk of hybridisation with *B. nivalis* is lower as Cass River than it is at Tasman. There have been no recordings of *B. nivalis* in the mountain ranges surrounding the Cass River. However, suitable *B. nivalis* habitat is still present upriver of the potential translocation site and it is possible that *B. nivalis* are there and that they could be accidentally washed downstream. The nearest *B. nivalis* habitat is about 8.8 km upriver.

5.5.4 Snowy

Ground cover characteristics

Snowy River was not included in Woolmore's (2011) vegetation assessment. The ground cover results for Snowy from this study are reported earlier in the chapter. Overall, Snowy had high levels of vegetation cover (Table 5.3) as flooding events are not common there.

Predation

Mammalian predator monitoring showed that hedgehogs, mice, mustelids and cats are present at this site. This was expected because there is no mammalian predator control occurring in the area and there are no future plans for Te Manahuna Aoraki to extend their boundary to encompass Snowy River in their predator control network. Therefore, the pressure of mammalian predators would be lower at Cass and Tasman Rivers.

Hybridisation

The risk of hybridisation is low at Snowy River. *Brachaspis nivalis* have never been recorded in the mountains surrounding Snowy River. The nearest potentially suitable *B. nivalis* habitat is about 8 km upriver of the *B. robustus* population.

5.5.5 Ōhau

Ground cover characteristics

Ōhau River was characterised by having exceptionally high levels of vascular-plant cover as well as high levels of moss cover. Characteristics of high levels of vegetation represent more stable surfaces and less disturbance and are older in terms of floodplain development (Woolmore, 2011). There were high levels of exotic plant species found on Ōhau River. Grasslands and *Rosa rubiginosa* were common on this river. Conversely, there were also areas where bare rock made up 97% of the ground cover. Considerably fewer vegetation plots were sampled on Ōhau River (18) by Woolmore than that of the Tasman River (152). This was due to the significant difference in size between the two river beds as plots were sampled at an intensity of 23 plots per 1000 ha. It is clear from my vegetation analysis for this current study, and Woolmore's analysis that the Tasman River suffers much less disturbance and exotic weed invasion than Ōhau or Cass Rivers (Woolmore, 2011).

Predation

I attempted to replicate my tracking tunnels methods carried out at Snowy River and Paterson's Terrace at the Ōhau River site, but permission from the Department of Conservation was not granted. Placing semi-permanent tracking tunnels at Ōhau River would've come with some difficulties as the public have access to the riverbed, and members of the public often walk and drive vehicles through the site. This may have caused some interference with the tracking tunnels. The Ōhau River is not included in the Te Manahuna Aoraki mammalian predator control network (Figure 5.5) and most likely will not be in the future either.

Hybridisation

The risk of hybridisation at the Ōhau site is very low as there are no potential pathways for *B. nivalis* to travel down from nearby scree slopes and reach the riverbed. The Ōhau canal blocks any potential pathway from the Benmore Range.

5.5.6 Conclusion

One significant finding of this study was *B. nivalis* on the Tasman Riverbed and upper Forks stream. This species is an alpine grasshopper, usually occurring on scree slopes above 1500 m. There is substantial concern that *B. nivalis* and *B. robustus* may hybridise if they come in contact. Observing *B. nivalis* individuals on the river beds where it is being considered to translocate *B. robustus* is therefore cause for concern. It is feasible that *B. nivalis* could travel down from nearby scree slopes and become present on Cass River. This topic warrants further research to determine the level of risk and if the two species would produce viable offspring.

There are key mammalian predators present at all current habitats of *B. robustus*. Predation is thought to be one of the biggest causes of decline for this species. It is imperative that mammalian predator monitoring is thoroughly conducted as it is important to understand predator pressure at translocation sites before translocation of this species is conducted. Knowing what species of predators and if they are in high or low numbers is important. Studies have shown that only using one type of predator control to suppress mammalian predators may negatively impact non-target species if only top predators are supressed (McIver, 2020; Norbury, Heyward, & Parkes, 2009).

Mesopredator release occurs when the decline of a top predator allows for lower trophic predators to increase in numbers (Prugh et al., 2009). Modelling shows that current mammalian predator control regimes in New Zealand dryland ecosystems will likely result in predator release of animals such as mice and rabbits. This will have a cascading effect throughout the food chain and potentially detrimental effects on native fauna (Ramsey & Norbury, 2009). The introduction of the rabbit haemorrhagic disease in 1997 presented an opportunity to study vegetation and ground invertebrate fauna response to declining rabbit numbers (Norbury et al., 2009). It was confirmed that ground vegetation and invertebrates mainly had positive responses to rabbit removal. However, the responses were much more complex than expected, and the hypothesis that rabbits reduce ground invertebrate abundance by reducing vegetation cover and providing prey to mammalian predators (namely cats and ferrets) could not be proved. Norbury, Heyward, and Parkes (2009) suggest that other important factors influence ground invertebrate fauna, such as climatic conditions.

Each of the translocation sites have some characteristics that meet the requirements of *B. robustus* suitable habitat. However, through my analysis and previous studies (Woolmore, 2011), it is evident that the Tasman River and Cass River are very different to each other. The Tasman River is also much larger than the Cass River. Due to the nature of braided river systems, the channels and islands on the riverbed are constantly changing. The estimated size of the Tasman River is ~6178 ha, Cass River is ~1583 ha, and Ōhau ~712 ha (O'Donnell, 2000). However, these areas do not represent the suitable habitat for *B. robustus* as some of the above estimates would include river channels or habitats with ground composition not suitable to *B. robustus*.

The Tasman River has more intensive mammalian predator control on and surrounding the riverbed than Cass River. The Tasman River has a lower proportion of vegetation cover and even lower exotic vegetation than Cass River. The Tasman River is also closer to the Southern Alps than the Cass River. It is hypothesised that habitats closer to the Southern Alps and at a higher elevation will mitigate future climate warming impacts. There are native lizard species present on the Tasman River, whereas no lizards were detected during my study on Cass River. Results from my ground cover analysis show that the upper Tasman River is most similar to Snowy River in vegetation and bare rock composition. My study and previous studies (Schori, 2020) show that *B. robustus* select against habitats with dense vegetation. Results also show that the potential translocation site on the lower Tasman River is most similar to the current population sites of Ōhau River. Ground cover composition is just one factor that contributes to suitable habitat for *B. robustus*. The biological requirements of this species at all life stages must be considered. Considering the results gained from this study, as well as information from previous studies, it appears that the Tasman River, specifically the upper Tasman where I surveyed the habitat, would be the most suitable habitat for a translocated population of *B. robustus,* as long as the threat of hybridisation with *B. nivalis* can be shown to be minimal.

Chapter 6

Conclusions and recommendations

6.1 Translocations site selection recommendations

Determining the most suitable translocation site for *B. robustus* requires considering all possible threats to species at each site and weighing up which threats are more important. The Tasman River appears to be the most suitable for *B. robustus*, based on the ground cover composition data. My research on the habitat preferences of *B. robustus* show that there is not a significant difference between male and female preferences of this species. The Tasman River has fewer exotic plants than the Cass River and a more intensive mammalian predator control network surrounding the site, so it may be more protected against the threat of mammalian predators than at Cass River. However, the risk of *B. robsustus* coming in contact with *B. nivalis* on the Tasman River bed is moderately high. Therefore, it is vital that the threat of hybridisation between the two species is assessed before translocation to this site is considered. The Tasman River also experiences a lower frequency of flood disturbance than the Cass River (Woolmore, 2011). It seems that *B. robustus* might need a habitat that is less disturbed and has stable river gravels. This is because mosses and lichens that this species relies on for nutrient and moisture intake are slow growing (Hale Jr, 1959), and flooding events often remove these species (Caruso, Edmondson, et al., 2013). Research has shown that dense vegetation is unfavourable for *B. robustus* as it prevents movement and provides habitat for mammalian predators (Schori, 2020). Additionally, my microhabitat preference study shows that *B. robustus* were almost never found in close proximity to moss or lichen, or any other vegetation. Flooding events are an important element of braided rivers as they help wash away invasive vegetation (Caruso, Ross, et al., 2013). Since the establishment of the Waitaki hydro scheme, severe flooding events no longer occur on several of the rivers that *B. robustus* inhabit. The loss of these natural disturbance events had resulted in an increase in tall exotic vegetation and, therefore, loss of habitat for *B. robustus* (O'Donnell, Sanders, Woolmore, & Maloney, 2016). This makes Tasman and Cass Rivers suitable for *B. robustus* translocation as they still have high disturbance levels from flooding and suffer from lower weed invasion. Results from Chapters 4 and 5 are summarised in Table 6.1 as an assessment of how suitable each potential translocation site would be for *B. robustus*. Current population sites are also included.

Table 6.1 Assessment of suitable ground cover and mitigation for threats to *B. robustus* at each potential translocation site and current population sites.

	Suitable ground cover	Mitigation of mammalian predation	Mitigation of hybridisation	Mitigation of global warming
Upper Tasman River	High	High	Low	Medium
Lower Tasman River	High	High	Low	Medium
Cass River	Medium	Medium	Medium	Medium
Snowy River	High	Low	Medium	Low
Ōhau River	High	Low	High	Low

6.2 Translocation logistics

Lack of adequate predator control and habitat suitability are identified as the two main contributing factors for why animal translocations fail (Sheean et al., 2012). Poniatowski & Fartmann (2010) recommend that it would be more beneficial for flightless Orthopteran species to improve the habitat quality where they are currently found, rather than focusing on providing habitat corridors. Although a connection between fragmented habitats would also be valuable, this often isn't practical for flightless species as they need very close stepping stones to disperse to other patches of suitable habitat (Poniatowski & Fartmann, 2010).

Considering the known threats to *B. robustus* and the level of these threats posed at the current population sites verse the level of threats at potential translocation sites, it would be easier to translocate the species rather than manage the threats in situ. However, the short- to medium-term solution would be to try manage threats in situ as well as beginning to undertake translocations. The current population sites had higher levels of invasive plants that the translocation sites. Herbaceous weeds would be incredibly hard to control on a riverbed as management would have to involve herbicides. Usually flooding manages vegetation levels on braided rivers (Caruso, Ross, et al., 2013), but Ōhau River is affected by a hydroelectricity dam above it and disturbance levels are decreased. Snowy is not affected by hydroelectricity and has been known to flood naturally, but it usually runs dry (Schori, 2020; White, 1994).

Hedgehogs, mice and mustelids were all recorded in tracking tunnels at Snowy River and Paterson's Terrace. Trap data from Te Manahuna Aoraki show that these species are also present on Cass River and Tasman River (Te Manahuna Aoraki, 2020). This study did not aim to quantify mammalian predator density, but rather presence/absence. We therefore do not have a conclusive answer for if mammalian predator pressure is higher at current *B. robustus* population sites, or at the proposed translocation sites, but it is fairly safe to assume the mammalian predation pressure would be less at the Tasman River where intensive trapping is occurring than one that had no form of control occurring.

Setting up a mammalian predator control network requires significant time and money (Norbury, Hutcheon, et al., 2014). There is an existing, intensive mammalian predator control network that encompasses the potential translocation sites and Te Manahuna Aoarki will most likely not be extending their mammalian predator control boundary to the lower Ōhau River and Snowy River. Mammalian predator control is most successful when conducted at a landscape scale. If a small area has a trapping network, but all the surround landscape doesn't, the area will constantly be invaded by mammalian predators (Reardon et al., 2012). Therefore, it would be more efficient to translocate *B. robustus* to a habitat that already receives mammalian predator control.

One option to be considered when selecting microhabitats for *B. robustus* release is to place them on braid islands on riverbeds. Braid islands are natural features on braided riverbeds and can be used as a barrier to mammalian predators. This method has been used to some effect for the black-fronted terns on the upper Ōhau River (Schlesselmann, O'Donnell, Monks, & Robertson, 2018). Stoats are known to be strong swimmers and are capable of swimming >1 km (King, Veale, Patty, & Hayward, 2014). However, it is unlikely that a stoat would seek out *B. robustus* as prey if it meant swimming through a potentially swift current to gain access to the grasshoppers, as it is assumed that the presence of water acts as a deterrent (Duncan, Hughey, Cochrane, & Bind, 2008). Many native birds now rely on braid islands to breed due to predation from introduced mammals. However, water extraction from riverbeds for irrigation and hydroelectricity alter flow regimes. Duncan et al. (2008) found that a flow rate of 60–90 $m^3 s^{-1}$ is needed to successfully defend braid islands against mammalian predator invasion (Duncan et al., 2008). However, braided rivers are dynamic and water channels and braid islands are often changing (Caruso, 2006). Therefore, conservation managers should ensure that more than one braid island is populated to ensure gene flow within a metapopulation of *B. robustus* (Armstrong & Seddon, 2008; Maag, Karpati, & Bollmann, 2013).

6.3 Captive rearing for a translocation

Translocations of species usually involves extracting a number of individuals from their current biogeographic range and moving them to a habitat to reduce the risk of extinction (IUCN, 2013). However, in some cases, the remaining population is so small that removing individuals from the source population is not viable (Watts et al., 2008). Endangered insects can be reared in captivity and released to establish a new population (Stringer & Chappell, 2008). This method was used for the extremely endangered Middle Island tusked wētā (Winks, 2002). A total of 134 wētā were produced from one breeding pair and released to Double Island and Red Mercury Island between 2000 and 2003 (Watts et al., 2008).

A restoration programme for the rare field cricket (*Gryllus campestriss*) and the wart-biter bush cricket (*Decticus verrucivorus*) was established in 1991 in the UK (Pearce-Kelly et al., 1998). Because the remaining population of *G. campestris* was so small (<100 individuals) wild-to-wild translocations were not viable for this species (Pearce-Kelly et al., 1998). The cricket species *D. verrucivorus* was in a similar situation with records of only one population in 1991. Captive rearing programs were developed for both species to provide the large numbers of crickets necessary to successfully begin new populations (Pearce-Kelly et al., 1998).

Winks et al. (2002) investigated the biology of Middle Island tusked wētā and their suitability to be bred in captivity. He then successfully translocated the reared wētā to other Mercury Islands. Data on developmental rates, mortality, oviposition substrate preference, fecundity, and behaviour were collected during the trial. Microhabitat data was collected on the proposed translocation site (Double Island) to enable successful laboratory rearing of the wētā. Air temperature, soil temperature, soil moisture content, rainfall and substrate type data were recorded. Conditions on Double Island were mimicked as best as possible in the rearing laboratory. Preference of substrate type for oviposition was measured by providing an impregnated female wētā with various substrate types. Statistical analysis showed a strong ($P< 0.001$) correlation between substrate type and oviposition. The duration of Middle Island tusked wētā life stages was closely examined, offering further insights into their biology. These trials carried out on captive wētā and micro-habitat data collected from proposed translocation sites allowed for a well-informed translocation project. This thorough and in-depth captive rearing guide by Winks et al. (2002) resulted in the successful establishment of Middle Island tusked wētā on Double and Red Mercury Islands, possibly saving the species from extinction (Stringer & Chappell, 2008). Similar studies have been done prior to the successful translocation of giant land snails *Placostylus ambagiosus* and *P. hongi* (Stringer & Grant, 2007). Similar methods need to be carried out in a captive rearing trial for *B. robustus* to be able to successfully produce *B. robustus* nymphs in a laboratory for translocation.

Using captive-reared insects can also be more successful than using wild-caught specimens. The level of stress inflicted on captive-reared insects is lower when it comes to the day of translocation (Parker et al., 2015). Often, specimens can be transported and released from the container that they were bred in, reducing the amount of handling. All prior measurements of individuals can be done well in advance of translocation day, which reduces stress on the day. Insects are reared in climate-controlled conditions with high food availability, resulting in good condition of insects that develop faster than those in the wild (Sherley, 1998). Captive rearing can be used to coincide with optimal release times, such as after predator control programs and during spring/summer (Seddon, Armstrong, & Maloney, 2007). Collecting specimens from the wild takes many hours and people. This accumulates cost for the conservation program. Captive rearing insects is a much more cost-effective method as potentially hundreds of individuals could be produced with minimal man power required (Stringer et al., 2018).

Rearing *B. robustus* individuals in a laboratory will be the most effective way to gather a sufficient number of individuals to support a viable population at the translocation site. This is because the natural populations of *B. robustus*, the source population, are already small and patchy in distribution. It would be illogical to deplete the source population so that the species dies out. Captive rearing in the field and in a laboratory for this species has not been overly successful in the past as the survivorship of nymphs was low (Schori, 2020). One of the reasons that *B. robustus* nymphs died in the laboratory experiment was due to a fungal infection of an entomopathogenic *Beauveria* fungus. Research into the impacts of this fungus need to be undertaken urgently. Having an in-depth understanding of the biology of *B. robustus* will greatly improve captive rearing success (Schori, 2020).

6.4 Post translocation monitoring methods

Post-release dispersal has become a significant challenge for animal translocations (Armstrong & Seddon, 2008). Habitat suitability and sufficient area of such suitable habitat is known as the foundation for translocations (IUCN, 2013). However, due to the stress caused by aspects of the translocation, animals often disperse away from the carefully selected, or restored, habitat (Dickens et al., 2010). Animals that move away from the release site and suitable habitat are at risk of starvation and predation (Letty et al., 2007). Dispersal from the release site can be okay if the surrounding landscape is also suitable habitat for that species (Armstrong et al., 2013). Although many translocation studies highlight stress from handling a reason for dispersal from release site (Dickens et al., 2010) (Parker et al., 2015), most of the grasshoppers that were captured, marked and re-captured were found within 16 m of the original capture location, even after 17 days.

Post-release monitoring allows conservation managers to assess the success of the translocation and to inform future translocations. Monitoring is therefore a fundamental aspect of translocation (Berger-Tal et al., 2020). As described in Chapter 3, there are many ways to mark insects for monitoring. Mark-recapture was used in this study and was useful but there were significant challenges in finding individuals of this cryptic species and mark-loss due to moulting. Radio telemetry shows promise in being a successful monitoring tool for *B. robustus* if the technology can be refined.

Schori (2020) conducted an experimental translocation of *B. robustus* in the Mackenzie Basin. They found that at least half of the grasshoppers released survived the initial translocation. More may have survived but because of mark loss and their cryptic nature they could not be found. Nevertheless, after initial losses either to death or dispersal, the number resighted remained consistent through to winter and almost all individuals present at the end of autumn were observed again post winter. This is despite the fact that natural mortality should be expected as some of the individuals would be adults that would have finished breeding by this point (Schori, 2020; Murray & Schori, unpublished report, 2021).

Dispersal from a translocation release site on the Tasman River may still result in the successful establishment of a *B. robustus* population as the Tasman River is a large area, estimated to be ~6178 ha (O'Donnell, 2000). There is likely more suitable habitat for *B. robustus*, other than where was surveyed for this study, available on the wider riverbed. However, a sufficient number of individuals should be kept close enough together after release so that they are able find each other and breed. If releasing grasshoppers from captivity, they could be released after breeding.

Fences have been used delayed release translocation of lizards in New Zealand (Knox, Jarvie, Easton, & Monks, 2017; Norbury, Van Den Munckhof, et al., 2014). One study showed the post-release dispersal of delayed release jewelled geckos was less than immediate released jewelled geckos. Site fidelity increased and home ranges decreased for geckos that were delay released. This had positive implications for translocation of these geckos (Knox et al., 2017). Similar methods could be used for *B. robustus* translocation. Keeping the translocated grasshoppers in a small area for a period after release may increase post translocation survival and allow for monitoring of the species. The fence would also exclude most mammalian predators from the area, further increasing grasshopper survival potential.

6.5 Climate change

The impending threat of climate warming is hypothesised to be somewhat mitigated at Tasman and Cass River compared with Snowy and Ōhau Rivers where *B. robustus* currently exists. This is due to the higher elevation, and the proximity to the Southern Alps. Measurements taken from the Hermitage, Mt Cook (top of Tasman River) show a mean air temperature of 8.4, mean-daily-maximum of 13.4, and mean-daily-minimum of 3.4 (Woolmore, 2011). Lake Tekapō is the closest weather station to Snowy River and in 2011 the mean air temperature was 8.8, the mean-daily-maximum was 14.4, and mean-daily-minimum was 3.3. Lake Tekapō actually received more frost days (mean = 175.8) than upper Tasman River (mean = 131.7) (Woolmore, 2011). Current data, therefore, does not support the hypothesis that Tasman River will be significantly colder than Snowy or Ōhau Rivers. Alpine regions are colder though, so when the climate change projections come true, closer to the Southern Alps should be colder.

6.6 Future research recommendations.

There remains much to understand about the biology of *B. robustus* to enable conservation managers to successfully translocate this species. Further work needs to be done to increase the robustness of ground cover preference data collected in this study. Methods used in this study should be replicated to increase the sample size and it should be conducted earlier in the season to coincide with adult female *B. robustus* presence. It could also be informative to repeat the microhabtiat survey at night. Mammalian predation has been identified as one of the main causes for decline in this species. It is therefore vital mammalian predator control is conducted at proposed translocation sites before translocation of *B. robustus* is performed.

Further work needs to be conducted to assess the likelihood of *B. nivalis* and *B. robustus* hybridising in the wild. As highlighted in Chapters 2 and 5, the two species could potentially come in contact if *B. robustus* are translocated to the Tasman River. If the species mate and produce viable offspring this may destroy the genetic integrity of these rare endemic species. Research on hybridisation, and fungal diseases that affect *B. robustus*, need to be conducted in conjunction with any captive rearing research.

Investigating the temperature threshold of *B. robustus* and gleaning information on the obligatory diapause this species goes through would allow more successful management of *B. robustus,* informing both captive rearing and translocation. It would be helpful to know the length of time and temperature requirements needed to break *B. robustus* dispause; this would inform conservation managers on where to translocate this species to. Therefore, accurate climate data must be collected for proposed translocation sites and climate warming predications made for those sites too. Temperature threshold information for *B. robustus* would be useful for monitoring of the species as it is possible that *B. robustus* become inactive above a certain temperature.

There still remains a knowledge gap in the requirements *B. robustus* has in an environment and the species general biology. This thesis aimed to partially fill that gap by understanding the ground cover composition preferences of the species. However, there is still much to understand about the food and moisture requirements of this species, whether they have different requirements in a habitat at different life stages and what the full suite of threats to this species is.

Appendix A

Common vascular dicot plants species found in ground cover composition plots

Figure A.1 *Echium vulgare* (naturalised species) at Snowy River in January 2020.

Figure A 2 A young *Echium vulgare* (naturalised).

Figure A 3 ***Epilobium melanocaulon*** **(native). Although, there is a possibility it could be** ***Epilobium glabellum***

Figure A 4 ***Eschscholzia californica*** **(naturalised) at Ōhau River.**

Figure A 5 ***Sedum acre*** **(naturalised).**

Figure A.6 *Rosa rubiginosa* (naturalised).

Figure A 7 *Rumex acetosella* (naturalised).

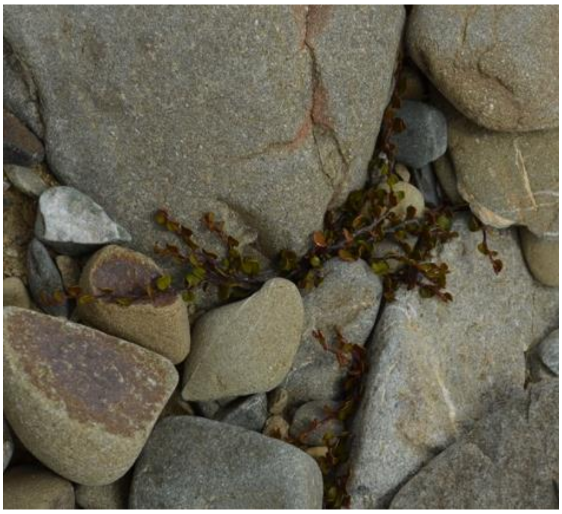


Figure A.8 *Muehlenbeckia axillaris* (native)

Figure A 9 *Arenaria serpyllifolia* (naturalised)

Appendix B

Means and interquartile range of ground cover variables

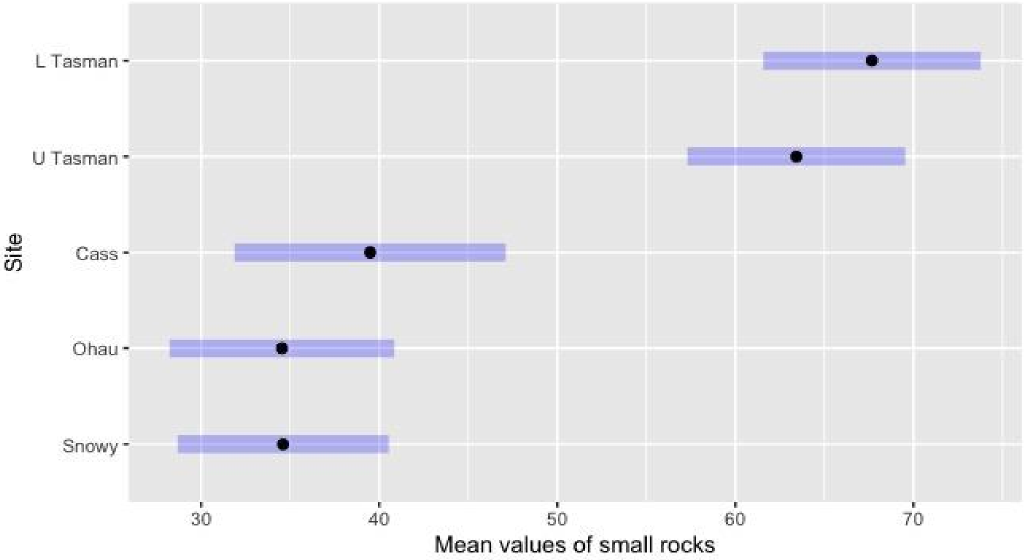


Figure A 10 Cass, Ōhau and Snowy Rivers had similar means and IQR for small rock cover. Upper and lower Tasman were similar to each other.

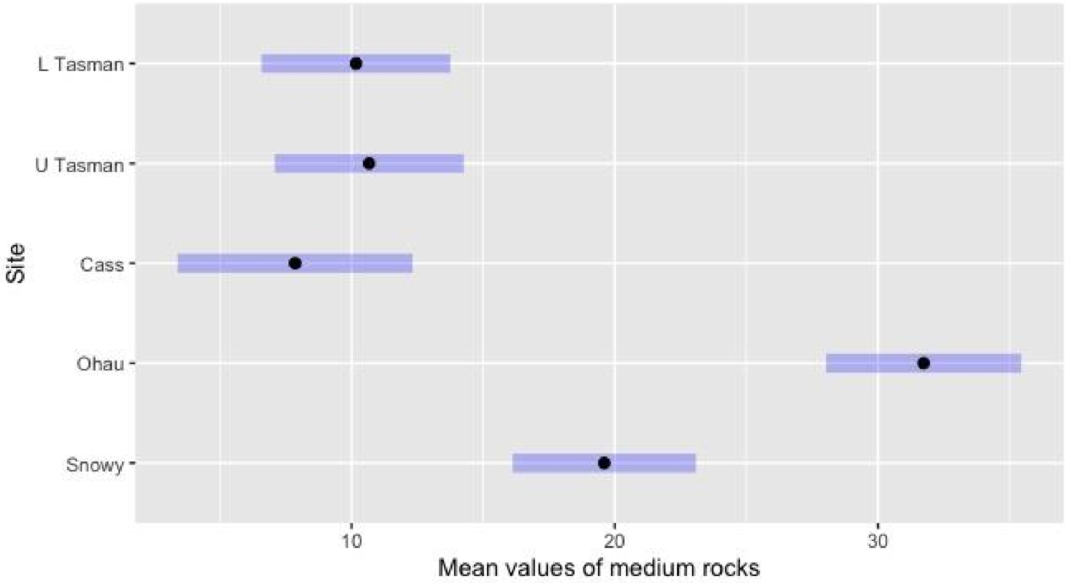


Figure A 11 The Tasman River sites and Cass all had similar values for composition of medium sized rocks. Ōhau River had the highest mean for medium rocks.

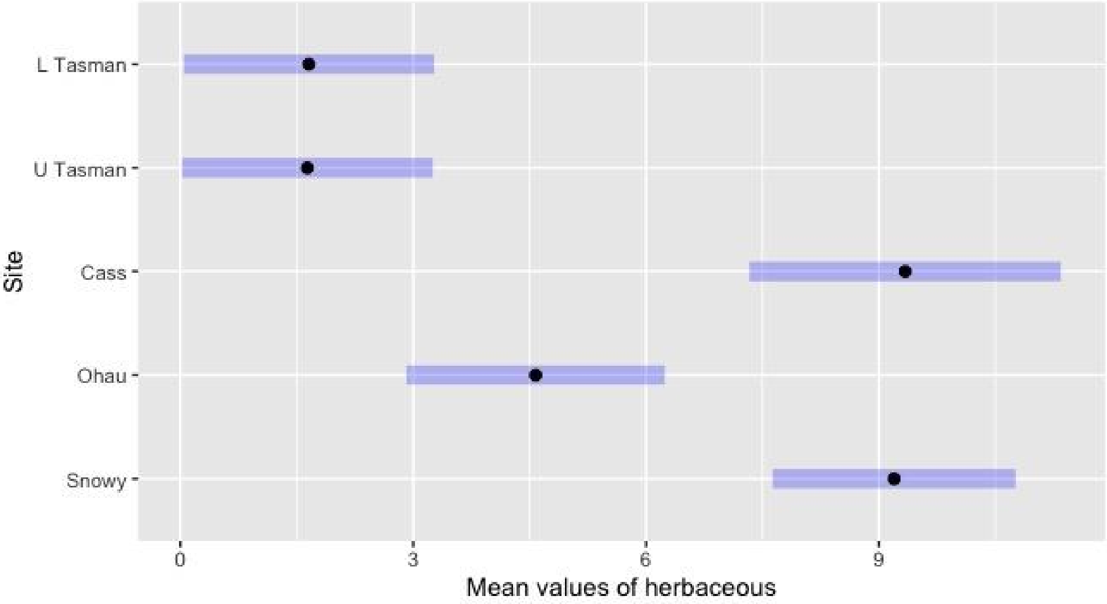


Figure A 12 Both Tasman River sites had very similar means for herbaceous vegetation cover. Snowy and Cass Rivers were also similar to each other.

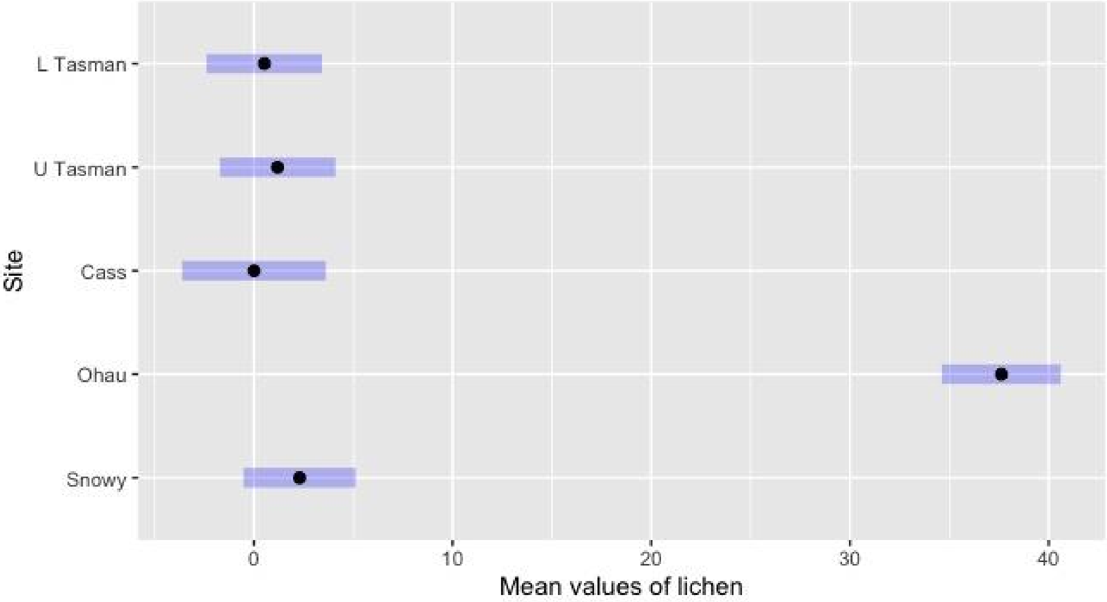


Figure A 13 All sites, except Ōhau, had very low levels of lichen in the ground cover.

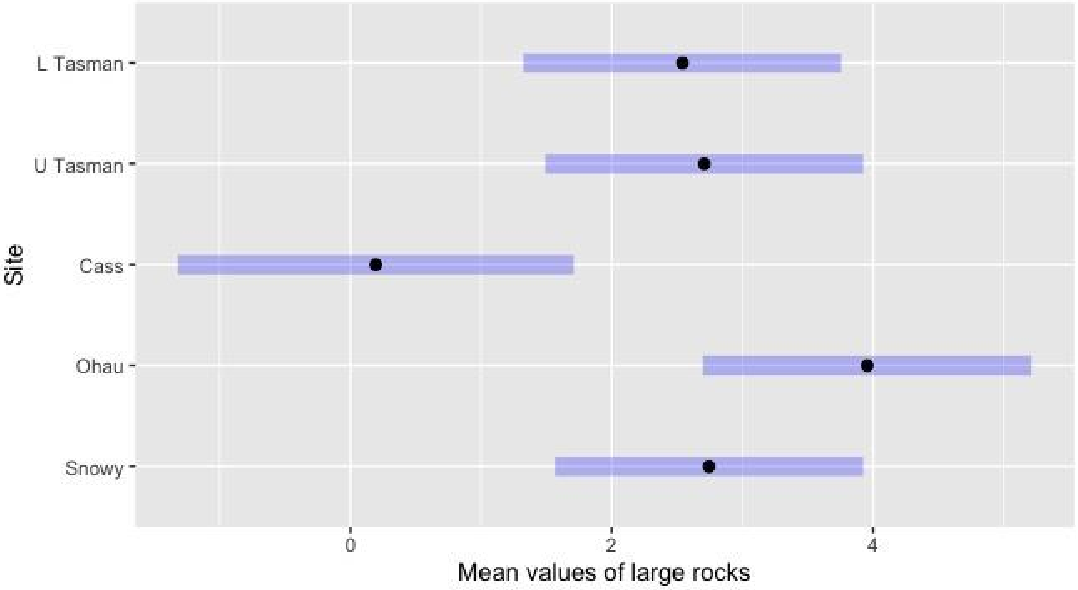


Figure A 14 Both Tasman sites and Snowy had similar means for the cover of large sized rocks. Cass had a lowest mean for large rock cover.

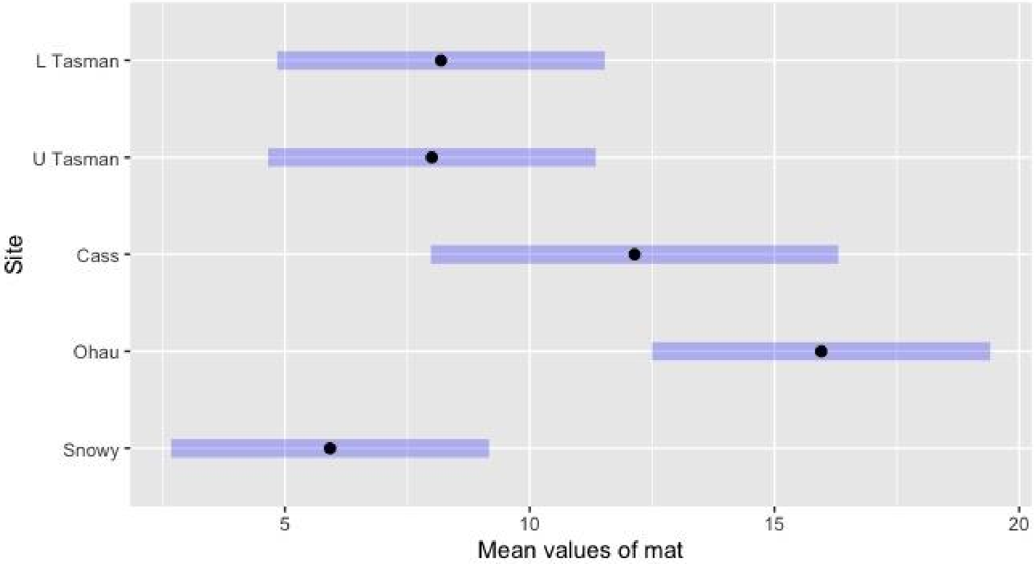


Figure A 15 The two Tasman River sites had similar means for mat forming plant cover, Snowy had the lowest mean.

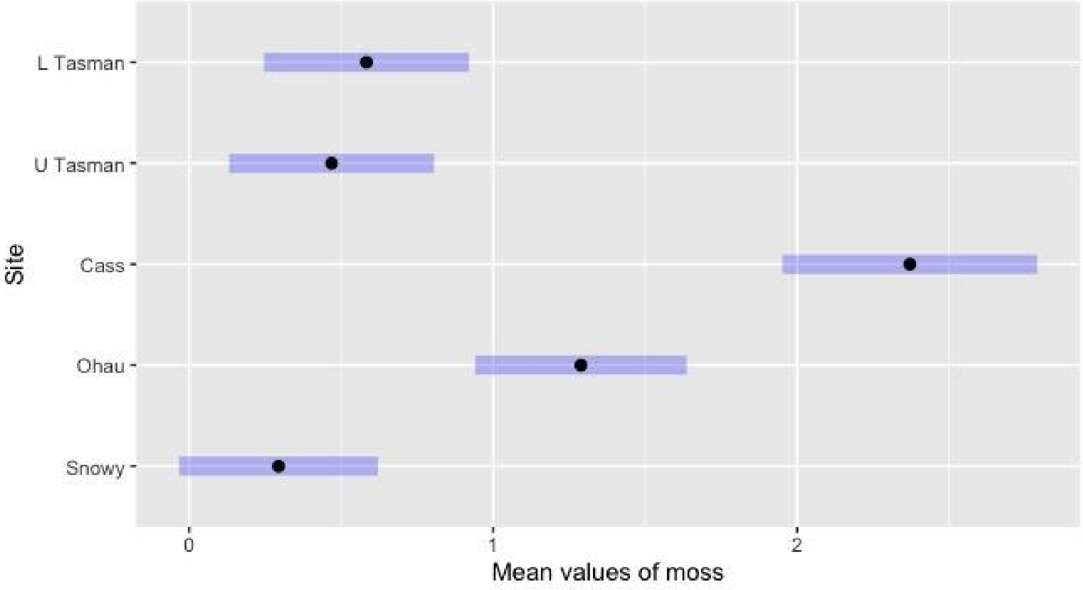


Figure A 16 Cass River had a higher mean value for the cover of moss than any of the other sites.

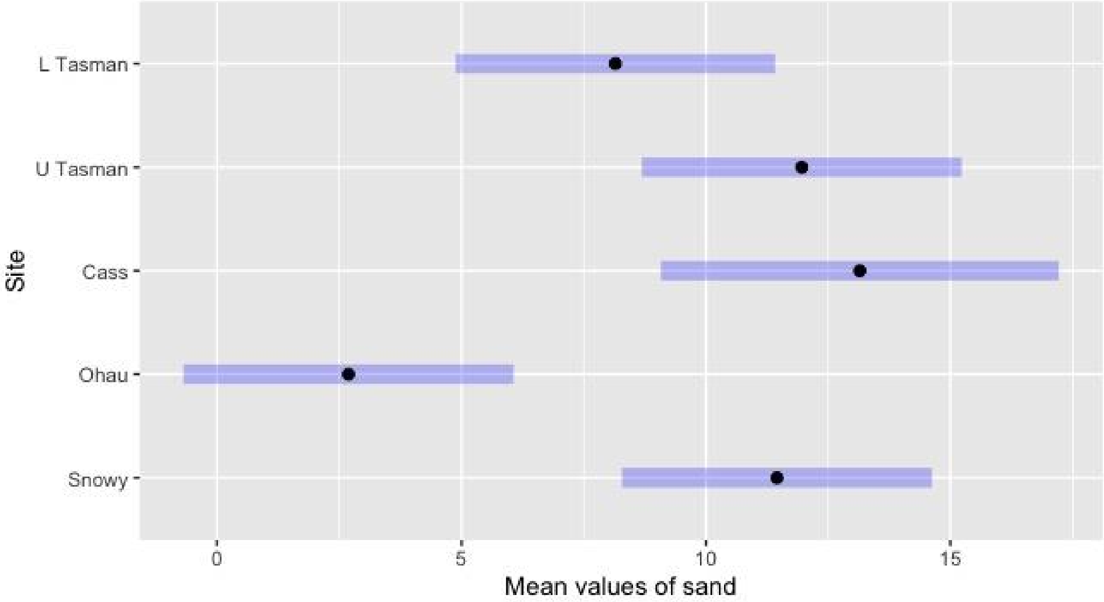


Figure A 17 Ōhau River had the lowest mean value for the cover of sand that any of the other sites.

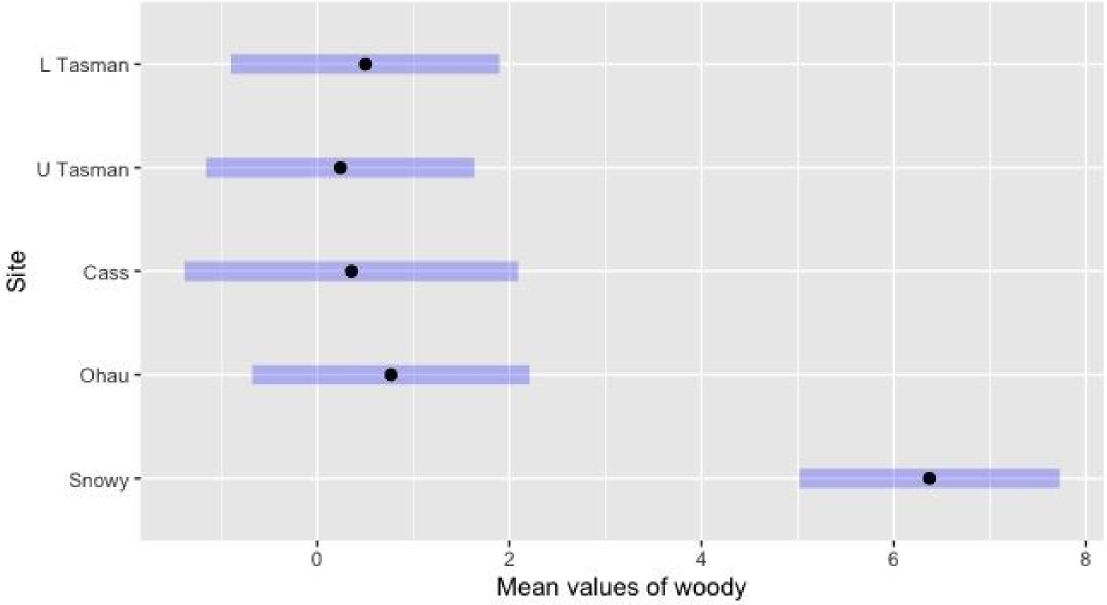


Figure A 18 All sites, except Snowy, had similar low mean values for woody vegetation cover.

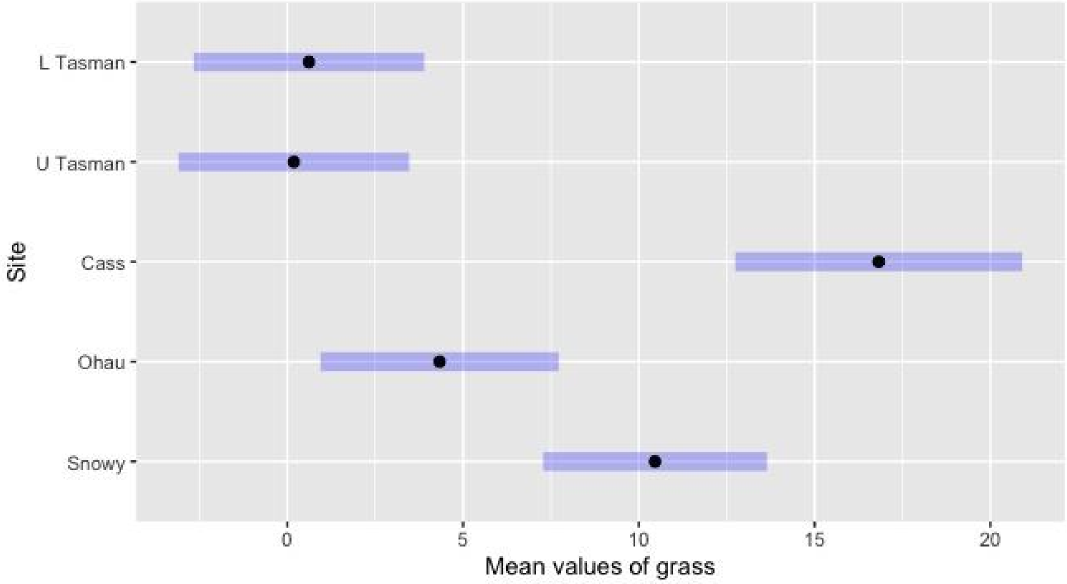


Figure A 19 Both Tasman River sites had similar mean values for grass cover, and Cass River had the highest mean value.

Appendix C

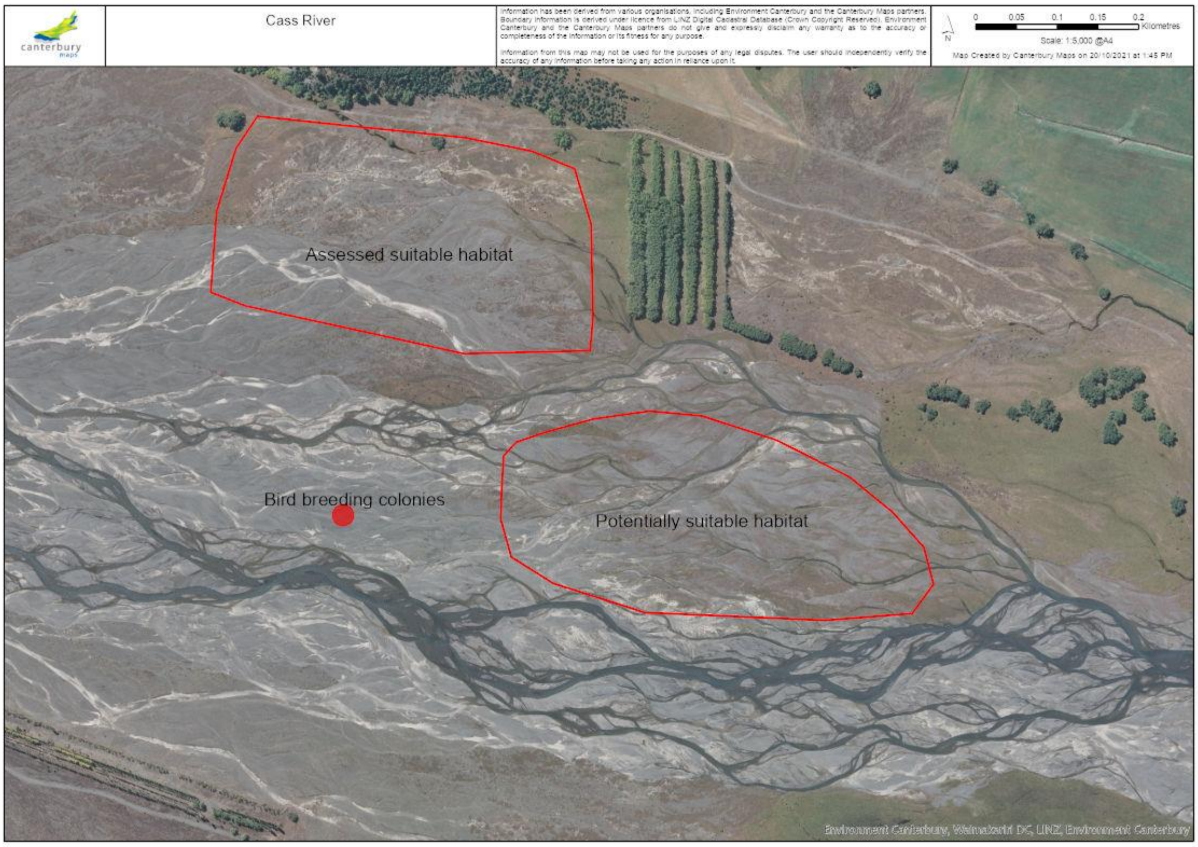


Figure A 20 The locations of the assessed suitable habitat for *B. robustus* at Cass River, and the potentially suitable habitat that was not assessed. The location of the breeding bird colonies is also marked on the map (red dot).

References

Allen, R. B., & Lee, W. G. (2006). *Biological invasions in New Zealand*. Berlin: Berlin : Springer.

Anderson, S., & Woolmore, C. (2009). Proposal for predator control to protect a black-fronted tern colony on an island in the upper Ohau River 2009–2013. *Department of Conservation, Twizel Regional Office*.

Arkins, A. M., Winnington, A. P., Anderson, S., & Clout, M. N. (1999). Diet and nectarivorous foraging behaviour of the short-tailed bat (Mystacina tuberculata). *J. Zoology, 247*(2), 183–187. doi:10.1111/j.1469-7998.1999.tb00982.x

Armstrong, D. P., Hayward, M. W., Moro, D., & Seddon, P. J. (2015). *Advances in reintroduction biology of Australian and New Zealand fauna*. Collingwood: CSIRO Publishing.

Armstrong, D. P., McArthur, N., Govella, S., Morgan, K., Johnston, R., Gorman, N., . . . Richard, Y. (2013). Using radio-tracking data to predict post-release establishment in reintroductions to habitat fragments. *Biological conservation, 168*, 152–160.

Armstrong, D. P., & Seddon, P. J. (2008). Directions in reintroduction biology. *Trends in Ecology & Evolution, 23*(1), 2–25. doi:10.1016/j.tree.2007.10.003

Bajomi, B., Pullin, A. S., Stewart, G. B., & Takács-Sánta, A. (2010). Bias and dispersal in the animal reintroduction literature. *Oryx, 44*(3), 358–365. doi:10.1017/S0030605310000281

Batcheler, C. (1967). Preliminary observations of alpine grasshoppers in a habitat modified by deer and chamois. *Proceedings of the New Zealand Ecological Society, 14*, 15–26.

Beirinckx, K., Van Gossum, H., Lajeunesse, M., & Forbes, M. (2006). Sex biases in dispersal and philopatry: insights from a meta-analysis based on capture–mark–recapture studies of damselflies. *Oikos, 113*(3), 539–547.

Bellis, J., Bourke, D., Williams, C., & Dalrymple, S. (2019). Identifying factors associated with the success and failure of terrestrial insect translocations. *Biological Conservation, 236*, 29–36.

Berger-Tal, O., Blumstein, D., & Swaisgood, R. R. (2020). Conservation translocations: a review of common difficulties and promising directions. *Animal Conservation, 23*(2), 121–131.

Besnard, A., Piry, S., Berthier, K., Lebreton, J. D., & Streiff, R. (2007). Modeling survival and mark loss in molting animals: Recapture, dead recoveries, and exuvia recoveries. *Ecology, 88*(2), 289–295. doi:10.1890/05-0811

Bigelow, R. S. (1967). *The grasshoppers (Acrididae) of New Zealand, their taxonomy and distribution*. Christchurch: University of Canterbury.

Bowie, M. (2001). *Restoration of Quail Island invertebrates*: Lincoln University. Ecology and Entomology Group: Lincoln, Canterbury.

Bowie, M., Hodge, S., Banks, J., & Vink, C. (2006). An appraisal of simple tree-mounted shelters for non-lethal monitoring of weta (Orthoptera: Anostostomatidae and Rhaphidophoridae) in New Zealand nature reserves. *Journal of Insect Conservation, 10*(3), 261–268. doi:10.1007/s10841-005-5594-3

Bowie, M. H., &. (2008). *Ecological restoration of the invertebrate fauna on Quail Island (Ōtamahua)* Hamilton, New Zealand: University of Waikato.

Bowie, M. H., Allen, W., McCaw, J., & van Heugten, R. A. (2013). Factors influencing occupancy of modified artificial refuges for monitoring the range-restricted Banks Peninsula tree weta Hemideina ricta (Anostostomatidae). *New Zealand Journal of Ecology*, 132–138.

Bowie, M. H., Kavermann, M., & Ross, J. G. (2011). The Quail Island story - thirteen years of multi-species pest control: successes, failures and lessons learnt. In *Island invasives: eradication and management* (pp. 157–161). Gland, Switzerland: IUCN.

Bowie, M. H., Madi Corodji, J., Ford, D., Ross, J. G., McLennan, I., Fewster, R. M., & Murphy, E. C. (2018). Mouse management on Ōtamahua/Quail Island-lessons learned. *New Zealand Journal of Zoology, 45*(3), 267–285. doi:10.1080/03014223.2018.1482929

Bröder, L., Tatin, L., Hochkirch, A., Schuld, A., Pabst, L., & Besnard, A. (2019). Optimization of capture-recapture monitoring of elusive species illustrated with a threatened grasshopper. *Conservation biology : the journal of the Society for Conservation Biology*. doi:10.1111/cobi.13449

Bubac, C. M., Johnson, A. C., Fox, J. A., & Cullingham, C. I. (2019). Conservation translocations and post-release monitoring: Identifying trends in failures, biases, and challenges from around the world. *Biological Conservation, 238*, 108239.

Buchvveitz, M., & Walter, R. (1992). Individualmarkierung von Heuschrecken-ein Erfahrungsbericht.

Caruso, B., Ross, A., Shuker, C., & Davies, T. (2013). Flood hydraulics and impacts on invasive vegetation in a braided river floodplain, New Zealand. *Environment and Natural Resources Research, 3*(1), 92.

Caruso, B. S. (2006). Project River Recovery: restoration of braided gravel-bed river habitat in New Zealand's high country. *Environmental Management, 37*(6), 840–861.

Caruso, B. S., Edmondson, L., & Pithie, C. (2013). Braided river flow and invasive vegetation dynamics in the Southern Alps, New Zealand. *Environmental management, 52*(1), 1–18.

Caruso, B. S., Pithie, C., & Edmondson, L. (2013). Invasive riparian vegetation response to flow regimes and flood pulses in a braided river floodplain. *Journal of environmental management, 125*, 156–168.

Chapman, R. F., & Joern, A. (1990). *Biology of grasshoppers*: John Wiley & Sons.

Cree, A., & Hare, K. M. (2016). Reproduction and life history of New Zealand lizards. In *New Zealand Lizards* (pp. 169–206): Springer.

Cromarty, P. L., & Alderson, S. L. (2013). Translocation statistics (2002–2010), and the revised Department of Conservation translocation process. *Notornis, 60*(1), 55–62.

Daugherty, C. H., Patterson, G. B., & Hitchmough, R. A. (1994). Taxonomic and conservation review of the New Zealand herpetofauna. *New Zealand Journal of Zoology, 21*(4), 317–323. doi:10.1080/03014223.1994.9518002

Dempster, J. (1963). The population dynamics of grasshoppers and locusts. *Biological Reviews, 38*(4), 490–529.

Department of Conservation, D. O. C. (2012). *Herpetofauna: artificial retreats*.

Dickens, M. J., Delehanty, D. J., & Romero, L. M. (2010). Stress: an inevitable component of animal translocation. *Biological Conservation, 143*(6), 1329–1341.

Duncan, M., Hughey, K. F., Cochrane, C., & Bind, J. (2008). River modelling to better manage mammalian predator access to islands in braided rivers.

Dunn, R. R. (2005). Modern Insect Extinctions, the Neglected Majority. *Conservation Biology, 19*(4), 1030–1036. doi:10.1111/j.1523-1739.2005.00078.x

Dwyer, P. (1960). New Zealand bats. *Tuatara, 8*(2), 61–71.

Easterling, D. R., Horton, B., Jones, P. D., Peterson, T. C., Karl, T. R., Parker, D. E., . . . Jamason, P. (1997). Maximum and minimum temperature trends for the globe. *Science, 277*(5324), 364–367.

Edström, T. (2012). *Effect of prey dorsal colour and pattern on detection by an avian predator*. Uppsala University, Uppsala, Sweden.

Field, L. H. (2001). *The biology of wetas, king crickets and their allies*. Wallingford, UK: CABI Publishing. doi:10.1079/9780851994086.0000

Forsman, A. (1999). Temperature influence on escape behaviour in two species of pygmy grasshoppers. *Ecoscience, 6*(1), 35–40.

Fraser, I. (1999). *Robust grasshopper (Brachaspis robustus) sightings database and monitoring methods trial. Project River Recovery Internal Report 99/01*. Twizel, New Zealand: Department of Conservation.

Gibbs, G. (2009). The end of an 80-million year experiment: a review of evidence describing the impact of introduced rodents on New Zealand's 'mammal-free' invertebrate fauna. *Biological Invasions, 11*(7), 1587–1593. doi:10.1007/s10530-008-9408-x

Gibbs, G. W. (1998). Why are some weta (Orthoptera: Stenopelmatidae) vulnerable yet others are common? *Journal of insect conservation, 2*(3), 161–166. doi:10.1023/A:1009660200402

Gibbs, G. W. (2010). Do New Zealand invertebrates reflect the dominance of birds in their evolutionary history? *New Zealand Journal of Ecology, 34*(1), 152–157.

Gillies, C., & Williams, D. (2013). DOC tracking tunnel guide v2. 5.2: using tracking tunnels to monitor rodents and mustelids. Department of Conservation. *Science & Capability Group, Hamilton, NZ*.

Gitay, H., Suárez, A., Watson, R. T., & Dokken, D. J. (Eds.). (2002). *Climate change and biodiversity*. Geneva, Switzerland.: Intergovernmental Panel on Climate Change.

Griffith, B., Scott, J. M., Carpenter, J. W., & Reed, C. (1989). Translocation as a species conservation tool: status and strategy. *Science (New York, N.Y.), 245*(4917), 477. doi:10.1126/science.245.4917.477

Hafernik, J. E. (1992). Threats to invertebrate biodiversity: implications for conservation strategies. In *Conservation biology* (pp. 171–195): Springer.

Hagler, J. R., & Jackson, C. G. (2001). Methods for marking insects: current techniques and future prospects. *Annual review of entomology, 46*, 511.

Hale Jr, M. E. (1959). Studies on lichen growth rate and succession. *Bulletin of the Torrey botanical club*, 126–129.

Hall, L. S., Krausman, P. R., & Morrison, M. L. (1997). The habitat concept and a plea for standard terminology. *Wildlife society bulletin*, 173–182.

Hare, K. M., Chapple, D. G., Towns, D. R., & van Winkel, D. (2016). The ecology of New Zealand's lizards. In D. G. Chapple (Ed.), *New Zealand Lizards* (pp. 133–168): Springer.

Hill, J., & Daniel, M. (1985). *Systematics of the New Zealand short-tailed bat Mystacina Gray, 1843 (Chiroptera: Mystacinidae)*: British Museum (Natural History).

Hoare, J. M. (2006). *Novel predators and naïve prey: how introduced mammals shape behaviours and populations of New Zealand lizards*. Victoria University., Wellington, New Zealand.

Hoare, J. M., Adams, L. K., Bull, L. S., & Towns, D. R. (2007). Attempting to Manage Complex Predator-Prey Interactions Fails to Avert Imminent Extinction of a Threatened New Zealand Skink Population. *Journal of Wildlife Management, 71*(5), 1576–1584. doi:10.2193/2006-488

Hoegh-Guldberg, O., Hughes, L., McIntyre, S., Lindenmayer, D., Parmesan, C., Possingham, H. P., & Thomas, C. (2008). Assisted colonization and rapid climate change. *Science, 321*(5887), 345–346.

Horikoshi, M., & Tang, Y. (2016). ggfortify: Data Visualization Tools for Statistical Analysis Results. Retrieved from https://CRAN.R-project.org/package=ggfortify

Huey, R. B., Peterson, C. R., Arnold, S. J., & Porter, W. P. (1989). Hot rocks and not-so-hot rocks: retreat-site selection by garter snakes and its thermal consequences. *Ecology, 70*(4), 931–944.

Hughey, K. F. (1997). The diet of the Wrybill (*Anarhynchus frontalis*) and the Banded Dotterel (*Charadrius bicinctus*) on two braided rivers in Canterbury, New Zealand. *Notornis, 44*(3), 185–193.

IUCN. (1998). *IUCN Guidelines for Re-introductions*. Gland, Switzerland.: IUCN/SSC Reintroduction Specialist Group.

IUCN. (2013). IUCN Guidelines for Reintroductions and Other Conservation Translocations. Version 1.0. *Gland, Switzerland: IUCN Species Survival Commission Reintroduction Specialist Group*.

Jamieson, I. G., Forbes, M. R., & McKnight, E. B. (2000). Mark-recapture study of mountain stone weta *Hemideina maori* (Orthoptera: Anostostomatidae) on rock tor'islands'. *New Zealand Journal of Ecology*, 209–214.

Janks, M. R., & Barker, N. P. (2013). Using Mark-recapture to provide population census data for use in Red Listing of invertebrates: the rare terrestrial snail Prestonella bowkeri as a case study. *Biodiversity and Conservation*, *22*(8), 1609–1621.

Jones, C., Moss, K., & Sanders, M. (2005). Diet of hedgehogs (*Erinaceus europaeus*) in the upper Waitaki Basin, New Zealand : implications for conservation. *New Zealand Journal of Ecology, 29*(1), 29–35.

Kassambara, A. (2020). ggpubr: 'ggplot2' Based Publication Ready Plots. R package version 0.4.0. . Retrieved from https://CRAN.R-project.org/package=ggpubr

Kassambara, A., & Mundt, F. (2020). factoextra: Extract and Visualize the Results of Multivariate Data Analyses. R package version 1.0.7. . Retrieved from https://CRAN.R-project.org/package=factoextra

Keedwell, R. J. (2002). *Black-fronted terns and banded dotterels: causes of mortality and comparisons of survival.* Massey University, Palmerston North, New Zealand.

Keedwell, R. J. (2005). Breeding biology of Black-fronted Terns (*Sterna albostriata*) and the effects of predation. *Emu-Austral Ornithology, 105*(1), 39–47.

Keesing, V., & Wratten, S. D. (1998). Indigenous invertebrate components in ecological restoration in agricultural landscapes *New Zealand Journal of Ecology, 22*(1), 99–104.

Kellner, C. J., Brawn, J. D., & Karr, J. R. (1992). What is habitat suitability and how should it be measured? In *Wildlife 2001: populations* (pp. 476–488): Springer.

Kemp, L., Norbury, G., Groenewegen, R., & Comer, S. (2015). The roles of trials and experiments in fauna reintroduction programs. *Advances in reintroduction biology of Australian and New Zealand fauna*, 73–90.

King, C., &. (2005). *The Handbook of New Zealand Mammals.: Oxford University Press*: USA.

King, C. M., Veale, A., Patty, B., & Hayward, L. (2014). Swimming capabilities of stoats and the threat to inshore sanctuaries. *Biological Invasions, 16*(5), 987–995.

Kissling, D. W., Pattemore, D. E., & Hagen, M. (2014). Challenges and prospects in the telemetry of insects. *Biological Reviews, 89*(3), 511–530.

Knox, C., Jarvie, S., Easton, L., & Monks, J. (2017). Soft-release, but not cool winter temperatures, reduces post-translocation dispersal of jewelled geckos. *Journal of Herpetology, 51*(4), 490–496.

Knudsen, E., Lindén, A., Both, C., Jonzén, N., Pulido, F., Saino, N., . . . Ergon, T. (2011). Challenging claims in the study of migratory birds and climate change. *Biological Reviews, 86*(4), 928–946.

Koot, E. M. (2018). *The ecology and evolution of New Zealand's endemic alpine grasshoppers.* (Doctoral). Massey University, Wellington, New Zealand Retrieved from http://hdl.handle.net/10179/14558

Koot, E. M., Morgan-Richards, M., & Trewick, S. A. (2020). An alpine grasshopper radiation older than the mountains, on Kā Tiritiri o te Moana (Southern Alps) of Aotearoa (New Zealand). *Molecular Phylogenetics and Evolution, 147*, 106783.

Lalas, C. (1977). *Food and feeding behaviour of the Black-fronted tern, Chlidonias hybrida albostriatus*. University of Otago.

Lenth, R. (2021). emmeans: Estimated Marginal Means, aka Least-Squares Means. R package version 1.7.0. . Retrieved from https://CRAN.R-project.org/package=emmeans

Lettink, M., & Monks, J. M. (2016). Survey and monitoring methods for New Zealand lizards. *Journal of the Royal Society of New Zealand: New Zealand herpetology: Tony Whitaker's legacy. Guest Editors: Don Newman and Kelly Hare, 46*(1), 16–28. doi:10.1080/03036758.2015.1108343

Letty, J., Marchandeau, S., & Aubineau, J. (2007). Problems encountered by individuals in animal translocations: lessons from field studies. *Ecoscience, 14*(4), 420–431.

Lloyd, B. D., & Powlesland, R. G. (1994). The decline of kakapo *Strigops habroptilus* and attempts at conservation by translocation. *Biological Conservation, 69*(1), 75–85. doi:10.1016/0006-3207(94)90330-1

Lövei, G. L., Stringer, I. A., Devine, C. D., & Cartellieri, M. (1997). Harmonic radar-a method using inexpensive tags to study invertebrate movement on land. *New Zealand Journal of Ecology*, 187–193.

Maag, N., Karpati, T., & Bollmann, K. (2013). Semi-natural river system maintains functional connectivity and gene flow of the critically endangered gravel grasshopper (*Chorthippus pullus*). *Biological Conservation, 158*, 88–97.

Macara, G., Woolley, J.-M., Zammit, C., Pearce, P., Stuart, S., Wadhwa, S., . . . Collins, D. (2019). *Climate change projections for the Otago Region*. Auckland, NZ: NIWA.

Macara, G. R. (2016). *The climate and weather of Canterbury*: NIWA, Taihoro Nukurangi.

Mason, P. C. (1971). *Alpine grasshoppers (Orthoptera: Acrididae) in the southern alps of Canterbury, New Zealand*. University of Canterbury, Christchurch, New Zealand.

McAulay, J., Seddon, P. J., Wilson, D. J., & Monks, J. M. (2020). Stable isotope analysis reveals variable diets of stoats (Mustela erminea) in the alpine zone of New Zealand. *New Zealand Journal of Ecology, 44*(2), 1–13.

McCartney, J., Armstrong, D. P., Gwynne, D. T., Kelly, C. D., & Barker, R. J. (2006). Estimating abundance, age structure and sex ratio of a recently discovered New Zealand tusked weta *Motuweta riparia* (Orthoptera, Anostostomatidae), using mark-recapture analysis. *New Zealand Journal of Ecology*, 229–235.

McGuinness, C. A. (2001). *The conservation requirements of New Zealand's nationally threatened invertebrates*. Wellington, N.Z.: Wellington, N.Z. : Biodiversity Recovery Unit, Dept. of Conservation.

McHalick, O. (1998). *Translocation database summary*. Wellington, N.Z.: Wellington, N.Z. : Biodiversity Recovery Unit, Dept. of Conservation.

McIntyre, M., &. (2001). *The ecology of some large weta species in New Zealand* UK: CABI: Wallingford, UK.

McIver, L. R. (2020). *Multi-trophic effects of predator exclusion fencing for the robust grasshopper (Brachaspis robustus): implications for invertebrate conservation*. University of Canterbury, Christchurch, New Zealand.

McKay, J., Anderson, A., Light, C., & Hancock, B. (1978). Tekapo B power project, construction report, part II—canal earthworks. . *Ministry of Works and Development. New Zealand*

Menzel, A., & Fabian, P. (1999). Growing season extended in Europe. *Nature, 397*(6721), 659–659.

Metcalfe, D. C., & Ridgeway, P. A. (2013). A case of web entanglement and apparent predation of the skink Lampropholis delicata (De Vis, 1888)(Sauria: Scincidae: Lygosominae) by the red-back spider Latrodectus hasseltii Thorell, 1870 (Aranea [sic]: Araneomorpha: Theridiidae) in an autochthonous mesic habitat in coastal southeast Australia. *Herpetology Notes, 6*, 375–377.

Meyer, D. B. (1977). The avian eye and its adaptations. In *The visual system in vertebrates* (pp. 549–611): Springer.

Miskelly, C. M., Dowding, J. E., Elliott, G. P., Hitchmough, R. A., Powlesland, R. G., Robertson, H. A., . . . Taylor, G. A. (2008). Conservation status of New Zealand birds, 2008. *Notornis, 55*(3), 117–135.

Miskelly, C. M., Tennyson, A. J., & Bishop, C. R. (2018). New locality records for two species of protected weevils, *Anagotus fairburni* (Brookes, 1932) and *Hadramphus stilbocarpae* Kuschel, 1971 (Coleoptera: Curculionidae), from southern Fiordland, New Zealand. *Tuhinga: Records of the Museum of New Zealand Te Papa Tongarewa*, 20.

Morris, S. D., Brook, B. W., Moseby, K. E., & Johnson, C. N. (2021). Factors affecting success of conservation translocations of terrestrial vertebrates: a global systematic review. *Global Ecology and Conservation*, e01630.

Murphy, E. C., Keedwell, R. J., Brown, K. P., & Westbrooke, I. (2004). Diet of mammalian predators in braided river beds in the central South Island, New Zealand. *Wildlife Research, 31*(6), 631. doi:10.1071/WR03033

Mussgnug, G. L. (1972). *The structure and performance of an adult population of Aulocara elliotti (Thomas)(Orthoptera, Acrididae) near Billings, Montana*. Montana State University-Bozeman, College of Agriculture, Montana, USA.

Narisu, Lockwood, J. A., & Schell, S. P. (1999). A novel mark-recapture technique and its application to monitoring the direction and distance of local movements of rangeland grasshoppers (Orthoptera: Acrididae) in the context of pest management. *Journal of Applied Ecology*, 604–617.

Norbury, G. (2001). Conserving dryland lizards by reducing predator-mediated apparent competition and direct competition with introduced rabbits. *Journal of Applied Ecology, 38*(6), 1350–1361. doi:10.1046/j.0021-8901.2001.00685.x

Norbury, G., Heyward, R., & Parkes, J. (2009). Skink and invertebrate abundance in relation to vegetation, rabbits and predators in a New Zealand dryland ecosystem. *New Zealand Journal of Ecology*, 24–31.

Norbury, G., Hutcheon, A., Reardon, J., & Daigneault, A. (2014). Pest fencing or pest trapping: A bio-economic analysis of cost-effectiveness. *Austral Ecology, 39*(7), 795–807. doi:10.1111/aec.12147

Norbury, G., Van Den Munckhof, M., Neitzel, S., Hutcheon, A., Reardon, J., & Ludwig, K. (2014). Impacts of invasive house mice on post-release survival of translocated lizards. *New Zealand Journal of Ecology, 38*(2), 322–327.

Northcroft, M. A. (1967). *The immature stages of Phaulacridium marginale (Walker) and Sigaus campestris (Hutton)(Orthoptera: Acrididae)*. University of Canterbury, Christchurch, New Zealand.

Norton, D. A. (2009). Species invasions and the limits to restoration: learning from the New Zealand experience. *Science (New York, N.Y.), 325*(5940), 569. doi:10.1126/science.1172978

Nunney, L., & Campbell, K. A. (1993). Assessing minimum viable population size: Demography meets population genetics. *Trends in Ecology & Evolution, 8*(7), 234–239. doi:10.1016/0169-5347(93)90197-W

NZTCS. (2021). *Current Statistics*. Retrieved April 4, 2021,from https://nztcs.org.nz/.

O'Donnell, C. F. (2000). *The significance of river and open water habitats for indigenous birds in Canterbury, New Zealand.* Christchurch, New Zealand.: Environment Canterbury.

O'Donnell, C. F., Sanders, M. D., Woolmore, C. B., & Maloney, R. (2016). *Management and research priorities for conserving biodiversity on New Zealand's braided rivers*. Wellington, New Zealand.: Department of Conservation.

O'neill, S. B., Chapple, D. G., Daugherty, C. H., & Ritchie, P. A. (2008). Phylogeography of two New Zealand lizards: McCann's skink (*Oligosoma maccanni*) and the brown skink (*O. zelandicum*). *Molecular Phylogenetics and Evolution, 48*(3), 1168–1177. doi:10.1016/j.ympev.2008.05.008

Parker, K. A., Adams, L., Baling, M., Kemp, L., Kuchling, G., Lloyd, B., . . . Watts, C. (2015). Practical guidelines for planning and implementing fauna translocations. *Advances in Reintroduction Biology of Australian and New Zealand Fauna*, 255.

Parmesan, C., Ryrholm, N., Stefanescu, C., Hill, J. K., Thomas, C. D., Descimon, H., . . . Tammaru, T. (1999). Poleward shifts in geographical ranges of butterfly species associated with regional warming. *Nature, 399*(6736), 579–583.

Parmesan, C., & Yohe, G. (2003). A globally coherent fingerprint of climate change impacts across natural systems. *nature, 421*(6918), 37–42.

Pearce-Kelly, P., Jones, R., Clarke, D., Walker, C., Atkin, P., & Cunningham, A. (1998). The captive rearing of threatened Orthoptera: a comparison of the conservation potential and practical considerations of two species' breeding programmes at the Zoological Society of London. *Journal of Insect Conservation, 2*(3), 201–210.

Poniatowski, D., & Fartmann, T. (2010). What determines the distribution of a flightless bush-cricket (*Metrioptera brachyptera*) in a fragmented landscape? *Journal of Insect Conservation, 14*(6), 637–645.

Prugh, L. R., Stoner, C. J., Epps, C. W., Bean, W. T., Ripple, W. J., Laliberte, A. S., & Brashares, J. S. (2009). The rise of the mesopredator. *Bioscience, 59*(9), 779–791.

Psychoudakis, D., Moulder, W., Chen, C.-C., Zhu, H., & Volakis, J. L. (2008). A portable low-power harmonic radar system and conformal tag for insect tracking. *IEEE Antennas and Wireless Propagation Letters, 7*, 444–447.

R Core Team, R. (2021). R: A language and environment for statistical computing. Vienna, Austria: R Foundation for Statistical Computing. Retrieved from https://www.R-project.org/

Ramsay, G., &. (1978). *A review of the effect of rodents on the New Zealand invertebrate fauna*: Department of Lands and Survey Wellington,, New Zealand.

Ramsey, D. S., & Norbury, G. L. (2009). Predicting the unexpected: using a qualitative model of a New Zealand dryland ecosystem to anticipate pest management outcomes. *Austral Ecology, 34*(4), 409–421.

Reardon, J. T., Whitmore, N., Holmes, K. M., Judd, L. M., Hutcheon, A. D., Norbury, G., & Mackenzie, D. I. (2012). Predator control allows critically endangered lizards to recover on mainland New Zealand. *New Zealand Journal of Ecology, 36*(2), 141–150.

Ricciardi, A., & Simberloff, D. (2009). Assisted colonization is not a viable conservation strategy. *Trends in ecology & evolution, 24*(5), 248–253.

Richardson, K. M., Doerr, V., Ebrahimi, M., Lovegrove, T. G., & Parker, K. A. (2015). Considering dispersal in reintroduction and restoration planning. *Advances in reintroduction biology of Australian and New Zealand fauna*, 59–72.

Rout, T. M., McDonald-Madden, E., Martin, T. G., Mitchell, N. J., Possingham, H. P., & Armstrong, D. P. (2013). How to decide whether to move species threatened by climate change. *PloS one, 8*(10), e75814.

Sales, J. (2005). The endangered kiwi: A review. *Folia zoologica (Brno), 54*(1–2), 1–20.

Sanders, M. D., & Maloney, R. F. (2002). Causes of mortality at nests of ground-nesting birds in the Upper Waitaki Basin, South Island, New Zealand: a 5-year video study. *Biological conservation, 106*(2), 225–236.

Schlesselmann, A.-K. V., O'Donnell, C. F., Monks, J. M., & Robertson, B. C. (2018). Clearing islands as refugia for black-fronted tern (*Chlidonias albostriatus*) breeding colonies in braided rivers. *New Zealand Journal of Ecology, 42*(2), 137–148.

Schori, J., Steeves, T., & Murray, T. (2020a). Informing the design of a long-term population density monitoring protocol for a Nationally Endangered grasshopper: removal sampling as a basis for estimating individual detection probabilities. *Journal of Insect Conservation, 24*(5), 841–851.

Schori, J. C. (2020). *Improving the success of insect conservation translocations: A case study of the Nationally Endangered robust grasshopper (Brachaspis robustus Bigelow)*. University of Canterbury, Christchurch, New Zealand.

Schori, J. C., Maloney, R. F., Steeves, T. E., & Murray, T. J. (2019). Evidence that reducing mammalian predators is beneficial for threatened and declining New Zealand grasshoppers. *New Zealand Journal of Zoology, 46*(2), 149–164. doi:10.1080/03014223.2018.1523201

Schori, J. C., Steeves, T. E., & Murray, T. J. (2020b). Designing monitoring protocols to measure population trends of threatened insects: A case study of the cryptic, flightless grasshopper Brachaspis robustus. *Plos one, 15*(9), e0238636.

Seddon, P., Griffiths, C. J., Soorae, P. S., & Armstrong, D. (2014). Reversing defaunation: Restoring species in a changing world. *Science, 345*(6195), 406–412. doi:10.1126/science.1251818

Seddon, P. J. (2010). From reintroduction to assisted colonization: moving along the conservation translocation spectrum. *Restoration Ecology, 18*(6), 796–802.

Seddon, P. J., Armstrong, D. P., & Maloney, R. F. (2007). Developing the science of reintroduction biology. *Conservation Biology, 21*(2), 303.

Seddon, P. J., Moro, D., Mitchell, N. J., Chauvenet, A. L., & Mawson, P. R. (2015). Proactive conservation or planned invasion? Past, current and future use of assisted colonisation. *Advances in Reintroduction Biology of Australian and New Zealand Fauna*, 105–126.

Seddon, P. J., Soorae, P. S., & Launay, F. (2005). Taxonomic bias in reintroduction projects*Cambridge University Press.* Symposium conducted at the meeting of the Animal Conservation Forum

Sheean, V. A., Manning, A. D., & Lindenmayer, D. B. (2012). An assessment of scientific approaches towards species relocations in Australia. *Austral Ecology, 37*(2), 204–215.

Sherley, G. (1998). Translocating a threatened New Zealand giant orthopteran, *Deinacrida sp.* (Stenopelmatidae): some lessons. *Journal of Insect Conservation, 2*(3–4), 195–199. doi:10.1023/A:1009664301311

Sherley, G., & Hayes, L. (1993). The conservation of a giant weta (*Deinacrida n. sp*. Orthoptera: Stenopelmatidae) at Mahoenui, King Country: habitat use, and other aspects of its ecology. *New Zealand Entomologist, 16*(1), 55–68.

Sherley, G., Stringer, I., & Parrish, G. R., & C. New Zealand. Department of. (2010). *Summary of native bat, reptile, amphibian and terrestrial invertebrate translocations in New Zealand* Wellington, New Zealand.: Wellington, N.Z: Pub. Team, Dept. of Conservation.

St Clair, J. J. H. (2011). The impacts of invasive rodents on island invertebrates. *Biological Conservation, 144*(1), 68–81. doi:10.1016/j.biocon.2010.10.006

Staples, D. J. (1967). *Colour and size variation within a population of Brachaspis collinus (Hutton)(Orthoptera: Acrididae)*. University of Canterbury., Christchurch, New Zealand.

Stratton, G. E. (1984). Behavioral studies of wolf spiders: a review of recent research. *Revue arachnologique, 6*(2), 57–70.

Stringer, I., & Chappell, R. (2008). Possible rescue from extinction: transfer of a rare New Zealand tusked weta to islands in the Mercury group. *An international journal devoted to the conservation of insects and related invertebrates, 12*(3), 371–382. doi:10.1007/s10841-008-9149-2

Stringer, I., & Grant, E. A., & C. New Zealand. Department of. (2007). *Captive rearing and biology of the endangered giant land snails Placostylus ambagiosus and P. hongii (Pulmonata: Bulimulidae)* Wellington, N.Z: Wellington, N.Z: Science and Technical Pub., Dept. of Conservation.

Stringer, I., Parrish, G., & Sherley, G. (2018). Homing, dispersal and mortality after translocation of long-lived land snails *Placostylus ambagiosus* and *P. hongii* (Gastropoda: Bothriembryontidae) in New Zealand. *Molluscan Research, 38*(1), 56–76. doi:10.1080/13235818.2017.1323368

Stringer, I., Watts, C., Thornburrow, D., Chappell, R., & Price, R. (2014). Saved from extinction? Establishment and dispersal of Mercury Islands tusked weta, *Motuweta isolata*, following translocation onto mammal-free islands. *Journal of Insect Conservation, 18*(2), 203–214. doi:10.1007/s10841-014-9631-y

Te Manahuna Aoraki, T. M. A. (2020). *Te Manahuna Aoraki Annual Report 2020*. Twizel, New Zealand. .

Toro, M., & Caballero, A. (2005). Characterization and conservation of genetic diversity in subdivided populations. *Philosophical Transactions of the Royal Society B: Biological Sciences, 360*(1459), 1367–1378.

Towns, D. R. (1994). The role of ecological restoration in the conservation of Whitaker's skink (*Cyclodina whitakeri*), a rare New Zealand lizard (Lacertilia: Scincidae). *New Zealand Journal of Zoology, 21*(4), 457–471. doi:10.1080/03014223.1994.9518016

Trewick, S. (2001). Identity of an endangered grasshopper (Acrididae: Brachaspis): Taxonomy, molecules and conservation. *Conservation Genetics, 2*(3), 233–243. doi:10.1023/A:1012263717279

Van Geystelen, A., Benaets, K., de Graaf, D. C., Larmuseau, M., & Wenseleers, T. (2016). Track-a-Forager: a program for the automated analysis of RFID tracking data to reconstruct foraging behaviour. *Insectes Sociaux, 63*(1), 175–183.

Vink, C. J., et al. (2011). The invasive Australian redback spider, *Latrodectus hasseltii* Thorell 1870 (Araneae: Theridiidae): current and potential distributions, and likely impacts. *Biological Invasions*, 3(4): 1003–1019. doi.org/10.1007/s10530-010-9885-6.

Watts, C., Empson, R., Thornburrow, D., & Rohan, M. (2012). Movements, behaviour and survival of adult Cook Strait giant weta (*Deinacrida rugosa*; Anostostomatidae: Orthoptera) immediately after translocation as revealed by radiotracking. *Journal of Insect Conservation, 16*(5), 763–776.

Watts, C., Stringer, I., Sherley, G., Gibbs, G., & Green, C. (2008). History of weta (Orthoptera: Anostostomatidae) translocation in New Zealand: lessons learned, islands as sanctuaries and the future. *Journal of Insect Conservation, 12*(3), 359-370. doi:10.1007/s10841-008-9154-5

Watts, C., & Thornburrow, D. (2009). Where have all the weta gone? Results after two decades of transferring a threatened New Zealand giant weta, *Deinacrida mahoenui*. *Journal of Insect Conservation, 13*(3), 287–295. doi:10.1007/s10841-008-9170-5

Wei, T., & Simko, V. (2021). R package 'corrplot': Visualization of a Correlation Matrix (Version 0.92). . Retrieved from https://github.com/taiyun/corrplot

Weyer, J., Weinberger, J., & Hochkirch, A. (2012). Mobility and microhabitat utilization in a flightless wetland grasshopper, *Chorthippus montanus* (Charpentier, 1825). *Journal of Insect Conservation, 16*(3), 379–390.

White, E. G. (1994). *Ecological research and monitoring of the protected grasshopper Brachaspis robustus in the Mackenzie Basin*. Wellington, N.Z.: Wellington, N.Z. : Dept. of Conservation.

Whyte, R., & Anderson, G. (2017). *A field guide to spiders of Australia*. Victoria, Australia: CSIRO Publishing.

Wikelski, M., Kays, R. W., Kasdin, N. J., Thorup, K., Smith, J. A., & Swenson Jr, G. W. (2007). Going wild: what a global small-animal tracking system could do for experimental biologists. *Journal of Experimental Biology, 210*(2), 181–186.

Willott, S. (1997). Thermoregulation in four species of British grasshoppers (Orthoptera: Acrididae). *Functional Ecology*, 705–713.

Winks, C. J. (2002). *Captive-rearing of the Middle Island tusked weta*. Wellington, N.Z.: Wellington, N.Z. : Dept. of Conservation.

Woolmore, C. B. (2011). *The vegetation of braided rivers in the Upper Waitaki basin: South Canterbury, New Zealand*: Canterbury Conservancy, Department of Conservation.

Xiao, N. (2018). ggsci: Scientific Journal and Sci-Fi Themed Color Palettes for 'ggplot2'. R package version 2.9. . Retrieved from https://CRAN.R-project.org/package=ggsci

Young, R., Smart, G., & Harding, J. (2004). Impacts of hydro-dams, irrigation schemes and river control works. *Freshwaters of New Zealand*, 37.31–37.15.