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*Translocation of the Eastern Bristlebird and factors
associated with a successful program*

A thesis submitted in fulfilment of the requirements for the degree of Doctor of
Philosophy from the University of Wollongong

By

David Bain, BSc (Hons)

School of Biological Sciences

2006

Declaration

This thesis is submitted in accordance with the regulations of the University of Wollongong in fulfilment of the requirements of the degree of Doctor of Philosophy. The research carried out for this thesis was conducted in accordance with the following

permits:

University of Wollongong animal ethics: AE02/10

Department of Environment and Conservation scientific permit: S10166

Department of the Environment and Heritage threatened species permit: E2002-35012

Booderee National Park research permit: BDR04/00019

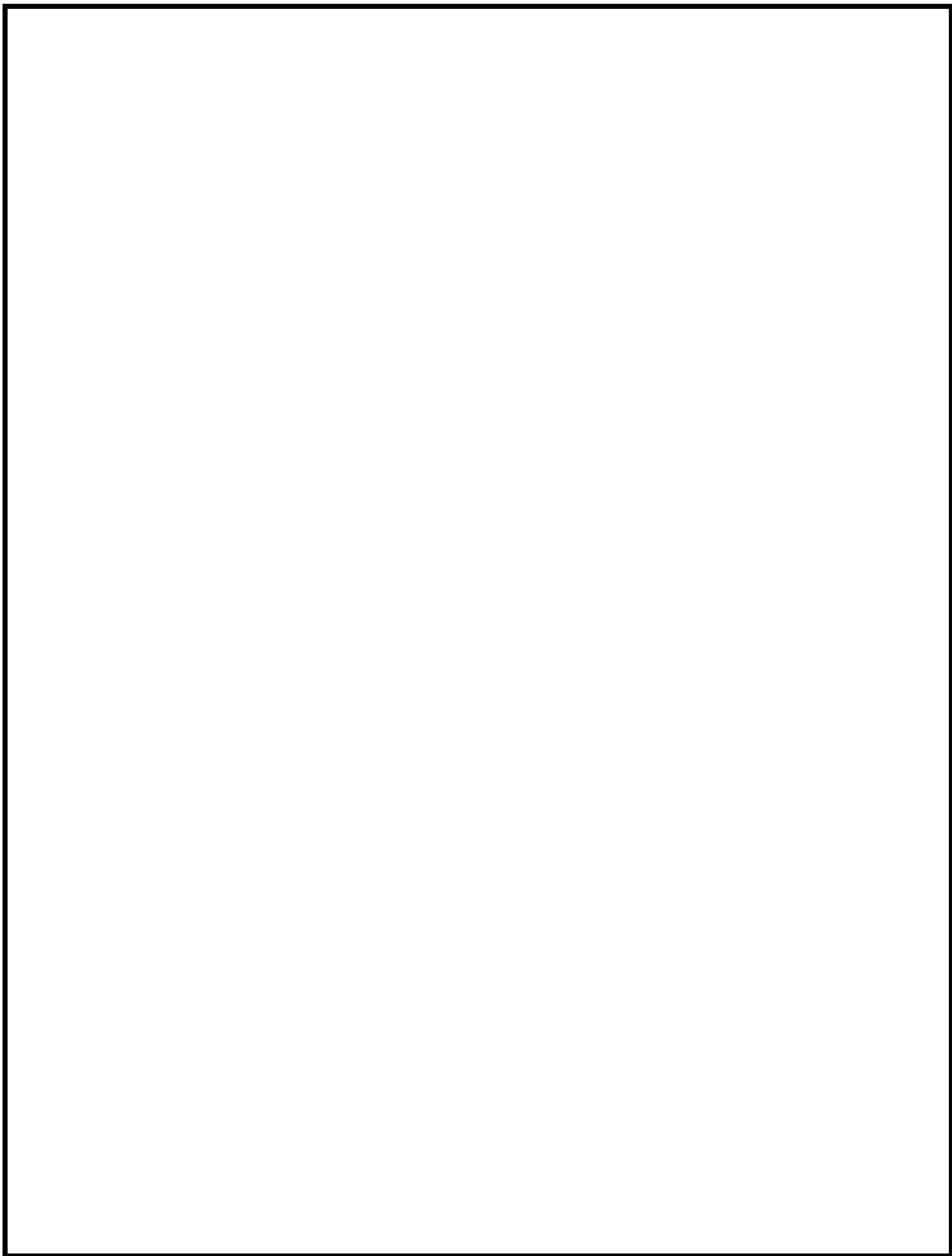
Beecroft Peninsula research permit: 4/02

The work in this thesis has been carried out by me and has not been submitted to any other university or institution.



David Bain

Date 08/05/2007



Acknowledgements

I first met the Eastern Bristlebird on a field trip as an undergraduate student at the University of Wollongong. I should say heard it, as an excited NSW National Parks and Wildlife Service officer smiled broadly and pointed enthusiastically in the general direction of a bird call. “Pretty birdie”, “Pretty birdie” he kept repeating. Little did I know that this bird and the excited Parks officer pointing at it would form the core of my post-graduate studies three years later.

When the idea of post-graduate study was first raised with me by Rob Whelan and Jack Baker, I was not at all confident. Do I really have it in me? I have always been passionate about the natural world and with some encouragement set about getting my hands and brain dirty with ecology and conservation.

First and foremost the support of my supervisors has been amazing. They are all passionate about ecology and conservation and all provided knowledge, wisdom, support and encouragement throughout my studies. Associate Professor Kris French (University of Wollongong) always had an open door, most often resulting in new doors being opened in my work. Professor Rob Whelan (University of Wollongong) always challenged me, asking questions and providing a platform for me to find my own way to an answer. Dr Jack Baker (NSW Department of Environment and Conservation) provided buckets of advice on all aspects of my work, from investigating ecological questions to how to talk to a television reporter.

Nothing would have happened in this project had it not been for my field assistants and volunteers, in particular, Jean Clarke and Jodie Dunn. Jean was with me on the roller-coaster ride from the pure elation of capturing a bristlebird to the desperate lows of their escape. Jean not only looked after the witchcraft necessary to catch a bristlebird, but also the well being of the rest of the team. Jodie awoke every morning to a 40 minute drive before sunrise, dodging wildlife on her way to keeping track of all the translocated birds.

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A work-horse of the team, Jodie regularly put in 14 hour days - as long as the food was kept in steady supply! There were numerous other people who volunteered their time and their assistance does not go unrecognised. It is comforting to know there are people willing to give their time to help in the conservation of the natural world.

There are many people from the various agencies involved with this project that need to be thanked. From Beecroft Peninsula thanks to Mark Armstrong, Steve Moore and Crissy Locke. From Booderee National Park thanks to Matt Hudson, Martin Fortescue, Nick Dexter and Tony Carter. From the NSW Department of Environment and Conservation thank you to Damon Oliver and Bruce Gray. Thank you to the Wreck Bay Aboriginal Community and Jerrinja Aboriginal Community for their support of the project. All of these people contributed time and or support to the translocation and aided the project significantly.

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Translocation of the Eastern Bristlebird and factors associated with a successful program

Abstract

In the ongoing concern for the conservation of biodiversity around the globe, intensive, hands-on management of threatened species is becoming commonplace. The translocation of organisms to establish, re-establish or augment populations is one of the intensive strategies being used. This thesis explores the contemporary use of translocation in conservation, with a focus on the reintroduction of the Eastern Bristlebird (*Dasyornis brachypterus*) as a case study.

Translocation can be defined as the movement of living organisms from one area to free release in another. It is becoming increasingly common in the conservation of threatened species of a range of taxa around the world. Translocations have generally suffered from high failure rates, which have been mainly attributed to low habitat quality of the release site, a small number of individuals released, ignoring species-specific behaviours, poor management of the original threats to the species and stochastic environmental events. Aspects that have been associated with success include high habitat quality of the release site, reintroduction into part of the former range of the species, large number of individuals released and the use of a wild source population. Recent reviews have identified five key aspects of translocation projects that are required for a well-formed translocation program. These are the completion of a feasibility analysis, the use of criteria by which to assess success, the inclusion of experimental designs, financial accountability, and the effective communication of outcomes.

The bristlebird is an endangered Australian passerine. It is a small cover-dependent, semi-flightless bird that is restricted to a few isolated populations over a large geographic

range. It is threatened by habitat loss and fragmentation, inappropriate fire regimes and introduced predators. Reintroduction was identified by the National Recovery Team as a potential management strategy for the conservation of this bird. This reintroduction program was established with the following aims: Successfully translocate the bristlebird; investigate the post-release dispersal of reintroduced birds; monitor the impact of removing birds from a population.

To critically assess the effectiveness of the reintroduction program, seventeen criteria were established prior to commencement. The criteria were developed to be adaptable to a range of species or projects using a timescale that is measured in generation time rather than a set unit such as years. At the time of writing, the reintroduction was a resounding success. All criteria within the first four years were reached. Overall 13 of the 17 criteria have been achieved, including breeding being recorded in the reintroduced population and complete recovery of the source population following the removals.

The costs of the reintroduction program were analysed against other conservation options for the bristlebird. The analysis revealed that this reintroduction was much cheaper than many other translocation programs. It was also the cheapest option reviewed to potentially expand the area of occupancy of the bristlebird around Jervis Bay.

Two main study areas were established in the Jervis Bay region, NSW; one in the vicinity of the proposed source population at Booderee National Park and NSW Jervis Bay National Park and the other at the proposed release environment at Beecroft Peninsula. Over three years (2003 – 2005), fifty-one bristlebirds were caught using mist nets and 50 were transported to the release location and immediately released. All bristlebirds were banded, measured, had pin feathers collected for DNA analysis and radio transmitter attached before release.

In the release environment, reintroduced birds were radio-tracked for up to 34 days after release, to allow estimation of initial dispersal distances and mortality. Beyond this, transect surveys and targeted call playback were used to monitor the reintroduced

population once a year. In the reintroduced population, male bristlebirds dispersed further and moved more than females. Released bristlebirds did not disperse away from previously released conspecifics and settled quickly in nearby habitat. The translocated bristlebirds moved over much greater areas than bristlebirds in their native habitat; one bird dispersed over 4 km from the release location. During this monitoring, it was shown that bristlebirds can disperse a long way through continuous habitat, although they appear to prefer to settle in proximity to other bristlebirds. This has the potential to exacerbate the effects of habitat fragmentation as bristlebirds may not colonise nearby or tenuously connected habitat if there is a lack of conspecifics to enhance settlement.

Two monitoring sites were established in the source population, one where removals were to take place and another as a control to assess the impact of the removals on the population. The removal of 51 bristlebirds over three years from a single area in the source population had no detectable impact. Individuals that were removed appeared to have been replaced within six months of their removal. The origin of the replacement bristlebirds was unknown but the quick recovery was suggested to be a result of a surplus of non-calling or non-territorial birds within the population, perhaps combined with some juvenile dispersal. Such a surplus may be a mechanism for population persistence in an unpredictable environment or a result of insufficient suitable habitat for population expansion.

During the reintroduction, a wildfire burned a large proportion of bristlebird habitat in the location of the source population at Jervis Bay. The bristlebird has been described as fire-sensitive, with fire implicated in the decline of the species. The frequency of occurrence of bristlebirds was investigated in the second week after the fire in a range of sites varying in fire intensity. Bristlebirds were found in burned habitats but were more common in the less intensively burnt sites than in the more intensively burnt sites. Bristlebirds had been surveyed along transects in this area 2 months prior to this fire and were surveyed again 1, 9 and 13 months post-fire. Bristlebird numbers decreased in burnt areas after the fire and increased in unburnt areas. This pattern was evident for up to 9 months post-fire after which bristlebird numbers returned towards pre-fire levels in both

burnt and unburnt vegetation. This is in contrast to some previous research on bristlebirds and fire. It is suggested that bristlebirds avoided the fire by moving to unburnt areas. By the onset of the next breeding season, displaced bristlebirds were returning to pre-fire home ranges. It is speculated that the apparent lack of impact from this fire on bristlebirds was due to the close proximity of unburnt habitat and other refuges. The dispersal of juveniles and non-territory holding floaters from unburnt habitat combined with feral predator control probably contributed to the observed response, although these were not tested. It is suggested that the response of bristlebirds and presumably other birds to fire is strongly context-dependant and that fire management and bristlebird conservation may not be mutually exclusive.

Over fifty percent of bird species are sexually monomorphic and the bristlebird was previously considered to be part of this majority. I measured morphological characteristics on live and preserved bristlebirds, with sex determined genetically for live individuals using a common molecular technique, to test this suggestion. Males were significantly heavier, had larger heads, longer wings and longer tails than females. Univariate sexing criteria were developed based on the differences between males and females in two of these measures, weight and head-bill length and these measures were used to sex fifteen additional birds for which sex had been determined genetically. A discriminant function was also derived from the two characters. When the discriminant function was used in conjunction with the sexing criteria, 80 % of results agreed with molecular results, 7 % disagreed and 13 % were inconclusive. I speculate that this inaccuracy was due to juvenile males and the time of year of trapping, but the technique can be used to sex an individual in the hand with 80% accuracy, and can therefore provide a relatively quick and inexpensive method to investigate sex ratios in bristlebird populations and aid in the selection of individuals during further translocation projects.

Translocation can be a useful tool in conservation if planned and funded well and the outcomes published for the dissemination of information. This reintroduction has been a success, with bristlebirds surviving and breeding in the release environment. This has expanded the area of occupancy for the bristlebird and helped to reduce threats to the

species from stochastic events such as fire. Using experimental techniques, new aspects of bristlebird behaviour and population dynamics have been described, along with implications for the future management of the species.

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Chapter 1

General introduction

Translocation

Definition

Contemporary literature most often considers translocation as a conservation technique, a tool for wildlife managers in efforts to save threatened species (Danks 1994; Short and Turner 2000; Pickett 2002). The process of translocation is more than just this, involving domestic plant and animal releases, release of biological control agents and the movements of stowaways. All these types of translocations have occurred by human activities, either unwittingly or deliberately, for over 10,000 years (Heinsohn 2003). This thesis will just concentrate on translocation in relation to conservation biology.

The IUCN defines translocation as ‘the movement of living organisms from one area with free release in another’ (IUCN 1987), encompassing accidental and deliberate introductions, reintroductions and re-stocking or augmentation. Introduction is ‘the intentional or accidental dispersal by human agency of a living organism outside its historically known native range’. Reintroduction is ‘the intentional movement of an organism into a part of its native range from which it has disappeared or become extirpated in historic times as a result of human activities or natural catastrophe. Re-stocking or augmentation is ‘the movement of numbers of plants or animals of a species with the intention of building up the number of individuals of that species in an original habitat’. These definitions are now becoming accepted in both the scientific literature

(Griffith *et al.* 1989; Serena and Williams 1994; Wolf *et al.* 1996; Fischer and Lindenmayer 2000) and in government policy (NPWS 2001).

There are two sources of organisms for translocations: wild caught, where organisms are taken from wild populations; and captive-raised, where organisms are born in captivity, or taken from wild populations, and raised in captivity in purpose-built compounds. Both of these sources of individuals can be considered a source population. There are two broadly encompassing release strategies used in translocations, which pertain more specifically to animals. Soft release is where animals are held in purpose-built enclosures in the host environment for a period of acclimation with various, staged release strategies being employed (Kleiman 1989). Hard release is where animals are released straight into the host environment with no acclimation period, although often with supplementary food, water or nesting resources (Kleiman 1989). The host environment is the area previously secured for the establishment of the translocated or release population.

History

Translocations have been carried out for over 10 000 years. The first direct evidence of translocations comes from the Roman Empire, with their expansion of the range of rabbits and fallow deer (Woodford and Rossiter 1994), presumably for food and clothing resources. Today, translocations are carried out all over the world, predominantly for the management of wild populations. Translocation, as a conservation technique, has been used for just over 100 years. New Zealand has been a world leader, carrying out over 400 translocation projects (Armstrong and McLean 1995) since at least the 1880s (Saunders 1994). A broad spectrum of animals have been translocated. Examples include: mammals – the Golden Lion Tamarin (*Leontopithecus r. rosalia*) in Brazil (Kleiman *et al.* 1991), the Arabian Oryx (*Oryx leucoryx*) in Oman (Stanley Price 1989) and the Burrowing Bettong (*Bettongia lesueur*) in Australia (Short and Turner 2000); birds – the Californian Condor (*Gymnogyps californianus*) (Toone and Wallace 1994) and the New Zealand Stitchbird (*Notiomystis cincta*) (Castro *et al.* 1994); reptiles – the Tuatara (*Sphenadan*

punctatus) in New Zealand (Armstrong and McLean 1995) and the Shingle-back (*Tiliqua rugosus*) in Australia (Copley 1994); amphibians – the Natterjack Toad (*Bufo calamita*) in the United Kingdom (Seigel and Dodd 2002); and invertebrates – the Mahoenui giant Weta (*Deinacrid spp.*) in New Zealand (Sherley 1995).

Contemporary needs and failures

Reviews

Translocations around the world have been characterised by a high failure rate (Fischer and Lindenmayer 2000). In an attempt to understand the reasons behind these failures and the attributes of the many projects that have been successful, there have been a number of reviews on translocation (Griffith *et al.* 1989; Short *et al.* 1992; Copley 1994; Wolf *et al.* 1996; Wolf *et al.* 1998; Fischer and Lindenmayer 2000). Correlates with success included high quality of habitat at the release site, reintroduction into part of the former range of the species, large number of individuals released and the use of a wild source population. The main correlates with failure were low quality of habitat, small number of individuals released, ignoring species-specific behaviours, poor management of the original threats to the species and stochastic environmental events. The reasons associated with success and failure are useful to understand when planning translocations for the conservation of threatened fauna. These reviews highlight five aspects of translocations that have been lacking in many projects: (1) The completion of a feasibility analysis prior to commencement, (2) the use of criteria by which to critically assess outcomes, (3) the use of experimental designs that can assess hypotheses scientifically, (4) lack of financial accountability and (5) the publication of outcomes.

Feasibility Analysis

The completion of a feasibility analysis is essential prior to the commencement of any translocation to evaluate biological, political, social and resource issues relating to the species and the project (Stanley Price 1989; Kleiman *et al.* 1994). Kleiman *et al.* (1994) developed a checklist of 13 yes/no questions to test the appropriateness of a proposed

translocation. The questions relate to the status of the species, environmental conditions, bio-political conditions and biological and other resources. This checklist provides a quick gauge on the current situation regarding a species' potential for translocation. The IUCN *Position Statement on Translocation of Living Organisms* (IUCN 1987) advises the completion of a feasibility analysis prior to the commencement of any translocation project.

Within Australia and NSW; the *Policy for Translocation of Vertebrate Animals in Australia* (Australian National Conservation Agency 1994), the *Policy statement No.29 Translocation of Threatened Flora and Fauna* (CALM 1995) and the *Policy for the Translocation of Threatened Fauna in NSW* (NPWS 2001) stipulate the completion of a Translocation Proposal to analyse the feasibility and the need to translocate, which is reviewed before the commencement of the project. Checklists and proposals provide a simple means to understand the issues surrounding a translocation and possibly identify areas of theoretical and experimental interest that could be targeted for research. The science of translocations is a relatively new area of theoretical interest and our understanding falls short of providing a theoretical framework (May 1991), although progress is being made (Wolf *et al.* 1998; Fischer and Lindenmayer 2000).

Criteria for Success

The establishment of a clear set of criteria for the evaluation of success is critical before the commencement of a translocation. The success of any translocation can only be effectively assessed against a clear set of goals and objectives (Kleiman *et al.* 1994; Fischer and Lindenmayer 2000). These criteria need to be developed according to a timescale (Fischer and Lindenmayer 2000), because the outcome of a translocation after one year may be very different to its outcome after 20 years. Criteria may incorporate the survival of individuals, evidence of breeding or population growth and replacement in the source population. If a universal set of broad criteria, adaptable to all vertebrate translocation projects could be adopted across translocations, it would allow more meaningful comparisons between projects. This has been attempted in Chapter 2.

Prior to the commencement of a translocation project some initial preparation is required. This involves finding a potential source population and host environment and preparing a feasibility analysis and translocation proposal outlining the project. This initial preparation is an involved process but most translocations, particularly of threatened species, would fail without it. This initial preparation should be the first criterion. If initial preparation is attained, then failure in the translocation will be due to a shortcoming in the translocation itself. The primary objectives of a translocation are generally focused on the development of a self-sustaining, viable population. An equally important objective is the contribution of the project to conservation science and to the local human community (Serena and Williams 1994). All translocations are opportunities for experiments, so they should be planned and monitored accordingly, to ensure that our knowledge is advanced whether the particular program is a success or failure (May 1991; Armstrong *et al.* 1994; Soderquist 1994; Fischer and Lindenmayer 2000).

Studies within translocation projects

Experimental components should be incorporated into translocation projects (Armstrong *et al.* 1994; Soderquist 1994; Fischer and Lindenmayer 2000), allowing various aspects of translocations to be rigorously tested (Armstrong and McLean 1995; Sarrazin and Barbault 1996). If projects are well designed, valuable data may be collected about techniques or the species even in the event of failure (Soderquist 1994). If opportunities for experiments are considered prior to a translocation, then the project can be designed to test specific hypotheses developed for the taxa and system involved. The scope of such projects may often be limited by a paramount need to conserve the species of interest. To explore important questions related to translocation, it may be possible to use species that are declining, and hence of conservation interest, though not yet endangered (Serena and Williams 1994), or common species.

Recently, there have been a number of translocations that have included experimental components or that have been conducted specifically to investigate various aspects of translocation. Clarke and Schedvin (1997) translocated the common Noisy Miner

(*Manorina melanocephala*), monitoring the integration and survival of translocated individuals released into existing populations. They found that translocated individuals were not assimilated into the resident population and ranged widely from the release point. They concluded that augmentation of populations of communal species such as this may not work. Armstrong (1995) and Armstrong and Craig (1995) investigated the influence of familiarity of founding groups in translocations of two threatened New Zealand birds. For Saddlebacks (*Philesturnus carunculatus rufusater*) and New Zealand Robins (*Petroica australis longipes*), they found that familiarity had no influence on the success of the translocations. Relationships seemed to be context-dependent because most pairs broke bonds after translocation.

Five years after a translocation of New Zealand Robins which involved an initial release and then an augmentation after 14 months, Armstrong and Ewen (2001) tested the value of the augmentation using population viability analysis. This analysis revealed that the low population growth in the first year, which was the basis for the decision to augment the reintroduced population, was an initial effect of the translocation rather than a permanent attribute. They concluded that resources may have been better spent on monitoring the translocated population for several years rather than carrying out the follow-up translocation so soon. Monitoring would also provide information that may be applied to other management issues and other species. Armstrong and Ewen (2002) also modelled the New Zealand Robin population using data collected during the six years after the initial reintroduction and found evidence for a decline in juvenile survival as population size increased, that the distribution of recruits was affected by the number of surviving residents and that female fecundity was lower in the first year after translocation than in the succeeding six years. They advised caution in using data from the first year after translocation for predicting population trends. Armstrong *et al.* (2005) investigated population growth in Saddlebacks for six years after their reintroduction and found evidence for a decline in juvenile survival and overall reproductive output as the population increased. These results are now being used to plan management strategies for future reintroductions of the species. Bright and Morris (1994) investigated translocation success between wild-caught and captive-raised founding individuals, and between hard

and soft release strategies in Dormice (*Muscardinus avellanarius*). They found less dispersal in captive-raised founders than in wild founders and that hard-released individuals dispersed more widely and travelled for longer than soft-released individuals.

These translocation projects have studied important aspects in the translocation of fauna, and have begun the accumulation of experimental studies that may be used to plan future translocation projects. The studies involving New Zealand Robins and Saddlebacks highlight the point that monitoring and data analysis should not stop immediately after a translocation. These new populations often provide opportunities to investigate broader ecological theory, such as density-dependence and dispersal, and too little monitoring may lead to inaccurate predictions about these populations.

One of the biggest assumptions in conservation biology is using current habitat as an indicator of what represents optimal habitat for an endangered species. This assumption is usually not tested. In some instances, the present habitat has been found not to be optimal, or not the only habitat exploitable by a particular species (Craig 1994). Craig (1994) highlighted two examples: (i) The Takahe (*Porphyrio mantelli*) in New Zealand was thought to prefer alpine tussocks for feeding but, when translocated into an area lacking alpine tussocks, dominated by grass and forest habitats, the Takahe bred at an earlier age and mortality rates were lower. (ii) The Saddleback in New Zealand was thought to prefer tall old forest but, when translocated to an island mostly lacking mature forest, the Saddlebacks were found using any scrub habitat available including regenerating forest, and they have proceeded to breed at an earlier age and produce larger clutches.

These outcomes demonstrate that currently used habitat may not always be optimal habitat and, for some species, current habitat may be just a refuge from previous or current threats. However, when dealing with endangered species, the use of current habitat as a guide for management may be the only reasonable assumption if no other information is available.

Reviews of translocations have found that success is associated with the release of animals into their historic range (Griffith *et al.* 1989; Wolf *et al.* 1996; Wolf *et al.* 1998; Fischer and Lindenmayer 2000) and translocation guidelines often stipulate release into former ranges, if lacking a strong case for release outside the former range (IUCN 1987). However, many researchers have suggested experimental translocation into different habitats or outside of former ranges (May 1991; Craig 1994; Serena and Williams 1994). Properly designed, such programs could be used to test the assumption that current habitat is optimal and to explore the option of conserving endangered species in existing reserves that are outside of former ranges, rather than incurring large costs of restoring degraded habitat outside of reserves but within former ranges. Norton (1994) suggested that this sort of rationale should be extended to all areas of wildlife management, especially where scientific knowledge is lacking.

Cost

The costs of translocations are high (Serena and Williams 1994) and there is a lack of financial accountability in most projects (Fischer and Lindenmayer 2000). Of 180 published descriptions of translocations, Fischer and Lindenmayer (2000) found that only six reported the costs associated with the projects. Two examples are the reintroduction of Golden Lion Tamarins (*Leontopithecus r. rosalia*) in Brazil, which cost in excess of US \$1 000 000 per year, or approximately US \$22 000 per surviving Tamarin (Kleiman *et al.* 1991); and the Californian Condor (*Gymnogyps californianus*) reintroduction program, which was estimated to be US \$20 000 000 over 14 years, equating to approximately US \$300 000 per bird (Cohn 1993). The evaluation of the costs of projects is valuable information for land managers considering the use of translocation for the management of particular species.

Publishing/reporting

The publication of the findings of translocation projects is important (Copley 1994; Fischer and Lindenmayer 2000). Copley's (1994) review of South Australian translocations revealed that information on the source of individuals and data on the monitoring of released individuals was lacking in many projects and organisation of the

projects was poor. Fischer and Lindenmayer (2000) reviewed published accounts of translocations, reviewing 180 case studies, whereas in earlier reviews based on questionnaires sent to researchers, Griffith *et al.* (1989) assessed 421 translocation programs. This provides evidence to suggest there is a lack of published accounts of translocations compared to the number undertaken, restricting the development of a general set of successful protocols. This lack of published or accessible information on translocations is a common failure (Fischer and Lindenmayer 2000) and can make critical evaluation difficult and limits the contribution to conservation that these projects can offer. A possible reason for this lack of publication of translocation projects is in the priorities of those conducting them. The output of land managers, who are most often carrying out the translocations, is usually measured by other practical outcomes rather than scientific publication. This suggests that there is a need for a formal integrated reporting mechanism associated with translocation projects to promote the dissemination of information.

Policy documents (IUCN 1987; Australian National Conservation Agency 1994; NPWS 2001) all stipulate the publication and availability of information as a priority in translocation projects. Reporting and publication of translocation goals, methods, monitoring and outcomes should be enforced as part of an approval process for translocation projects. This could be achieved by the development of a register of translocation projects that could be managed during the licensing process for wildlife research. As part of the register, an annual newsletter could be published which reports on current translocation programs. Alternatively, the Reintroduction Specialist Group (RSG) within the IUCN already publishes a number of newsletters on current translocation programs, such as the RSG Oceania Newsletter. An Australian registry of translocation programs could be linked with this newsletter for a common outlet to report information on current programs.

Populations and individuals

Source population

The impact on a source population of the removal of individuals depends on the characteristics of the species involved. For example, the behaviour of individuals from a solitary species would be expected to differ markedly from a communal species, following the removal of individuals. Several outcomes of removal experiments on territorial species are common across a number of studies. Territoriality can be loosely defined as ‘any form of spacing behaviour that involves site-specific dominance, producing a dispersion pattern that is more regular than random, and which gives priority access to resources’ (Newton 1992). It would follow that territoriality should be more pronounced where resources are patchy in quality. Kluyver and Tinbergen (1953) described that outcome: territoriality in Titmice (*Parus spp.*) limited numbers more in better quality habitat and that any adjacent poorer habitat housed ‘overspill’ from the good quality habitat. Numbers were more variable year to year in the poor, than in the good, quality habitat. The idea that territorial behaviour can limit the numbers of individuals present in an area has been proposed in a number of other studies of removal of territorial passerines (Wesolowski 1981; Sherry and Holmes 1989; Monkkonen 1990). The extent of the influence territoriality can have on limiting the numbers of territory-holding individuals is illustrated in a removal experiment by Knapton and Krebs (1974) on Song Sparrows (*Melospiza melodia*): when all territory-holders were removed at once, the replacement territories were smaller, due to more territories being established in the same area at the same time. However, when territory-holders were removed one at a time, the replacement territories were the same size as others before the removal. In an experiment on Willow Warblers (*Phylloscopus trochilus*), the removal of individuals caused an influx of new arrivals along with the expansion of territories by some territory-holders that were not removed (Arvidsson and Klaesson 1984). Similar results were found in removal experiments of Crested Tit (*Parus cristatus*) and Willow Tit (*Parus montanus*), where there were no observed differences in density between control sites without removal and impact sites one year after removal (Cederholm and Ekman 1976). The influx of individuals were suggested to be coming from a non-territorial sector in the

population, more often described as ‘floaters’ (Newton and Marquiss 1991), which live in the same areas as the territory holders (Newton 1992), but have much larger home ranges than territory holders (Pedersen 1988).

Of 23 translocation projects which used wild-caught individuals as founders, published since 1983 (Table 1.1), 19 did not monitor (or did not report monitoring) the source population. In translocations of both the Numbat (*Myrmecobius fasciatus*) and the Noisy Scrub-bird (*Atrichornis clamosus*), both source populations have continued to increase in numbers despite the removals. At the largest wild Numbat population, numbers have increased from <400 in 1985 to >800 individuals in 1992 despite the removal of 10 to 30 Numbats per year (Friend and Thomas 1994). The rate of increase in the most secure source population of Noisy Scrub-birds has remained unchanged over the last nine years despite the removal of 109 birds for six translocation projects over that time (Danks 1994). However, since 1995 this rate of increase has not continued, although this cannot be accounted for by birds removed for translocation (A. Burbidge pers. comm. 2007). In contrast, 30 Southern Emu-wrens (*Stipiturus malachurus intermedius*) have been recently translocated in South Australia (Pickett 2001). After six months, 70% of home ranges left vacant from removal of individuals for translocations had still not been occupied. However, it is unclear whether these vacancies were filled during the ensuing breeding season. After the removal of all known pairs of Saddlebacks from a single breeding area for translocation, unsurprisingly, no breeding was recorded in that area in the subsequent breeding season (Armstrong and Craig 1995).

Table 1.1: Species (23) translocated during 1983 – 2003 and whether or not monitoring of the source population was reported.

Founding individuals

In 34 recent (last 25 years) conservation-based translocation projects from around the world (Table 1.2), the majority discuss post-release movements of animals, although very few investigate the process of colonisation by these translocated individuals analytically. Three of the 34 projects neglected to mention anything about the released individuals, 31 projects discuss post-release behaviour of translocated individuals but only 6 used an experimental component to their monitoring. Only two of the projects actually compared

the behaviour of translocated individuals to the behaviour of animals in their natal areas, in an attempt to include a meaningful interpretation of movements.

Table 1.2: 34 conservation-based translocation projects and their investigation of post-release behaviour of released individuals. An experimental design meant using a rigorous experimental approach to test a hypothesis. It was also investigated whether the behaviour of translocated individuals was compared to the behaviour of individuals in native surroundings.

The arrival of founders to a release site in a reintroduction project can be considered analogous to some other colonisation events. These may be island colonisation events or colonisation of an area following disturbance, such as fire. Many of these events have been studied in more detail than colonisation events relating to translocations (Crawley 1986; Baker and Jenkins 1987; Baker *et al.* 1990; Grant and Grant 1995; Woinarski and Recher 1997; Clegg *et al.* 2002; Grant 2002; Burbidge 2003). Some of the key findings from this research that relate to reintroductions include the type of species or individuals that do well in colonising new areas, and genetic changes following colonisation events. The poor colonisers are generally species that naturally have a low rate of increase (Crawley 1986), because population growth is slow, along with the replacement of any individuals who die. However, this does not preclude them from successful colonisation if conditions are favourable. In areas where Marsh Harriers (*Circus aeruginosus*) were removed in the Netherlands, young animals began the colonisation process (Altenburg *et al.* 1987). Newton and Marquiss (1991) also suggested that bird translocations may have more success with younger birds. Conversely, as it is the founding individuals that bring the behavioural traits which will set up the behaviours for a new population (Baker and Jenkins 1987), bird translocations may have more success with older and more

experienced birds. Perhaps a mixture of age classes would facilitate successful translocation in a range of species.

Genetics

Genetic factors should be considered in all translocation proposals. With supplementation, there is a possibility of mixing different genetic stocks. With reintroduction or introduction founder effects and bottlenecks will have inevitable genetic consequences. Introduced populations of Common Mynas (*Acridotheres tristis*) were found to have a lower average heterozygosity, lower number of alleles per loci and a lower percent of polymorphic loci than native populations (Baker and Jenkins 1987), consistent with theoretical predictions from inbreeding caused by bottlenecks or founder events (Lacy 1992). Interestingly, in Silvereyes (*Zosterops lateralis*) there was only a small loss of alleles at founder events, although this resulted in a gradual decline in allelic diversity over several sequential founder events (Grant 2002). When considering genetic implications in translocations of threatened species, it is worth noting that the effects of inbreeding are much reduced in historically small and slowly declining species (Lande 1995). Colonisation events of Large Ground Finches (*Geospiza magnirostris*) in the Galapagos Islands coming from a small initial population showed only slight inbreeding depression (Grant and Grant 1995). In experiments on Mosquitofish (*Gambusia holbrooki*), Leberg (1993) concluded that mixed genetic stocks did not have better population growth than non-mixed stocks and that the mixing of genetic stocks for founder groups in translocations may not be necessary to maintain genetic diversity. These results may depend on the genetic diversity contained in the non-mixed founder groups. However, the non-mixed stocks had varying levels of heterozygosity depending on the locations they were collected from and no differences in population growth was evident between these non-mixed founding populations either.

Australian avian translocations

Within Australia, there have been numerous attempts to translocate animals. Of an estimated 50 species since 1907: 25 have been birds, 22 mammals and 4 reptiles (Menkhorst *et al.* 1990; Short *et al.* 1992; Brown *et al.* 1994; Burgman *et al.* 1994; Copley 1994; Danks 1994; Gibson *et al.* 1994; Short *et al.* 1994; Priddel and Wheeler 1999; Garnett and Crowley 2000; Smales *et al.* 2000; Clarke *et al.* 2002; Pickett 2002; Richards and Short 2003). Of the 25 translocations of bird species in Australia from 1911 to 2001, nine projects have been purely for conservation purposes and all of these have occurred since 1980 (Table 1.3). This increase in popularity or necessity of translocations within bird conservation in Australia suggests the need for the reporting of projects, the use of success criteria and experimental tests of the impact of removals and monitoring of population changes following translocation to help inform future translocations. Within the 9 Australian avian conservation translocations, only 3 stated success criteria prior to commencement, only 2 reported an experimental technique testing a hypothesis and, encouragingly, 8 published specific accounts of their projects (Table 1.3). This parallels the conclusions for fauna in general (Fischer and Lindenmayer 2000).

Translocations are significant undertakings and consequently they should be investigated, planned, monitored and reported accordingly. The feasibility of a translocation should be investigated before commencement. At a minimum during the translocation, numbers of individuals in both the source population and the host environment should be monitored. It is critical to monitor the source population as the translocation should in no way jeopardise its wellbeing. The reporting of various aspects of translocation programs will facilitate an increased ability to conduct successful translocations. As translocation becomes a common conservation technique in Australia, it is important that clear guidelines are established to help land managers and researchers in the preparation, execution, evaluation and reporting of translocation projects.

Table 1.3: Overview of translocations involving nine Australian bird species since 1980. Was success assessed against criteria developed prior to the translocation, was an experimental design used to test a hypothesis and where were the outcomes published.

The Eastern Bristlebird

Description

The Eastern Bristlebird (*Dasyornis brachypterus*) is a small (22 cm) brown, insectivorous Australian passerine. It is cryptic, mainly ground-dwelling and, with its short wings, a poor flyer. It is considered to be sexually monomorphic (Simpson and Day 1996; DEC 2004), but Chaffer (1954) suggested that, while both sexes look superficially similar, the females may be smaller than the males. Baker (2001) showed that the species lives in home ranges averaging ≥ 10 ha over 2 – 6 weeks that can overlap by up to 80 % and that bristlebirds can be heard calling from within these home ranges. It is one of three species of bristlebird in Australia, all of which are rare or threatened.

The bristlebird occurs on the east coast of Australia in a range of vegetation types including rainforest, heathland, sedgeland, woodland, tall forest and dune vegetation. The vegetation requirements for the species were investigated by Baker (2000) and were found to incorporate dense low vegetation, regardless of plant species composition.

Occurrence

Baker (1997) estimated that there were fewer than 2000 individuals distributed among 10 - 14 disjunct populations. Barren Grounds Nature Reserve and the adjacent Budderoo National Park on the Illawarra Escarpment west of Kiama, NSW and Bherwerre Peninsula at Jervis Bay, NSW are the largest populations of the bristlebird, each exceeding 600 individuals (Baker 1997), an estimate based mostly on calls. Nadgee Nature Reserve on the Victorian border is suggested to be the next largest with an estimated 120 individuals (Baker 1997) and 10 individuals were recently found at Red Rocks Nature Reserve near Kangaroo Valley, NSW (Bain and McPhee 2005). Howe Flat is the only known location still containing bristlebirds in Victoria, although < 10 birds are thought to remain (Clarke and Bramwell 1998, Baker 1997). In the north of its range, the

bristlebird is very restricted. In northern NSW and southern Queensland, there are estimated to be fewer than 50 individuals spread across 12 populations (Stewart 2004). Besides these populations, there are historic or more recent unconfirmed reports from areas in central eastern New South Wales: In Ku-ring-gai Chase National Park north of Sydney, NSW, the species was apparently common before 1904 (North 1904) but there has been only a single record in the last three decades (Saunders 1986). On Beecroft Peninsula at Jervis Bay, NSW, there are three recent but unconfirmed records between 1984 and 1995, but more recent, systematic searches have failed to detect any individuals (Baker 1997; Bain and McPhee 2005).

Status

The Eastern Bristlebird is listed as threatened in all jurisdictions of its range: nationally endangered under the *Environment Protection and Biodiversity Conservation Act 1999*, endangered in New South Wales under the *Threatened Species Conservation Act 1995*, endangered in Queensland under the *Queensland Nature Conservation Act 1992* and threatened in Victoria under the *Victorian Flora and Fauna Guarantee Act 1988*. It is one of three species of *Dasyornis* in Australia, all of which are rare or threatened.

Threats and threat abatement

Incremental habitat loss and fragmentation of remaining habitat is considered to be the main process causing declines in bristlebird populations (Smith 1977; DEC 2004). Habitat loss may be temporary, which happens after fire, or more extensively permanent, with the conversion of habitat to urban or agricultural land. Around Jervis Bay natural dispersal and population expansion out of reserved land is limited by land tenure and encroaching urban development. The natural colonisation of vacant but potential bristlebird habitat (Bain 2001; Bain and McPhee 2005) around Jervis Bay is considered doubtful.

The bristlebird has been described as fire-sensitive, based primarily on the work of Baker (1997; 2000; 2003). At least nine populations of bristlebird have gone extinct in recent years and many of these local extinctions have been attributed to fire (Holmes 1989; Woinarski and Recher 1997; Clarke and Bramwell 1998). Fire temporarily removes dense understorey vegetation, which is the bristlebird's preferred habitat. Being semi-flightless and cover-dependent, the bristlebird is not expected to be able to colonise new areas readily or to recolonise areas quickly following disturbances (Smith 1977).

Fire management for bristlebirds is complex. Currently, two major prescriptions exist, one for the northern populations and another for populations in the south of NSW. For the northern populations, it has been argued that fire would be required every seven to ten years to maintain suitable habitat (DEC 2004). However, for the southern populations, it has been recommended that fire be excluded from bristlebird habitat, unless site-specific data suggest otherwise (Baker 1997; 2000). In general, it has been recommended that large-scale fire should be suppressed in bristlebird habitat and, during fire-fighting, converging fire fronts should be avoided if possible (DEC 2004).

Bristlebirds nest on or close to the ground (Higgins and Peter 2002), leaving them susceptible to introduced predators such as the European Fox (*Vulpes vulpes*), Cat (*Felis catus*) and Black Rat (*Rattus rattus*). Baker and Clarke (1999) found compelling evidence that a fox was responsible for predation on an adult bristlebird. However, the extent of the impact of predation is unknown, but it has been suggested that the impact may increase following fire (DEC 2004). Currently, the control of cats and foxes is recommended for bristlebirds. In particular, following large-scale wildfires in bristlebird habitat, fox control is recommended to begin immediately to protect surviving birds near refuges (DEC 2004) because fox predation has been suggested to increase following fire (Nick Dexter, pers. comm.). Cat predation may also increase after fire, although direct evidence is lacking. Currently there are no broad scale control measures for cats.

In the two remaining big bristlebird populations, fire represents the most likely and least manageable threat. Both the Jervis Bay and Barren Grounds populations are mainly on reserved lands where urban encroachment is not a threat and feral predators are managed. Dispersal of these populations out of reserves is unlikely due to land-use and tenure and there is a need to help reduce the threat of fire to the species, therefore, there was a need for another large and secure population in another reserve.

As part of the conservation of the bristlebird in southern NSW, the translocation of bristlebirds was proposed with the aim of establishing another population on reserved land. Captive breeding has also begun in the management of the northern populations of the bristlebird (DEC 2004), where bristlebirds are being raised in captivity for future release into the wild.

The Translocation Proposal for the Eastern Bristlebird (Whelan and MacKay 2002) examined the feasibility of translocating bristlebirds in the Jervis Bay region and found no impediment to proceeding with a project. This is demonstrated in the summary of that assessment (Table 1.4) against the 13 issues listed by Kleiman *et al* (1994). Following this positive outcome of the Translocation Proposal, the translocation project at Jervis Bay was initiated.

Table 1.4: Feasibility analysis for the reintroduction of the Eastern Bristlebird in the Jervis Bay region. Checklist from Kleiman *et al.* (1994).

Aims

The principal aim of this thesis is to assess the potential of translocation for the conservation of small Australian passerines using the bristlebird as a case study. The key studies presented here are (i) the development of translocation methods and efficacy of the translocation, which aims to assess the success of the reintroduction and examine the costs of the technique and its value to conservation. (ii) Investigation into the potential for sexual dimorphism in the bristlebird and any potential for a field-based sexing technique. (iii) Investigating the post-release dispersal of reintroduced bristlebirds to test the hypotheses that there will be a difference in dispersal behaviour between males and females and between bristlebirds released with and without conspecifics present and differences to bristlebirds in native habitat. (iv) Monitoring of the source bristlebird population during the reintroduction to investigate the impact of the sustained removals over three years. This was used to test the hypothesis that removals will cause a drop in

density and cause a change in the distribution of bristlebirds across the habitat. (v)
Studying the post-fire recovery of bristlebirds to examine the effects of fire intensity on bristlebird occurrence immediately after fire and monitor recovery over the first 13 months post-fire. Bristlebird recovery after fire is currently thought to take up 10 to 15 years (Baker 1997). This investigation was used to test the hypothesis that this fire would cause a significant and sustained decline in the population. During the second year of the project a large wildfire burnt through approximately half the area containing the source population. This fire altered the project design and highlighted the importance of establishing a new population. The fire also provided the opportunity to investigate the impacts of fire on bristlebirds.

Structure of thesis

This thesis has been compiled as a series of stand alone data chapters (chapters 2, 3, 4, 5 and 6) which have been written as manuscripts. Abstracts have been removed and formatting has been kept consistent throughout the thesis. These manuscripts are bounded by a general introduction to the topics in Chapter 1 and a general discussion in Chapter 7. Consequently there will be some repetition between the first chapter and the introductory sections to each of the data chapters. A small amount of repetition may also be found between the methods sections of each of the data chapters. All species are reintroduced in each chapter. A page reminding the reader of repetition precedes each data chapter and indicates where each manuscript is to be submitted.

The efficacy of translocations and developing criteria for success: a case study with the Eastern Bristlebird

A manuscript for submission to
Journal of Wildlife Management *The Wildlife Society*

This chapter has been written as a manuscript and as such it will contain some repetition in the introductory section from Chapter 1 and some overlapping in the methods section with other data chapters. All species are reintroduced.

Chapter 2

The efficacy of translocations and developing criteria for success: a case study with the Eastern Bristlebird

Introduction

While people have been moving organisms around the globe for thousands of years (Heinsohn 2003), in recent decades, translocation has been used increasingly as a conservation strategy for threatened species management (Griffith *et al.* 1989; Armstrong and McLean 1995; Fischer and Lindenmayer 2000). Translocation is the movement of living organisms from one area with free release in another (IUCN 1987). It incorporates introductions, reintroductions and augmentation (IUCN 1987; Griffith *et al.* 1989). Individuals are sourced for translocations from captive or wild populations and are released using one of two methods. Soft release is when organisms are held in purpose built enclosures in the release environment for a period of acclimation and monitoring before various staged release strategies (Kleiman 1989). Hard release is when organisms are released directly with no acclimation period, although sometimes with supplementary resources such as food, water or nesting material (Kleiman 1989).

Determining success in a translocation project is not straight forward and is rarely done effectively. The use of goals and criteria stated prior to the start of a translocation project can allow critical evaluation of its success. However, there needs to be widely applied and generally accepted criteria to assess translocations (Fischer and Lindenmayer 2000). If a universal set of broad and adaptable criteria can be used for assessing translocations,

it would allow more meaningful comparisons among projects and contribute to the development of a more scientific approach. For example, an important criterion is the use of a broadly applicable timescale (Dodd and Seigel 1991; Fischer and Lindenmayer 2000), as the outcome of a translocation project after one year may be very different to its outcome after 20 years. The use of a universal factor associated with reproduction or population growth may be a more meaningful measurement of time than the use of a fixed number of years.

The translocation of species can be expensive (Kleiman 1989), although the costs are rarely analysed or reported. Of 180 published translocations, Fischer and Lindenmayer (2000) found that only six reported costs associated with the project. This paucity of information on costs associated with translocations can make assessing the feasibility and the planning of translocations difficult for land managers.

In the last 25 years in Australia, there have been translocation projects for 9 threatened bird species and the technique is becoming increasingly popular in conservation worldwide (Miller and Mullette 1985; Smales *et al.* 1990; Brown *et al.* 1994; Danks 1994; Priddel and Wheeler 1999; Burbidge 2001; Priddel and Carlile 2001; Clarke *et al.* 2002; Pickett 2002). The projects aimed to increase the number of individuals, the number of populations and to reduce threats to the survival of these species. The popularity and importance of translocation as a conservation technique means there is a responsibility to undertake properly planned, comprehensive programs that can effectively assess success and report on aspects valuable to future translocation projects.

The Eastern Bristlebird (*Dasyornis brachypterus*) is nationally endangered under Australia's Commonwealth *Environment Protection and Biodiversity Conservation Act 1999* and in all jurisdictions in its range. Current threats to the species include habitat fragmentation, introduced predators and inappropriate fire regimes (Baker 1997; DEC 2004). Bristlebirds are currently restricted to a small number of populations on the east coast between southern Queensland and northern Victoria (Baker 1997). Only two of these populations were estimated to exceed 600 individuals (Baker 1997) and

translocation was identified in the draft recovery plan (NPWS 2000a) as a method to reduce the impact of potential threats to the species, particularly catastrophic wildfire.

The aims of this chapter were: to use the bristlebird as a case translocation to show core elements in methodology and then to use this to develop criteria to assess the success of the reintroduction, and to compare the cost of reintroduction to the costs of other potential conservation strategies that may be used for the management of this endangered bristlebird.

Methods 1

Assessment of site selection

Source population

There are two locations, both in south-east NSW, recognised as the centres of the largest, and most secure bristlebird populations: Barren Grounds Nature Reserve (150°43', 34°42') and Bherwerre Peninsula (150°45', 35°04') at Jervis Bay. The Jervis Bay population is found predominantly within Booderee National Park, Jervis Bay Territory, and NSW Jervis Bay National Park. It was chosen as the source population for a number of reasons: it has a good network of management trails, which were important for access to bristlebird habitat and from which to catch bristlebirds; it was close to a potential release location, directly only 12 km away; there was support from the local community and land management agencies.

Host environment

The Draft Recovery Plan for the Eastern Bristlebird (NPWS 2000a) identified three potential host environments in south-east NSW for possible reintroduction: a Sydney Catchment Authority Special Area west of Wollongong (150°53', 34°26'), Ku-ring-gai Chase National Park (151°15', 33°36') north of Sydney and Beecroft Peninsula (150°48',

35°03') at Jervis Bay. Beecroft Peninsula was chosen because it is part of the former range of the species, is in close proximity (12 km) to the source population at Bherwerre Peninsula, the climate, altitude and vegetation communities are similar at both sites and the project had the support of the local community and land managers. It is assumed that bristlebirds once occupied Beecroft Peninsula, although Hindwood visited the site for a week in 1932 but failed to detect the species despite recording other heathland species such as the Southern Emu-wren (*Stipiturus malachurus*) and Ground Parrot (*Pezoporus wallicus*) (Hindwood 1933). They noted that the vegetation was mostly knee-high and sometimes quite open and low. Bristlebirds may have been confined to areas of denser vegetation because more recently (between 1984 and 1995) there were three reported observations of bristlebirds (Baker 1997). There are also historical notes on a series of fires between December 1942 and December 1946, each of which burnt the whole of the Peninsula (Dunphy 1957). If bristlebirds were present on Beecroft Peninsula in the two decades prior to the present study they were in very low numbers.

The diet of the bristlebird consists of invertebrates and a little vegetative matter, in particular seeds (Higgins and Peter 2002). A comparison of invertebrates at Barren Grounds Nature Reserve, Bherwerre Peninsula and Beecroft Peninsula suggested that invertebrate availability would be adequate to support reintroduced bristlebirds on Beecroft Peninsula (Gibson 1999; Gibson and Baker 2004).

Before the commencement of the reintroduction, it was important to understand and minimise the threats to the species at Beecroft Peninsula. One sixth of the land on Beecroft Peninsula is used primarily for Defence activities, and here, environmental goals are secondary to Defence interests (Bushfire and Environmental Services 2001). The remaining part of Beecroft Peninsula is managed for biodiversity conservation and some recreational activities, although due to the Defence activities public access to Beecroft Peninsula is restricted. Environmental management of the whole of Beecroft Peninsula is carried out by rangers from the Shoalhaven Defence Environment Team, employed by the Australian Department of Defence. Beecroft Peninsula encompasses 4,027 ha and the majority of the peninsula is of high conservation value, with 27 plant communities and

179 vertebrate species recorded, including 17 threatened plant and vertebrate species (Bushfire and Environmental Services 2001). Current threats to the bristlebird include catastrophic fire, inappropriate fire regimes and introduced predators. Fire management at Beecroft Peninsula already includes strategies for bristlebirds (Bushfire and Environmental Services 2001) such as maintaining areas of dense vegetation with appropriate fire regimes. With the commencement of the reintroduction, fire trails have been upgraded and a core area of habitat has been outlined to be managed primarily for the conservation of the reintroduced bristlebirds. There is an ongoing program for feral predator control on Beecroft Peninsula, particularly foxes.

Vegetation study

The structural attributes of the vegetation have been suggested to be more important to bristlebirds than is the species composition. Baker (2000) found that bristlebirds were associated with dense low vegetation regardless of floristic composition. An investigation comparing structural attributes of the vegetation at the source population and in the host environment was undertaken. Transects were located in bristlebird habitat in the source environment and through potential habitat in the host environment. Vegetation plots ($n = 62$) were spaced 75 m apart along transects and consisted of two 1m^2 quadrats. Seventy-five metres was chosen as bristlebirds have been observed crossing at least this distance through unsuitable habitat with little or no cover (J. Baker, pers. comm.). Within each quadrat vegetation structure was measured.

To investigate vegetation structure, a 1.5 m X 2.5 cm pole was placed at the four corners and randomly in the middle of each quadrat and the number of vegetation touches on the pole within three height classes, 0 to 0.3 m, 0.3 to 1 m and 1 to 3 m, was recorded. The scores for cover were averaged over all five pole positions at each plot. The results were analysed using non-metric Multi-Dimensional Scaling and the similarities between the source and release environments were investigated using an Analysis of Similarity and the SIMPER procedure using the Primer 5.2.9 software package (Primer-E Ltd).

Results 1

Vegetation study

The structure of the vegetation in the source environment and host environment looked similar. An analysis of similarities revealed that there were differences in structure between the sites ($P = 0.003$), although overall the sites were unable to be easily separated (Global $R = 0.306$). The MDS plot in Fig 2.1 illustrates how similar the sites were, with sites that are more similar being closer together on the plot. The SIMPER procedure (Primer-E Ltd) revealed that the main difference between the two locations was that the vegetation in the host environment was denser than that in the source environment (Table 2.1).

Table 2.1: Number (\pm sd) of vegetation touches on the pole in the three height classes in the source and host environments.

<i>Height Class</i>	<i>Source Environment</i>	<i>Host environment</i>
1 m – 3 m	6.7 ± 2.2	5.7 ± 4.1
0.3 m – 1 m	6.7 ± 2.4	10.7 ± 2.4
0 – 0.3 m	6.8 ± 1.7	9 ± 3.4

The tendency for the vegetation on Beecroft Peninsula to be denser in the lower levels than the vegetation on Bherwerre Peninsula is not of concern as bristlebirds have been shown to prefer dense low vegetation (Baker 2000). Dense low vegetation may reduce risks of predation. The similarities between Bherwerre Peninsula and Beecroft Peninsula and their proximity to one another makes the latter an ideal host location for a reintroduction. Beecroft Peninsula is connected by only a narrow neck of land and surrounded on three sides by large water bodies (Fig 2.2). Fires in the Jervis Bay region are unlikely to burn both Beecroft Peninsula and the source population on Bherwerre Peninsula hence, this reintroduction has good potential for risk spreading for the bristlebird.

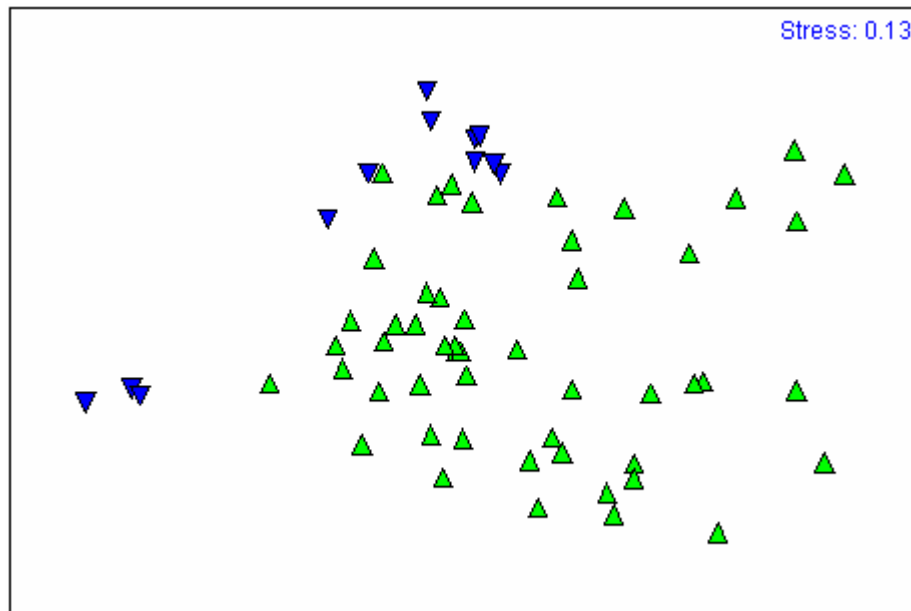


Figure 2.1: Non-metric Multi Dimensional Scaling plot of vegetation structure.

▲ the source environment, ▼ the host environment.

Methods 2

The reintroduction

Estimating numbers for translocation

In estimating the number of individuals needed to found a viable population, four elements of uncertainty need to be considered: demographic, environmental, genetic and catastrophic stochasticity (Shaffer 1981). At the start of this study, the sex of bristlebirds could not be determined whilst a bird was in the hand, so the demographics of the founder population was left to chance. The similarity of climatic conditions between the source and host environments and their close proximity suggest that environmental conditions will be similar at both sites. The main genetic factors that might reduce success is inbreeding depression as a result of a small founder population, especially if

post-translocation mortality further reduces genetic diversity. However, genetic theory predicts that species with historically small populations or those experiencing periodic population bottlenecks, such as the bristlebird (Smith 1977), will have previously purged any deleterious recessive genes, thus reducing the effects of inbreeding depression that may have been associated with a small founding population (Lacy 1992; Lande 1995). Natural catastrophes such as floods, droughts and fires can occur at unpredictable intervals and are very unpredictable in magnitude. Fire is managed with regard to bristlebirds at Beecroft Peninsula and this element of uncertainty did not influence the decision on the number of bristlebirds to relocate.

Population viability analysis (PVA) was not used to model different release scenarios, as currently life history data for the bristlebird is too lacking to allow a sensitive model to be developed (Harcourt 1995; Lindenmayer *et al.* 2003). However, a similar cover-dependant endangered Australian species, the Noisy Scrub Bird, *Atrichornis clamosus* has been the focus of numerous translocations. Only two translocations were successful from four attempts, the first with 18 males and 13 females in two releases over three years, and the second with 6 males and 5 females in three releases over three years (Danks 1994). The valuable lessons learnt during the Noisy Scrub Bird translocations were that the release of only a small number of individuals can lead to the establishment of a successful breeding population (Danks 1994).

A total of 45 bristlebirds was planned for the reintroduction, comprising of 15 birds each year for three years. These numbers represented a balance between minimising impacts of removal on the source population against the need to release a sufficient number of individuals to develop a viable new population. The removal of 15 bristlebirds each year was estimated to represent approximately 2.5 % of the source population on Bherwerre Peninsula (Baker 1997). It was considered that the source population could sustain this proportion being removed given that it is substantially less than the 14 % annual population growth rate estimated for the bristlebird population at Barren Grounds Nature Reserve (Baker 1997).

Capture of bristlebirds

The breeding season for the bristlebird extends from August till early February (Higgins and Peter 2002). As bristlebird calling activity peaks during that time, removals were planned approximately two months after the breeding season to maintain reasonable capture success while avoiding the removal of bristlebirds caring for juveniles.

Fifty-one bristlebirds were caught using methods developed by Baker and Clarke (1999). Approximately 100 m of mist nets were erected as one wall, along management trails within the national parks. Recorded bristlebird calls were played near the nets to attract bristlebirds and entice them to cross the net line. Often bristlebird calls imitated by researchers had the same effect. To minimise potential risks to birds, netting was only conducted when weather conditions were favourable: when it was not excessively windy, hot or wet. The capture of birds was restricted to the first five hours after sunrise to avoiding the heat of the day, and also so that released birds had an opportunity to find food and shelter before nightfall in their new habitat.

Processing of bristlebirds

Once caught, bristlebirds were placed into a calico bird bag prior to processing. Processing began within 10 minutes of capture and involved banding the bristlebirds, and taking measurements of weight, wing length, head-bill length, culmen length, tail length and tarsus length. To reduce stress a small quantity (few millilitres) of glucose solution was given via a dropper to each bird (Castro *et al.* 1994) and a small hood was placed over the head following measurement. After the banding and measuring, a DNA sample was taken by removing 2 to 6 pin feathers, which were stored in ethanol for later use in determining the sex of the bird. Then a small radio transmitter was attached using the methods of Baker and Clarke (1999). Transmitters were glued to the interscapular area of the birds, a common technique for radio transmitter attachment with birds (Raim 1978; Sykes *et al.* 1990; Johnson *et al.* 1991; Calvo and Furness 1992). Trialing another technique, in the second year nine individuals had a transmitter attached using a small backpack harness in addition to the glue (Bramley and Veltman 1998). A shoulder harness was used instead of a leg harness due to the ground-dwelling and semi-flightless

nature of the bristlebird (Rappole and Tipton 1991). The harness was made from easily degraded rubber bands that wrapped around the shoulders and included a weak link of cotton across the back. After the transmitter was attached, bristlebirds were placed in either a small (40 cm X 40 cm X 60 cm) foam lined bird cage filled with vegetation or a calico bird bag for transportation. All bristlebirds were released between 60 and 240 min after being captured.

Release of bristlebirds

Bristlebirds were transported by car to the release site, approximately 45 minutes drive away. Bristlebirds were released on Beecroft Peninsula using a hard release technique, involving opening the cage door or bag for the birds to leave at their own will. Hard release was used instead of soft release to reduce handling for the birds which are known to be very sensitive to stress (Baker and Clarke 1999). Soft release techniques also increase costs and personnel requirements. Two release sites approximately one kilometre apart were chosen on Beecroft Peninsula (Fig 2.2). Releases were made at site R₁ in the first year, site R₂ in the second year and at both sites in the third year. Fifteen bristlebirds were released in the first year, 20 in the second year and 15 in the third.

Success criteria

The timescale used for assessing success is important, such as years or a more biologically meaningful measurement such as generation time or the time to first breeding. Success should be measured in all parts of the translocation; not just within the host environment. This meant developing criteria for success in the source population as well as the release population using a biologically determined measurement of time. The 17 criteria by which this reintroduction was measured are in Table 2.2.

Table 2.2: Criteria for success (T_B = time to breeding age/generation time)

<i>Place</i>	<i>Criterion</i>
A. Pre-translocation	1. feasibility analysis favourable
B. Host environment	<p>Short-term (within 1 T_B)</p> <p>2. animals moved successfully</p> <p>3. animals survived settlement period</p> <p>4. animals survived first T_B</p> <p>5. evidence of any social behaviour</p> <p>Mid-term (1-3 T_B)</p> <p>6. population survived 3 T_B</p> <p>7. evidence of social behaviour, possibly breeding</p> <p>Long-term (>3 T_B)</p> <p>8. evidence of breeding</p> <p>9. population exceeds translocated number</p> <p>10. population survival for 10 T_B</p> <p>11. population survival for 20 T_B</p> <p>12. population survival for 50 T_B</p>
C. Wild source population	<p>Short-term (within 1 T_B)</p> <p>13. some sites of removal reoccupied within first T_B</p> <p>Mid-term (1-3 T_B)</p> <p>14. continued reoccupation of removal sites during next 3 T_B</p> <p>15. evidence of social interaction/breeding within 3 T_B</p> <p>Long-term (>3 T_B)</p> <p>16. evidence of breeding in removal sites within 5 T_B</p> <p>17. population \geq pre-removal within 10 T_B</p>

Monitoring of bristlebirds

Upon release, each bird was monitored using radio-tracking (Chapter 3). In the source population, the numbers and distribution of bristlebirds were monitored before and after the removals (Chapter 4). Over the three years the host environment was monitored using a combination of aural surveys along transects and call playback at point locations. Three transects, 2600 m, 1590 m and 4270 m, were located near the release sites (Fig 2.2). The transects and call playback locations were surveyed for bristlebirds two months before and six months after releases. Aural surveys involved slowly walking transects at 2 - 4 km/h, recording all bristlebirds seen or heard. Bristlebirds can be reliably mapped using these methods by competent observers (Baker 2001; Bain and McPhee 2005). The

transects were each surveyed twice, once in each direction. The survey with the maximum number of bristlebirds recorded was used for analysis. The number of bristlebirds per 500 m was analysed using a repeated measures analysis of variance to investigate any changes over time.

Elsewhere on Beecroft Peninsula, call playback was used to elicit calls from bristlebirds in the vicinity. This method allowed large areas to be surveyed to detect how far the translocated bristlebirds had dispersed. A tape of bristlebird calls was played for two minutes followed by a five minute listening period, the playback locations can be seen in Fig 2.2. With each detection an attempt was made to observe whether the bristlebird was banded, although attempts often failed due to the bird's cryptic nature and dense habitat. To estimate the total number of bristlebirds at Beecroft Peninsula each year, all birds from that years surveys were mapped. Any mapped bristlebird from a repeated survey that was within 200 m of a record from the previous survey was considered a recount. A radius of 200 m was used, as the area of the resulting circle (12.6 ha) approximates the ≥ 10 ha home range proposed by Baker (2001). As bristlebird home ranges have been recorded with up to 80 % overlap (Baker 2001), this is expected to give a conservative estimate of the number of bristlebirds at Beecroft Peninsula.

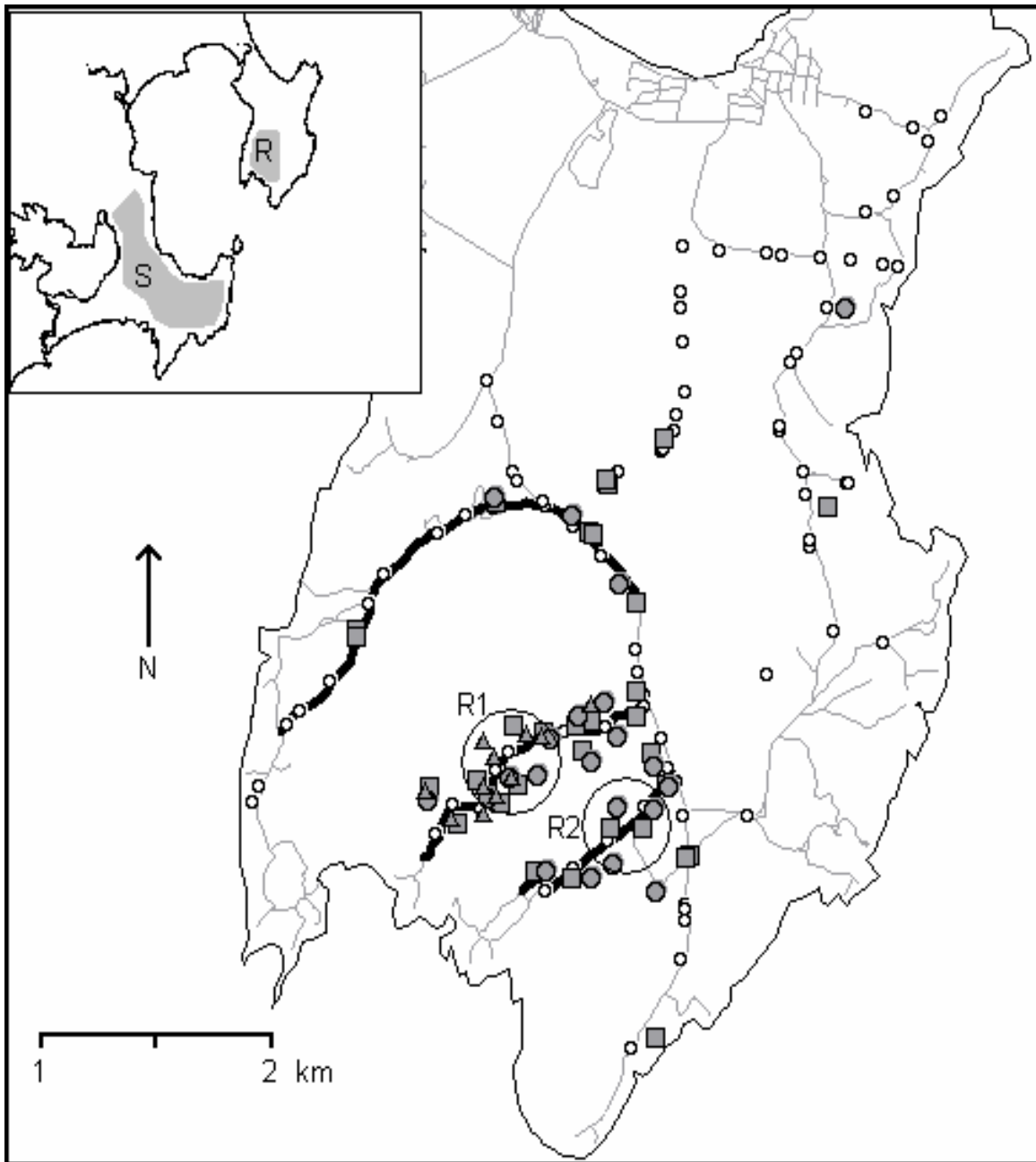


Figure 2.2: Beecroft Peninsula (Jervis Bay inset). 'S' denotes source population on Bherwerre Peninsula, 'R1' and 'R2' the release sites on Beecroft Peninsula. Thick black lines are transects, ○ are call playback locations, ▲ are mapped birds in 2003, ● are mapped birds in 2004 and ■ are mapped birds in 2005.

Cost of reintroduction

The costs of this project were compared with alternative methods of recovery of the bristlebird: the estimated costs associated with acquiring land to protect or regenerate further bristlebird habitat; and to a captive breeding program involving bristlebirds (DEC 2004). Costs have not been compared to ongoing management strategies, such as fire and feral predator control, in existing reserved lands containing bristlebirds. The costs considered here involve strategies to increase the number of individuals and the area of occupancy of the species.

Results 2

Monitoring and the success of the reintroduction

The reintroduction of the bristlebird to Beecroft Peninsula has initially been very successful. Thirteen of the 17 criteria have been achieved, one has not yet been achieved and 3 are as yet unknown. The longer-term success of the project still remains to be assessed, as not all timescales have been reached to assess all criteria. The assessment of criteria is presented in Table 2.3.

Fifty bristlebirds were translocated to Beecroft Peninsula over the three years. The population at the host site increased following the first two releases and was unchanged following the third release (Fig 2.3). This suggests that released birds were surviving for longer than one year in the host environment. However, repeated measures analysis of variance found that this increase over the three years was not significant at the 0.05 % level ($F_{5, 10} = 3.103$, $P = 0.06$), presumably owing to high variation between transects. Transect surveys together with the call playback surveys showed that translocated bristlebirds dispersed widely over Beecroft Peninsula (Fig 2.2), particularly after the

latter two releases. The maximum distance a bristlebird was detected away from the closest release site was 4 580 m in 2004 and 3 000 m in 2005.

Table 2.3: Assessing the success criteria.(T_B = breeding/generation time)

<i>Criteria</i>	<i>Achieved</i>	<i>Comments</i>
A. Initial		
1. feasibility analysis favourable	Yes	Completed prior to commencement (Whelan and MacKay 2002).
B. Released individuals		
Short-term		
2. animals moved successfully	Yes	One death during processing, no other fatalities
3. animals survived settlement period	Yes	Five deaths during first two days, no other recorded fatalities
4. animals survived first T _B	Yes	11 bristlebirds surveyed after first year
5. evidence of any social behaviour	Yes	Bristlebirds heard calling to one another within days of release
Mid-term		
6. population survived 3 T _B	Yes	30 bristlebirds surveyed after third year
7. evidence of social behaviour, possibly breeding	Yes	Duetting heard between bristlebirds. Two bristlebirds observed without bands
Long-term		
8. evidence of breeding	Yes	Two bristlebirds observed without bands
9. population exceeds translocated number	No	45 released and alive after settlement, 30 most recently surveyed
10. population survival for 10 T _B	Unknown	Project going for only four years
11. population survival for 20 T _B	Unknown	Project going for only four years
12. population survival for 50 T _B	Unknown	Project going for only four years
C. Source population (wild)		
Short-term		
13. some sites of removal reoccupied within first T _B	Yes	No change in numbers surveyed in source population (Chapter 5)
Mid-term		
14. continued reoccupation of removal sites during next 3 T _B	Yes	No change in numbers surveyed in source population across all three years (Chapter 5)
15. evidence of social interaction/breeding within 3 T _B	Yes	Duetting regularly heard between bristlebirds (Chapter 5)
Long-term		
16. evidence of breeding in removal sites within 5 T _B	Yes	Duetting regularly heard and replacement of removals. Although unknown origin of replacements (Chapter 5)
17. complete recovery within 10 T _B	Yes	No change observed in bristlebird numbers before and after removals (Chapter 5)
Success?	Yes (at 4 years)	13/17 Yes 3/17 Unknown 1/17 No

The total number of bristlebirds counted increased from year to year, from zero prior to the translocations to 30 birds, six months after the last translocation in 2005 (Table 2.4). The decreases seen in months 16 and 28 correspond to two summer surveys.

Bristlebirds were heard calling within days of release and continued to be heard for the duration of the program. Duetting, where two birds are heard calling to one another, was recorded within six months of the first release. This gave confidence that there was social interaction between translocated birds. Of the 30 bristlebirds surveyed on Beecroft Peninsula in 2005, eight of these individuals were observed closely enough to determine whether they were banded. Two bristlebirds were confirmed to be without bands and presumably these birds were the result of breeding on Beecroft Peninsula after the reintroduction. These birds were approximately 500 m and 3 000 m from the closest release site.

One bristlebird died, presumably from stress, during its processing and therefore represents a capture but not a translocation. Five bristlebirds were found dead after release. Four, all wearing the harnesses in the second year, were found dead following a severe storm on the night of their release and another was presumed to have been killed by a bird of prey due to the location and condition of the remains. One bird was recently found dead on the road, three years and 10 months after its release.

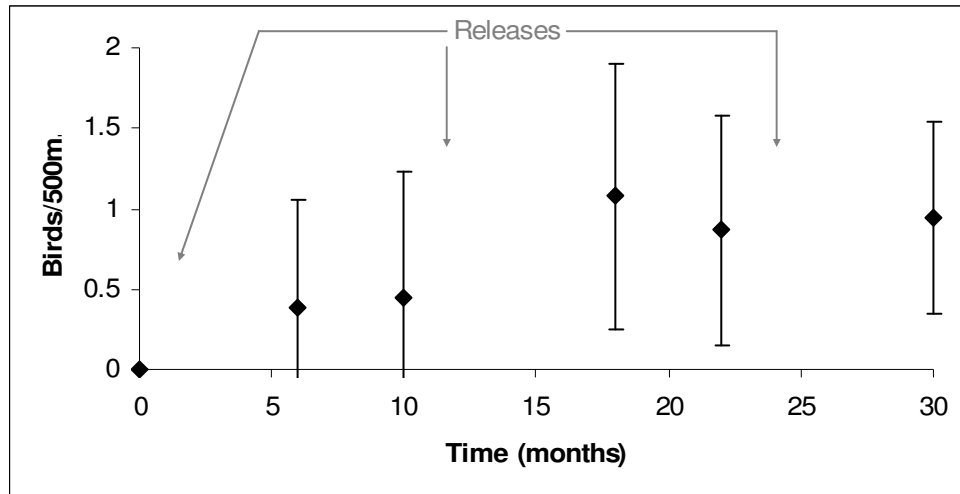


Figure 2.3: Mean number (\pm sd) of Eastern Bristlebirds surveyed per 500 m on three transects on Beecroft Peninsula. The timing of the release of translocated bristlebirds is indicated.

Table 2.4: Total number of Eastern Bristlebirds counted on Beecroft Peninsula and the timing and size (total) of the releases.

<i>Months</i>	<i>Number counted</i>	<i>Number released</i>
0 (Autumn 2003)	-	15
6 (Spring 2003)	11	-
10 (Summer 2004)	9	-
12 (Autumn 2004)	-	20 (35)
18 (Spring 2004)	21	-
22 (Summer 2005)	15	-
24 (Autumn 2005)	-	15 (50)
30 (Spring 2005)	30	-

Costs of recovery

This translocation is the least expensive conservation strategy for increasing the area of occupancy and spreading risks for the bristlebird from the options explored (Table 2.5). Land acquisition in the Jervis Bay area is very expensive and may not have as much potential to spread risks of catastrophe as much as translocation. Captive breeding is initially expensive but becomes cheaper once facilities are established. There are associated costs with the release of captive birds into the wild which have not been included here. When captive birds are finally released back into the wild they can be used to achieve similar outcomes to the translocation of wild individuals.

Table 2.5: Comparison of costs for the conservation of Eastern Bristlebirds from translocation, land acquisition and captive breeding. The four costing categories include 1) Personnel, covering wages for all involved staff. 2) Materials and Equipment, including all licences. 3) Consumables, covering all food, accommodation and vehicle expenses. 4) Reporting, including costs associated with publication of results and presentation at conferences. This table is only indicative of costs and other projects may vary considerably.

Discussion

It is not surprising that bristlebird numbers at the host site increased over the study, because 15 to 20 birds were added to the population by translocation each year. However, the steady increase in numbers of bristlebirds/500 m and total numbers at least confirms that birds are surviving. The decreases in bristlebird counts in months 16 and 28 correspond to summer surveys and are most likely due to different activity levels (ie. reduced detectability) at different times of the year (Higgins and Peter 2002). That bristlebirds were calling to each other within days of the first release gave confidence in proceeding with the reintroduction. This social interaction may have provided some stimulus for bristlebirds to not disperse widely from one another. It has been previously suggested that conspecifics can aid in settlement following translocation (Castro *et al.* 1994; Carrie *et al.* 1999).

There was very little mortality during the translocation process. Six from 51 bristlebirds (12 %) were known to have died during the translocation. The four birds that died during the storm event were all wearing the backpack harnesses to attach the radio transmitters. It was concluded that the harnesses had allowed the birds to become wetter than usual because of water travelling around the harness. Following this, the harnesses were not used anymore and the remaining transmitters were glued as before. A road-kill bird recovered in year 4 was banded as part of the year 1 translocation confirming bristlebirds can survive for at least three years in the host environment. However, this has raised a management issue for the new population. It is recommended that traffic advisory signs be placed along road at Beecroft Peninsula to educate drivers of the presence of this endangered bird.

The presence of 2 unbanded birds indicates that the environment may be capable of supporting breeding bristlebirds. Speculating on the growth of the population, I used a simple exponential growth equation $N_t = N_0 e^{rt}$ (Brewer 1988). I speculated using a growth rate (r) of 14 % estimated for another bristlebird population (Baker 1997). I did not use a growth rate calculated from population estimates in this study as these were

influenced by the addition of birds each year. Beginning with a population of 30 individuals at 3 years, after 7 years the population may have exceeded the number of released bristlebirds (50), reached 280 individuals after 20 years and will reach carrying capacity in approximately 28 years. Carrying capacity was estimated at 720 bristlebirds based on Baker's (2001) estimates of bristlebird density of 2 birds per 5 ha and 1 800 ha of core habitat in the bristlebird management area on Beecroft Peninsula (Pers. comm. Environmental Resources Management Pty Ltd 2006). However, these estimates are limited, as Baker's (1997) calculation of growth rate did not incorporate carrying capacity or migration factors. More complex population viability analysis (PVA) has not been used to model this population, as currently life history data for the bristlebird is too lacking to allow a sensitive model to be developed (Harcourt 1995; Lindenmayer *et al.* 2003).

The dispersal of some bristlebirds over 4 km from the release sites was unexpected. This provides some evidence of the potential of Beecroft Peninsula to support a population of over 720 bristlebirds because the majority of the Peninsula is potentially suitable bristlebird habitat. Another implication of this wide dispersal is the potential for fledged juvenile birds in existing populations to disperse over 4 km from their natal home ranges. Given available and connected habitat, it is suggested that bristlebirds may readily colonise new areas.

The achievement of 13 of the 17 criteria after four years indicates the current success of this reintroduction and gives confidence for the potential of more translocations to aid in the recovery of bristlebirds. The number of bristlebirds surveyed on Beecroft Peninsula still does not exceed the translocated number (Criterion 9) and is not predicted to for another four years. The project has not yet continued for long enough to assess the final three criteria that have not been achieved. Monitoring in the reintroduced population will continue over at least the next decade and after any monitoring, the criteria will be reassessed to follow the fate of this population and the long-term success of the project.

The criteria for assessing the success of this reintroduction were created to be adaptable to a range of species, as has been previously suggested (Dodd and Seigel 1991; Fischer and Lindenmayer 2000). One possible improvement could be the incorporation of a proportion in relation to criterion 3, allowing a quantitative measure. This proportion is likely to be species specific and determined through the development stage of the program. The use of generation time or the time to breeding age instead of years means that these criteria can be used in translocations of a range of species. The criteria will also allow a meaningful comparison, between any two translocations using these criteria, of the state of success during or following the translocation. This could be valuable in understanding areas of concern, or methods that may be broadly applicable to translocations in general.

The reporting of success and other outcomes of translocation projects is important for developing this area of conservation. Currently there is a lack of published outcomes from translocation projects (Copley 1994; Fischer and Lindenmayer 2000). This is possibly a result of land managers conducting many translocation programs who are not focused on publishing scientific papers. The policy documents IUCN (1987), Australian National Conservation Agency (1994) and NPWS (2001) all stipulate the publication and availability of information as a priority in translocation projects. This could be achieved by the development of a register of translocation projects that could be managed during the licensing process for wildlife research. As part of the register, newsletters such as the Reintroduction Specialist Group Oceania Newsletter (IUCN) could be used to publish reports on current translocation programs.

Translocation can be expensive. Of potential strategies for the conservation of bristlebirds, this reintroduction was the least expensive in achieving the goal of increasing the number of populations. Comparatively it was \$231 100/bristlebird cheaper than estimated for land acquisition and approximately the same for maintaining a bristlebird in captivity for one year, although not including the costs associated with future release. There was a staggering difference between other reports of the costs of translocating animals, with US\$300 000 spent per Californian Condor (*Gymnogyps californianus*)

released (Cohn 1993) and US\$22 000 per Golden Lion Tamarin (*Leontopithecus r. rosalia*) released (Kleiman *et al.* 1991) compared to AU\$2 900 per bristlebird released. Much of this difference between these projects and this bristlebird reintroduction was that these projects kept animals in captivity for extended periods, incurring large costs associated with animal care. The capture and release of wild caught animals will always be the cheapest translocation option available.

This translocation has been very effective. It has achieved initial goals of successfully translocating individuals and establishing them in the host environment. The program has run efficiently and within budget and produced outcomes, such as the criteria for success, that are useful to a wide range of translocation programs. The goal of increasing the number of populations of bristlebirds (NPWS 2000a) has the potential to increase the security of the species and, in the future, may contribute to it being down graded from endangered to vulnerable under the NSW *Threatened Species Conservation Act 1995*. However, it is important to note that an increase in the number of populations does not necessarily mean that the species is better off. If individual populations become less viable as a result of a translocation then the benefits to the species are lost. It would be valuable to quantify the net gain to a species through the evaluation of the viability or security of all populations throughout translocation programs. This is particularly important in relation to the source population (see Chapter 5).

Future translocations based on the methodology used at Jervis Bay may be used to further increase the security of bristlebirds, although this will only be viable where suitable source populations exist. In areas such as southern Queensland and northern New South Wales, where there are very few bristlebirds remaining (DEC 2004), translocation will be more expensive as captive breeding may be the only viable method for securing individuals for translocation.

Two potential sexing techniques for the Eastern Bristlebird

A manuscript submitted to

Australian Zoologist Royal Zoological Society of New South Wales

This chapter has been written as a manuscript and as such it will contain some repetition in the introductory section from Chapter 1 and some overlapping in the methods section with other data chapters. All species are reintroduced.

Chapter 3

Two potential sexing techniques for the Eastern Bristlebird

Introduction

Over fifty percent of bird species worldwide are sexually monomorphic (Griffiths *et al.* 1998), meaning that there are no discernable differences in the physical appearances of males and females. This similarity between the sexes can often lead to difficulties in undertaking ecological research on these species. The knowledge of sex ratios is particularly important in the conservation of small populations (Double and Olsen 1997; Lens *et al.* 1998). To determine the sex of an individual in sexually monomorphic species three options are generally available: (i) conduct intensive behavioural studies, (ii) undertake a laparoscopy on individuals to look for testes or ovaries, or (iii) take a DNA sample and use molecular sexing techniques, once genetic markers for sex have been established for the species.

The ability to sex an individual quickly and cheaply in the hand is vital in translocation or captive breeding programs, especially when only small numbers of birds can be captured and it is essential that both males and females are included in the sample. In such situations, behavioural studies or laparoscopy may not be appropriate because of the time required or potential risks to the individuals respectively. Molecular genetic sexing techniques are extremely valuable because they pose little risk to individuals and are practical in most applications. The drawbacks to molecular sexing techniques can be the time lag between sample collection and analysis and, in some circumstances, the cost.

What is needed in conservation programs is a quick and cheap method of distinguishing sex that is applicable in the field.

Many of Australia's birds are of conservation concern and some of these species show little or no sexual dimorphism. Examples include the Western Bristlebird (*Dasyornis longirostris*), Brown Thornbill (King Island subspecies - *Acanthiza pusilla archibaldi*), Ground Parrot (*Pezoporus wallicus*) and Eastern Bristlebird (*Dasyornis brachypterus*) (Higgins 1999; Garnett and Crowley 2000; Higgins and Peter 2002). The Eastern Bristlebird is listed as endangered under Australia's *Environment Protection and Biodiversity Conservation Act 1999*. It has been considered a sexually monomorphic species (Simpson and Day 1996; DEC 2004), although Chaffer (1954) suggested that, while both sexes look superficially similar, the females may be smaller than the males.

This study investigates sexual dimorphism in the bristlebird by investigating the relationship between morphological features and sex in individuals and specimens of bristlebirds collected over the last 141 years. It develops and evaluates two types of criteria for determining sex in the field, a set of morphological criteria and a discriminant function analysis.

Methods

Data Collection

Morphometrics were collated on 17 museum specimens of bristlebirds collected from 1864 to 1993 and deposited with the Australian Museum, (Sydney) and the Australian National Wildlife Collection, (Commonwealth Science and Industrial Research Organisation, Canberra). The sex of these birds was provided on the specimens. It is acknowledged that museum specimens may be subject to some shrinkage. The data from museum specimens were pooled with data from live bristlebirds to increase the small

sample size. Between 2003-2004, 31 bristlebirds were caught and the following morphological characters were measured: weight, flattened straightened wing length, tail length, tarsus length, head-bill length and culmen length (Rogers 1989). Three pin feathers were removed from each bird for DNA analysis, performed by the Museum of Victoria (Dr Janette Norman pers. comm.). A common DNA molecular sexing technique exploiting birds heterogameticity (i.e. male has ZZ sex chromosomes and female ZW) was used to determine the sex of each individual. This technique utilises polymerase chain reaction (PCR) to amplify two chromo-helicase-DNA-binding genes located on the sex chromosomes of most birds, expressed using gel electrophoresis (see (Griffiths *et al.* 1998; Bermudez-Humaran *et al.* 2002) for details). The age of the birds was unknown but were all considered to be adults as capture techniques were presumably biased towards the dominant individuals in the location.

Sexing Criteria

Student *t*-tests were used to determine differences between males and females, whose sex had been determined by DNA, in individual morphological measurements. Those measures that revealed statistically significant differences between males and females were considered to be suitable for sexing. As an independent test of the measures that were finally selected, another 15 bristlebirds were caught and measured and pin feather samples were taken for DNA analysis. They were categorised using the sexing criteria and the results compared to known sexes, based on the molecular analysis.

Discriminant Function Analysis

The morphological data were subjected to multivariate analysis by discriminant function analysis using the SPSS 12.0.1 statistical software package. Discriminant function analysis has been used previously to create morphological sexing techniques for many species (Lorentsen and Rov 1994; Glahn and McCoy 1995; Zavalaga and Paredes 1997).

The discriminant function analysis involved all bristlebirds that were not missing data ($n = 46$). The unstandardised discriminant function was then cross validated, where each case is classified by the function derived from all cases other than that case (Rencher 2002). The assumption for discriminant function analysis that there is homogeneity of within-group variance-covariance matrices was tested using Box's M-statistic test (Rencher 2002).

In both determining criteria for sexing and the discriminant function analysis, not all measures were available for all birds. Hence, sample sizes vary in the following analyses.

Results

Sexing Criteria

The DNA results were returned stating the sex of each bird. Male bristlebirds were heavier and had longer wings, tail and head-bill lengths than females (Fig 3.1, weight $t_{33} = 5.67$, $P < 0.001$, wing length $t_{46} = 3.81$, $P < 0.001$, tail length $t_{45} = 3.39$, $P = 0.001$ and head-bill length $t_{29} = 3.59$, $P = 0.001$). As wing length and tail length are subject to error resulting from wear during the year and the difficulties of repeatable accurate measurement of these morphometrics, these measures have not been included in the development of sexing criteria. Weight had the largest relative distance between the sexes means which is useful for developing criteria, although it can also be annually and diurnally variable. Combining weight with a measure of body size such as head-bill length was thought to provide a correcting influence for likely variations in weight.

To determine criteria for sexing bristlebirds, the pooled standard deviation (Afifi *et al.* 2004) of the morphometric measurements were used to develop upper and lower bounds for weight and head-bill length (Table 3.1). One pooled standard deviation below the male mean was used as an upper limit for females and one pooled standard deviation

above the female mean was used as a lower limit for males (Table 3.1). The sex of an individual was indicated by one or both of the measures. If both sexes were indicated, each by one measure then the sex of that bird was recorded as inconclusive.

Measurements falling between the male and female bounds were also classed as inconclusive. When weight and head-bill length were combined, of the 15 test birds 12 agreed with DNA sexing (80%), one disagreed (7%) and two were inconclusive (13%). Assuming my sample of bristlebirds is representative and weight and head-bill length in bristlebirds are normally distributed within each sex, then the probabilities of mis-identification are shown in Table 3.1.

Table 3.1: Morphological sexing criteria for the Eastern Bristlebird. Male lower boundary approximates the female average plus one pooled standard deviation (sd_p), female upper boundary approximates the male average minus one pooled standard deviation. Values falling between the male and female bounds are classed as inconclusive. The sex of an individual is indicated by one or both of the measures. If both sexes were indicated, each by one measure then the sex of that bird was recorded as inconclusive. Probabilities come from a normal distribution with the corresponding means and standard deviations.

	<i>Weight (g)</i>	<i>Head-bill length (mm)</i>
Female, $av + sd_p$	$39.2 + 1.7$	$39.5 + 0.9$
(n)	(16)	(14)
Male rules	> 41	≥ 40.5
Prob. female $>$ male boundary	0.18	0.14
Male, $av - sd_p$	$42.4 - 1.7$	$40.7 - 0.9$
(n)	(19)	(17)
Female rules	< 41	≤ 40
Prob. male $<$ female boundary	0.20	0.21

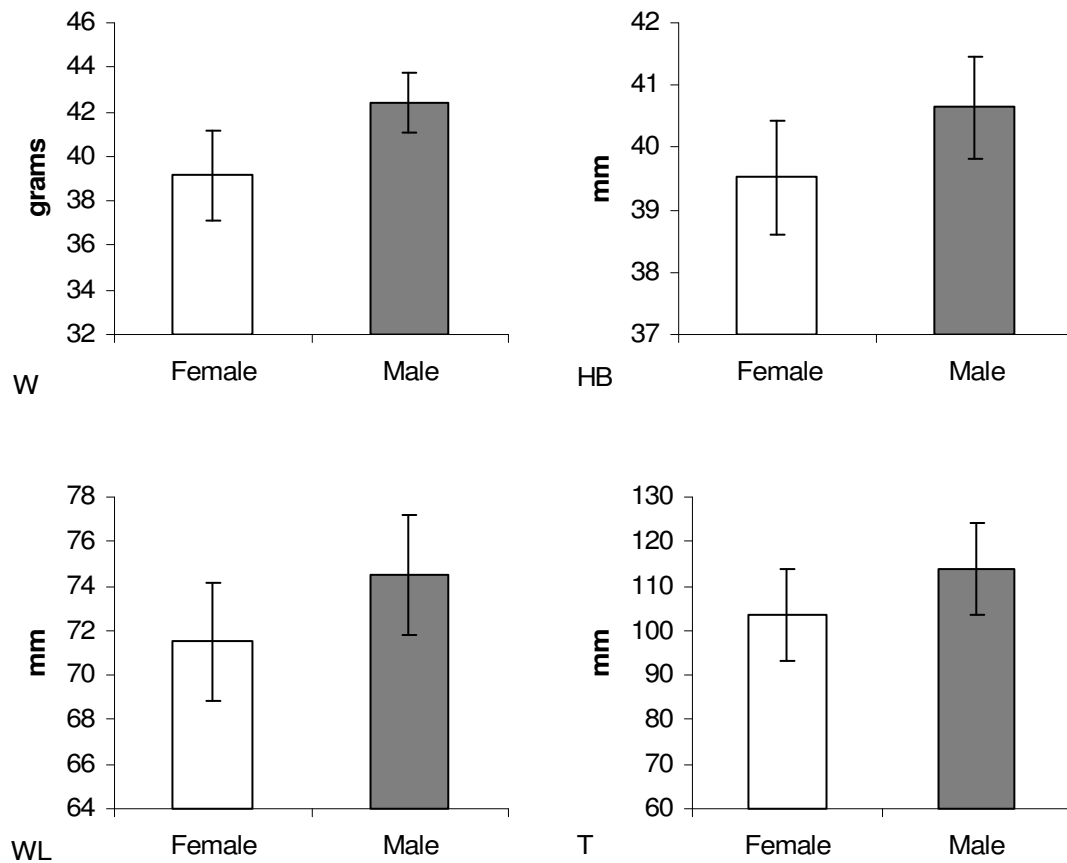


Figure 3.1: Morphological measurements (mean \pm sd) tested for use in developing sexing criteria. W: weight, HB: head-bill length, WL: wing length and T: tail length.

Discriminant function analysis

The assumption of homogeneity of within-group variance-covariance matrices was met as they were not significantly different (Box's M-statistic = 0.795, $F_{3, 248075.5} = 0.252$, $P = 0.86$). By incorporating the same characters, weight (W) and head-bill length (HB) simultaneously into a discriminant function analysis, the following unstandardised discriminant function was obtained:

$$D = -31.388 + 0.505(W) + 0.271(HB),$$

where $D > 0$ is male, $D < 0$ is female.

Using a cross-validation technique where each case is classified by the function derived from all other cases other than that case, 40 of 46 birds agreed with the DNA sexing (87%). Two females and four males were misclassified.

By examining the discriminant scores, no bird with a score above 0.27 (males) or a score below -1.02 (females) was misclassified. In estimating the probability of correctly allocating sex based on this discriminant function using Afifi *et al.* (2004) posterior probability equation, for greater than 75 % confidence in correctly sexing birds, then discriminant scores between -1.099 and 1.099 should be classed as inconclusive. The reliability of this function now dropped, to 46 % agreeing with DNA sexing and 54 % inconclusive.

Discussion

The results suggest that both the discriminant function and the sexing criteria can be used to sex adult bristlebirds. The discriminant function is easily applicable to existing data sets if there are no missing measurements, although with the inconclusive bounds it is very conservative. The sexing criteria may be more applicable in the field as no calculation is required and, in some instances, an individual can be accurately sexed even if one measurement is missing. It is suggested that the discriminant function can be used initially and then the sexing criteria can be used to re-examine any inconclusive individuals.

Using this technique, the bristlebird data were re-assessed. The discriminant function left 25/46 bristlebirds inconclusively sexed, of which, the sexing criteria correctly sexed 16/25, wrongly sexed 3/25 and left 6/25 inconclusive. Overall 80 % (37/46) agreed with DNA sexing, 7 % (3/46) disagreed with DNA sexing and 13 % (6/46) remained inconclusive.

When re-examining inconclusive individuals from the discriminant function, the sexing criteria possibly over-estimated female numbers by incorrectly sexing juvenile males. The three wrongly sexed individuals were all males. The inconclusive individuals included two females and four males. It is possible that the wrongly sexed and inconclusive males were juveniles, all were smaller overall than the other males and the netting of bristlebirds occurred soon after breeding season.

A likely explanation for the incorrect identifications is the classification of juvenile individuals. It was assumed that the majority of bristlebirds caught during this study were adults. With territorial bird species, call-playback should attract nearby territory holders (Newton 1992). Bristlebirds display some territorial behaviour (McNamara 1946; Chapman 1999; Baker 2001; Higgins and Peter 2002) and presumably the majority of bristlebirds attracted to the call playback and mist nets were adults that had reached sexual maturity and obtained home ranges. However, there was no conclusive evidence that all individuals were adults. Another source of error may be the possible shrinkage of the museum specimens, although these were not over-represented in the inconclusive or wrongly sexed individuals.

In using measurements of adults to categorise individuals, there will inevitably be restrictions on their application to juveniles. For bristlebirds, I suggest that caution be used in applying this technique during the months January to April and on individuals with a pale brown to brown iris, thought to be juveniles, rather than the red to red brown as with adults of the species (Higgins and Peter 2002). It is expected that during this period, results may under-estimate males and over-estimate females.

The bristlebird is an endangered species and the value of this sexing technique is in its application to conservation research. This study provided valuable support to the translocation project described in Chapter 2. Being able to determine the sex of individual birds during a translocation project, without waiting or paying for laboratory work, may provide substantial reductions in the effort and resources required to develop a viable translocated population. The bristlebird is sensitive to disturbance and particularly

susceptible to handling stress (Baker and Clarke 1999); the need to minimise handling is high. Future translocation projects will benefit from a quick field based sexing technique that minimises handling while preventing grossly skewed sex ratios of translocated birds.

Integration of this sexing technique into current research could be used to quickly and cheaply estimate sex ratios in remnant populations, to enhance our understanding of this endangered bird. However, the expansion of this analysis over a wider geographic range and on a larger sample size may be needed to increase the accuracy of the techniques. The knowledge of sex ratios in wild populations is an important component of conservation (Millar *et al.* 1997) as any attempts to develop captive populations, new populations or augment existing populations needs to understand the sex composition of the breeding system. The investigation of sex ratios has assisted in the conservation of threatened birds around the world (Double and Olsen 1997; Komdeur *et al.* 1997; Lens *et al.* 1998).

A number of birds in the Pardalotidae family in Australia are of conservation concern (Garnett and Crowley 2000). These include two other species of bristlebird, the Western Bristlebird and the Rufous Bristlebird (*D. broadbenti*), as well as others such as the Scrubtit (King Island subspecies - *Acanthornis magnus greenianus*), Slender-billed Thornbill (Western subspecies - *Acanthiza iredalei iredalei*) and the Brown Thornbill (King Island subspecies). It has been suggested that there is no size dimorphism between the sexes in these species (Simpson and Day 1996; Higgins and Peter 2002), although a similar situation has been eluded to in the Western Bristlebird as has been found here for the Eastern Bristlebird (A. Burbidge pers. comm. 2007). This sexing technique may have applications in the ongoing research projects currently investigating the conservation of these other species of bristlebird and pardalotids.

Post-release dispersal of reintroduced Eastern Bristlebirds

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This chapter has been written as a manuscript and as such it will contain some repetition in the introductory section from Chapter 1 and some overlapping in the methods section with other data chapters. All species are reintroduced.

Chapter 4

Post-release dispersal of reintroduced Eastern Bristlebirds

Introduction

Animals dispersing into novel habitats are expected to exhibit different behaviour patterns than those that are in familiar habitats (Davis 1983; Ruxton *et al.* 1997). Animals might investigate a new habitat widely in search of the best areas for foraging or nesting. Alternatively, animals might respond more cautiously showing increased vigilance and alertness. Lack of familiarity is likely to result in higher levels of stress and behaviours that are poorly matched to the surrounding conditions, resulting in increased mortality (Ruxton *et al.* 1997; Dale 2001). Dispersal is the time of maximum mortality in many animal species (Bonnet *et al.* 1999).

Increases in mortality might be more pronounced when individuals are translocated (Armstrong and McLean 1995) because animals may suffer increased stress during translocation. Measuring differences in behaviour between translocated individuals and those in familiar habitat is likely to give insights into the suitability of the habitat and settlement behaviour. Patterns across species may illustrate generalisations useful for improving the success of many translocation projects.

The translocation of threatened species is becoming common although our understanding of how species may respond is relatively poor (Wolf *et al.* 1996; Wolf *et al.* 1998; Fischer and Lindenmayer 2000). Inadequate monitoring of released individuals (Short *et al.* 1992; Fischer and Lindenmayer 2000) and a lack of comparisons with natural behaviours and survival patterns, continue to limit the understanding of

mechanisms behind the success or failure of translocation (Wolf *et al.* 1996; Clarke and Schedvin 1997; Fischer and Lindenmayer 2000). While translocated animals can be monitored through retrapping, colour banding, thread and spool and searches for evidence of presence, radio-tracking is the most informative and popular monitoring technique used, despite its expense (Kleiman *et al.* 1991; May 1991). Radio-tracking has been used to monitor translocated individuals in a range of species e.g. Yellow-shouldered Amazon Parrots (*Amazona barbadensis*) (Sanz and Grajal 1998), Rufous Hare-wallabies (*Lagorhynchus hirsutus*) (Gibson *et al.* 1994). For many cryptic species or species released into a large area of habitat, radio-tracking may be the only feasible way to monitor their dispersal. The period of radio-tracking is limited by the battery life of the transmitter and small species can only be monitored for a short term. However, monitoring individuals during the initial release period, when mortality is likely to be greatest, can yield important information on initial survival and dispersal of individuals (Bright and Morris 1994).

Dispersal following translocation may have many similarities to natural dispersal. Studying the dispersal of translocated animals may give insights into general dispersal mechanisms. Conversely, measurements of the dispersal and behaviour of translocated individuals can be difficult to interpret unless compared to the movement and behaviour of the species in its native home-range. Translocated sub-adult, male Brush-tailed Phascogales (*Phascogale tapoatafa*) dispersed over twice the distances of juveniles that were trapped, tagged and monitored in their natal habitat (Soderquist 1994). However, Danks (1991) reported the opposite effect. Young adult Noisy Scrub-birds (*Atrichornis clamosus*) dispersed up to 8 km from their natal home-ranges, whereas translocated Noisy Scrub-birds were recorded dispersing up to only 5 km from the release point (Danks 1994).

A number of species show differential sex-related dispersal behaviours. One theory, based on reducing the amount of inbreeding, suggests that one sex chooses breeding territories while the other sex disperses to find a mate or mates (Wolff and Plissner 1998). Greenwood *et al.* (1979) found evidence of female biased dispersal in the Great Tit (*Parus*

major), with only 10 % of females establishing their first territory on, or adjacent to, their natal one compared to 25 % in males. In the Splendid Fairy-wren (*Malurus splendens*), Russell and Rowley (1993) also found a female bias in dispersal, with 13 % of males and 24 % of females leaving their natal territory within their first year. Alternatively, females may disperse less than males as a result of limited mate searching or dispersal due to energetic and resource-based limits, such as a restricted breeding season. These limitations may decrease breeding success for females that spend an excess of time dispersing and finding a mate (Dale 2001) such as has been found with female Pied Flycatchers (*Ficedula hypoleuca*) (Slagsvold *et al.* 1988).

With translocated mammal species, males generally disperse further than females. Davis (1983) found that during the first year translocated male Martens (*Martes americana*) made more wide ranging movements than females. Short and Turner (2000) translocated Burrowing Bettongs (*Bettongia lesueur*) and found males dispersed further than females in one week, one month and three months after release. However, Dufty *et al* (1994) found male translocated Eastern Barred Bandicoots (*Perameles gunnii*) moved significantly more than females but took longer to disperse in the initial five months after translocation. In contrast, the few translocations of birds reporting male and female dispersal have shown no such trend. Armstrong and Craig (1995) found no difference in dispersal between male and female translocated Saddlebacks (*Philesturnus carunculatus rufusater*) and Castro *et al* (1994) found translocated female Hihi (*Notiomystis cincta*) moved over a greater area than translocated males.

Dispersing individuals are often affected by the presence of conspecifics. There is evidence that new arrivals to an area are attracted by residents of the same species (Smith and Peacock 1990). Serrano *et al.* (2001), for example, found that Lesser Kestrels (*Falco naumanni*) dispersed further the lower the breeding density was in the surroundings. Stamps (1991) found that for the lizard *Anolis aeneus*, settlement rates of dispersing individuals increased with an increase in the number of conspecifics in the area, suggesting that individuals assess habitat quality indirectly by cueing on the presence of conspecifics. Carrie *et al.* (1999) found that releasing pairs of translocated Red-cockaded

Woodpeckers (*Picoides borealis*) in close proximity to other released birds and resident groups provided the necessary social interaction for them to settle.

Presumably, the effects of resident conspecifics on translocated birds would be affected by the amount of vacant habitat available and the social organisation of the species. It may be predicted that communally breeding species might resist dispersing individuals that are new in the area whereas in species that breed in pairs, conspecifics may attract dispersing individuals. Clarke and Schedvin (1997) released the communally breeding Noisy Miner (*Manorina melanocephala*) into areas containing resident colonies and found that translocated birds dispersed large distances and were not assimilated into the resident colonies. Conversely, 68 Orange-bellied Parrots (*Neophema chrysogaster*), which breed in pairs, were released into summer breeding habitat containing conspecifics. They were assimilated into the population and most formed pairs with residents or other released birds (Smales *et al.* 2000).

The aims of this study were to investigate the dispersal and home range of translocated Eastern Bristlebirds (*Dasyornis brachypterus*) in the initial period following release. To examine differences in dispersal, the movements of translocated bristlebirds were compared to movements in native habitat, between males and females and between bristlebirds released into habitat without conspecifics compared to birds released into habitat containing previously released conspecifics.

Methods

The study was undertaken on the Beecroft Peninsula (150°48', 35°03') at Jervis Bay, NSW. Over three years between late March and early May, bristlebirds were caught on Bherwerre Peninsula and translocated to Beecroft Peninsula where they were released at one of two release sites within a single area of suitable habitat. In the first year, 15 bristlebirds were released at Site 1 and in the second year 20 bristlebirds were released at

Site 2. The third year was the second phase of releases for these sites, with eight bristlebirds released at Site 1 and seven bristlebirds released at Site 2.

Bristlebirds were caught using mist nets along fire-trails through known bristlebird habitat. Call playback was often used to attract birds to run across fire-trails and into nets. Mist nets were set with the lowest pocket on the ground as bristlebirds generally run across open ground rather than fly. Mist nets were opened approximately 15 minutes before sunrise and closed by midday if no bristlebirds were caught. Trapping methods closely followed the methods of Baker and Clarke (1999).

Small radio transmitters (LTM single stage transmitter, 12mm x 5mm x 2.5mm in size with a 250mm aerial, from Titley Electronics, Ballina, NSW) were attached following the methods of Baker and Clarke (1999). The transmitters weighed on average 3.6 % of the mass of the translocated bristlebirds, below the 5 % limit suggested (Raim 1978; Johnson *et al.* 1991). Prior to transmitter attachment, a small square of cotton gauze was glued to the base of the transmitters to help in the attachment process. Generally transmitters were glued to the interscapular area of the bristlebirds using 10 second Supa Glue (Shelleys®) once a small area of feathers in the shape of the transmitter were removed with blunt-nosed scissors. This glue is a cyanoacrylate which has previously been used safely with the bristlebird (Baker and Clarke 1999) and other passerines (Johnson *et al.* 1991). In an attempt to have the radio transmitters stay attached to the birds for a longer period, in the second year, nine individuals had a transmitter attached using a small backpack harness in addition to the glue (Bramley and Veltman 1998). The harness was made from easily degraded rubber bands that wrapped around the shoulders and included a weak link of cotton across the back. Both techniques were designed to have the transmitters fall off the birds before the end of the battery life (approximately six weeks) so transmitters could be recovered. Attaching the transmitter was a stressful experience for bristlebirds as they are known to be extremely sensitive to handling stress (Baker and Clarke 1999). To reduce stress, a small quantity (few millilitres) of glucose solution was given via a dropper to each bird (Castro *et al.* 1994) and a small hood was placed over its head prior to processing. Once the glue dried (approximately 3-5 minutes) bristlebirds were placed in

either a small (40cm X 40cm X 60cm) foam lined bird cage filled with vegetation or a calico bird bag for transportation. Within 1.5 hours of a bristlebird's capture, it was transported in a car for approximately 45 minutes to Beecroft Peninsula. Bristlebirds were released immediately upon arrival to the release sites.

After release, bristlebirds were radio-tracked every hour for the first five days between sunrise and sunset. After the fifth day, bristlebirds were only radio-tracked once in the morning (before 0800 hrs), once at mid-day (1130 - 1330 hrs) and once in the late afternoon (after 1500 hrs). Radio-tracking involved triangulation for location fixes from numbered positions along trails through the release site. Two to five bearings were taken when locating a bristlebird, depending on its position in relation to the trail. Radio-tracking was carried out on trails as the vegetation was dense and an observer often had to move quickly between radio-tracking positions to obtain accurate bearings for a transmitter location.

During the first year, hand-held antennas (AY/C, three element Yagi from Titley Electronics, Ballina, NSW) were used in conjunction with Telonics TR2 and TR4 receivers. In the second and third years, the same hand-held antennas and receivers were used in conjunction with four temporary towers, 5 metre tall, with similar hand-held antennas attached to their top. These towers increased the range of detection by rising above the vegetation and much of the topography, which can affect signal strength from the transmitters (pers obs, Pyke and O'Connor 1990). Location fixes were calculated using LOCATE II (Nams 1990) using tracking and tower positions recorded with a GPS and the associated bearings to each bird. These data were then mapped using ArcView GIS 3.3 (ESRI Inc.).

Five aspects of the translocated bristlebirds' movements were investigated to examine the distances over which translocated bristlebirds moved. (1) The maximum distance away from the release point each day, (2) the maximum distance moved in a day from the last position the day before and (3) home range size over four-day periods. Two aspects examined the faithfulness to an area and the rate of movement, (4) the distance between

positions on consecutive mornings and (5) the average distance moved per hour per day. Where possible, comparisons were made between sexes and between birds in the first or second phase of releases into the two release sites. First phase birds were released at the separate sites in years one and two, into habitat containing no conspecifics. Second phase birds were released in the third year at both of the previous release sites, containing the bristlebirds from the previous two years. Analysis of variance was used for comparisons and a Huynh-Feldt epsilon degrees of freedom correction for violating the sphericity assumption of a repeated measures analysis of variance was used as needed.

The movements and activities of the translocated bristlebirds were compared to those of bristlebirds in the source population, which were radio-tracked by Baker and Clarke (1999) and Baker (2001).

Home ranges were calculated using the Animal Movement extension for ArcView GIS 3.3 (Hooge and Eichenlaub 2000). Both the minimum convex polygon (MCP) (Anderson 1982) and kernel utilisation distribution (UD) (Worton 1987) were calculated, for direct comparison to previous research. The minimum convex polygons were calculated over four-day periods using only three location fixes per day, morning, mid-day and late afternoon, to maximise sample sizes with an equal survey effort. The calculations for the kernel utilisation distribution used all location fixes available for birds over the first four days only. Repeated measures analysis of variance was used for statistical analysis, which requires that there are no missing values. This was not always achievable with all tracked birds. Because of missing values, sample sizes in the analyses vary between analyses and often from day to day. Analyses have been carried out with a compromise between retaining sample sizes and maximising numbers of days in the analysis. The data were not normally distributed and were transformed before analysis using the square root transformation (Bartlett 1936) as group variances were proportional to the means (Zar 1984). Graphs and tables report untransformed data to enable direct comparisons to other studies.

Radio-tracking error, the error between the calculated transmitter position and the actual transmitter position is important to estimate, particularly when quantifying habitat use by animals (Kauhala and Tiilikainen 2002). Precise estimations of bristlebird locations were not critical for this study because habitat use was not primarily of interest. Radio-tracking error was estimated when transmitters had fallen off bristlebirds, by using the difference between the location where a transmitter was recovered (using a GPS) and the last triangulated location. The last location was triangulated after it was concluded a transmitter had fallen off a bird, usually suggested by failing to detect any movement from a transmitter for more than a day. Tracking error was analysed using a regression against the distance from the closest tracking position.

Results

Native versus new habitat

In the initial post-release period, transmitters were retained for a similar time in both studies. Translocated bristlebirds had larger maximum movements and moved through much larger home ranges than birds in the source population. Interestingly the average hourly movements were similar for both groups of birds. However, translocated bristlebirds ranged through MCP areas over five times the size of home ranges of non-translocated adult individuals in the source population (Table 4.1).

Table 4.1: Comparison of radio-tracking data between Eastern Bristlebirds tracked in their native habitat compared to bristlebirds tracked following their translocation to a new area. Statistics presented are averages with the range in parentheses.

Maximum distance away from the release point each day

All bristlebirds

During the first ten days there was a significant effect of time since release ($F_{9, 45} = 3.324$, $P = 0.03$) on the maximum distance bristlebirds moved from the release point (Fig 4.1). On their first day, bristlebirds averaged a maximum of 409 ± 271 (sd) m from the release point and by their tenth day their maximum distance was 898 ± 446 m from the release point. As sample sizes were small ($n = 7$) another two analyses were performed with larger sample sizes, for the first eight days ($n = 12$) and across the first five days ($n = 20$). Time had a significant effect on distance from the release point over both time periods (8 days, $F_{7, 70} = 2.506$, $P = 0.023$, 5 days, $F_{4, 72} = 5.536$, $P = 0.001$).

Differences between sexes

There was a significant difference between sexes ($F_{1,5} = 10.673$, $P = 0.022$) with males consistently further away from the release point than females, ranging from 26 m away on day 1 to 659 m away on day 9 (Fig 4.1). As sample sizes were small ($m = 5$, $f = 2$) another two analyses were performed with larger sample sizes, for the first eight days ($m = 8$, $f = 4$) and across the first five days ($m = 11$, $f = 9$). Males were significantly further away from the release point compared to females over eight days ($F_{1,10} = 6.12$, $P = 0.033$) but not over five days ($F_{1,18} = 1.249$, $P = 0.278$) (Fig 4.1 and 4.3).

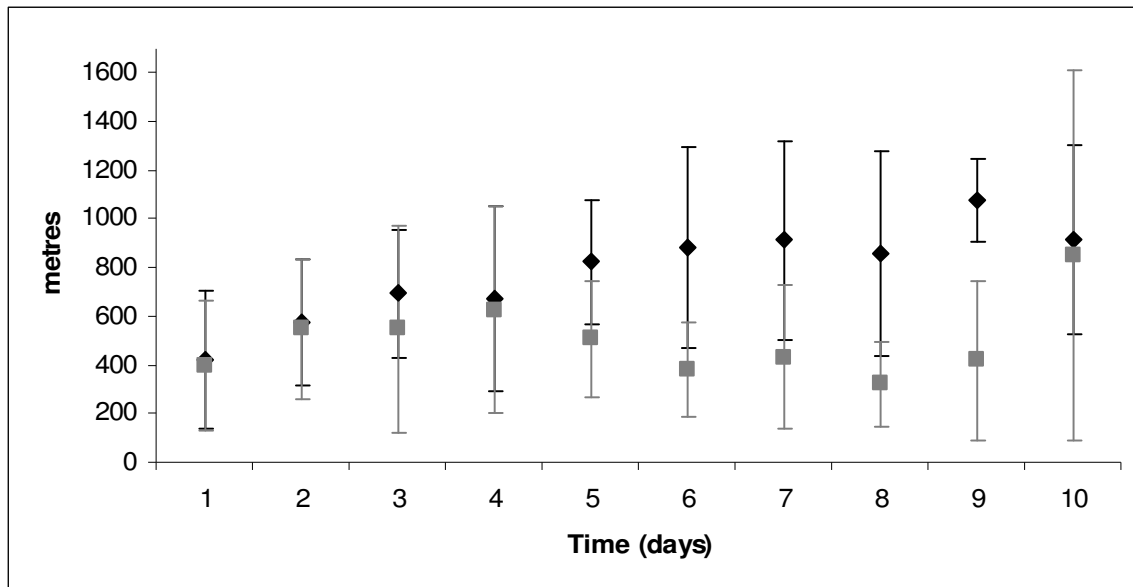


Figure 4.1: Maximum distance away from the release point each day (\pm sd). \blacklozenge males, \blacksquare females. Sample sizes vary and reflect the various analyses. Days 1-5: $n_m = 11$, $n_f = 9$. Days 6-8: $n_m = 8$, $n_f = 4$. Days 9 & 10: $n_m = 5$, $n_f = 2$.

Differences between first and second phase birds

Bristlebirds in the second phase of releases at the two release sites in the third year were significantly further from the release point over the first 8 days, averaging 303 ± 97 , than the birds released initially in the two sites in the first two years without conspecifics ($F_{1,10} = 5.248$, $P = 0.045$) (Fig 4.2 and 4.3).

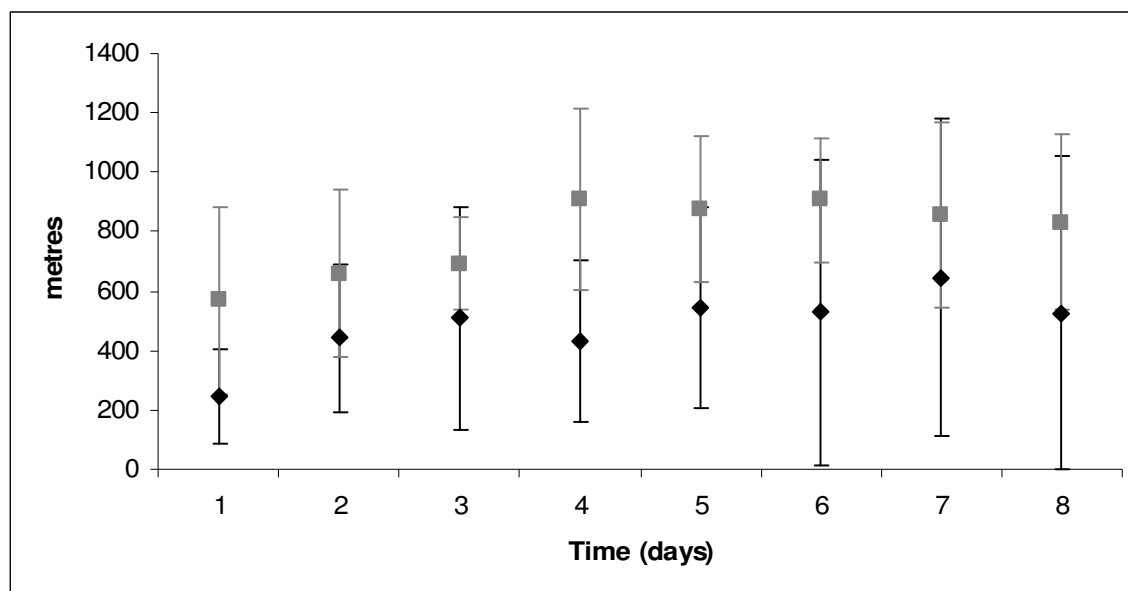


Figure 4.2: Maximum distance away from the release point each day (\pm sd). \blacklozenge 1st phase birds, \blacksquare 2nd phase birds. Sample sizes: $n_{1st} = 6$, $n_{2nd} = 6$.

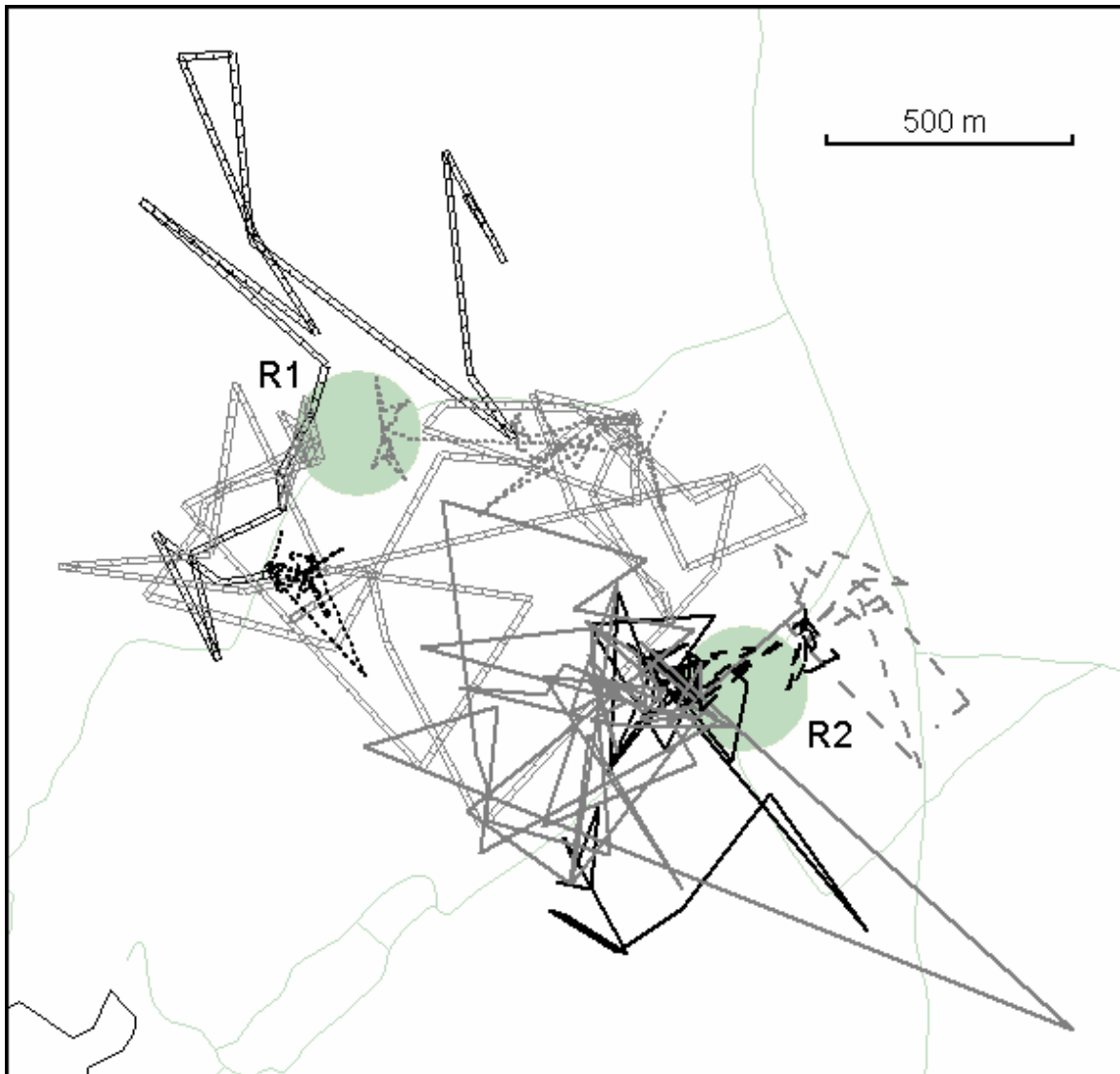


Figure 4.3: A sample of tracked Eastern Bristlebirds over the first four days. Solid and striped lines represent males (4), dashed lines represent females (4). Black represents 1st phase birds (4), grey represents 2nd phase birds (4). Both release points are identified.

Maximum distance moved in a day

All bristlebirds

During the first nine days after release, the maximum distance a bristlebird moved away from its last position the day before increased from 177 ± 107 (sd) to 529 ± 299 m ,

although this change was not significant ($F_{2.5, 9.9} = 3.416$, $P = 0.067$, df calculated with Huynh-Feldt correction) (Fig 4.4).

Differences between sexes

Small sample sizes ($n = 5$) precluded an analysis between the sexes over nine days. For the first five days males increased their maximum distance moved in a day from 125 ± 52 to 317 ± 182 m and females remained stable between 228 ± 92 and 210 ± 161 m (Fig 4.4). However, this difference between males and females was not significant ($F_{2.9, 31.5} = 2.409$, $P = 0.088$).

Differences between first and second phase birds

When the first versus second phase releases were compared over the first five days, second phase birds increased the maximum distance they moved in a day from 194 ± 104 to 346 ± 167 m (Fig 4.5). This was significantly different to first phase birds which decreased the maximum distance they moved from 159 ± 69 to 120 ± 34 m over the same period ($F_{1, 11} = 6.237$, $P = 0.03$).

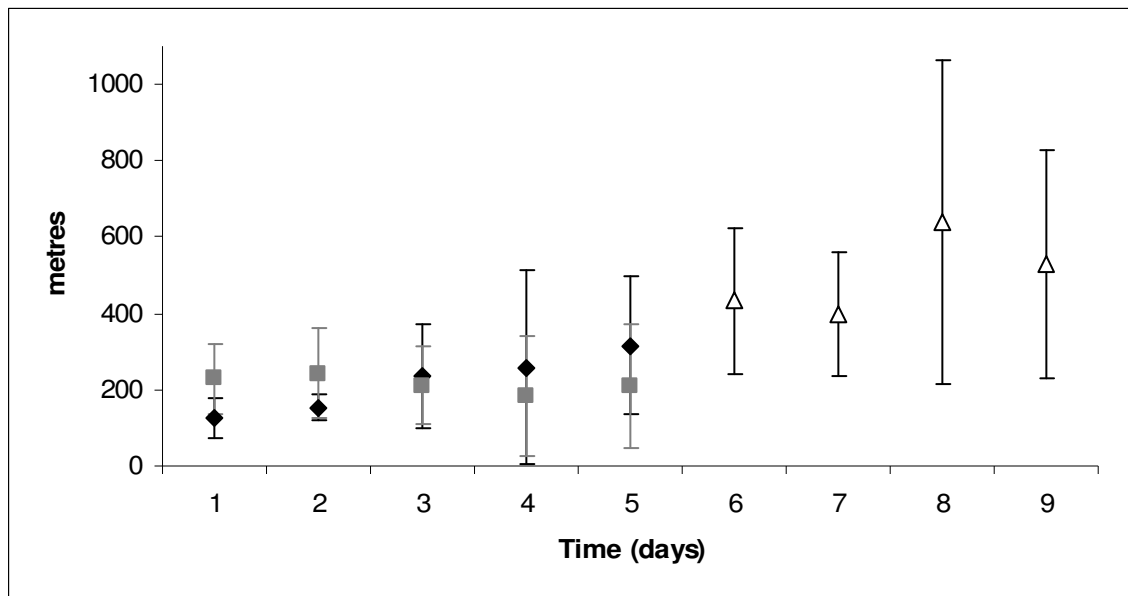


Figure 4.4: Maximum distance moved in a day (\pm sd). \blacklozenge males, \blacksquare females, \triangle both sexes. Sample sizes vary and reflect the various analyses. Days 1-5: $n_m = 6$, $n_f = 7$. Days 6-9: $n = 5$.

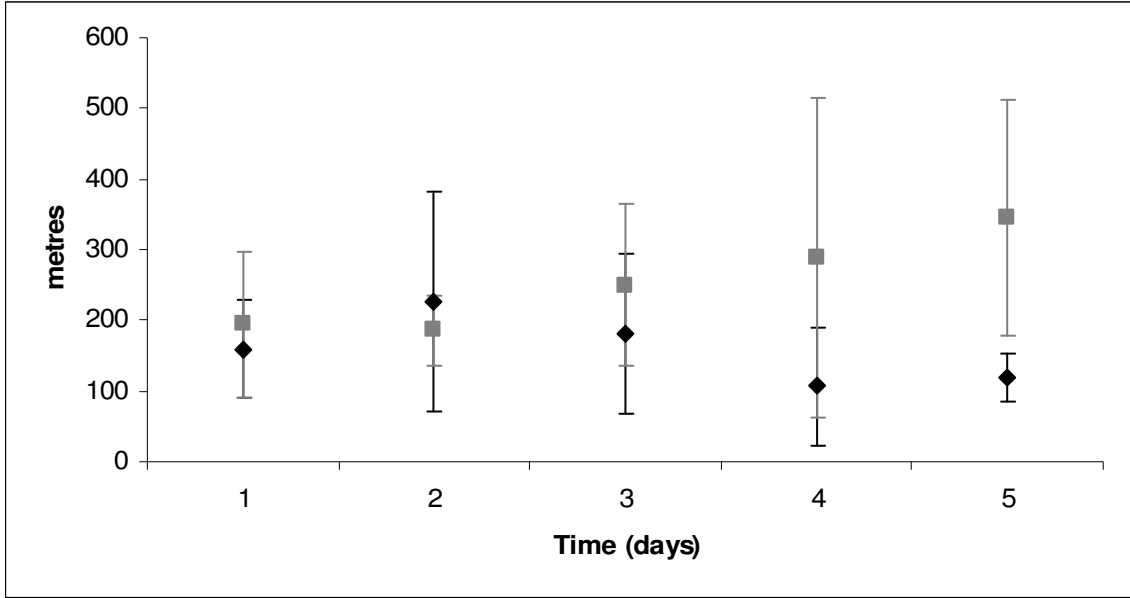


Figure 4.5: Maximum distance moved in a day (\pm sd). \blacklozenge 1st phase birds, \blacksquare 2nd phase birds. Sample sizes: $n_{1st} = 5$, $n_{2nd} = 8$.

Home range over four day periods

All bristlebirds

Bristlebirds tended to increase the MCP area of their home range over the first three, four-day periods (first 12 days) from 16 ± 15 (sd) to 37 ± 39 ha. However, this influence of time was not significant ($F_{1,4, 7.2} = 4.460$, $P = 0.06$).

Differences between sexes

Males tended to have larger MCP home ranges than females, with males attaining 19 ± 23 ha in the first four days to 61 ± 39 ha in days nine to twelve and females from 7 ± 7 ha to 19 ± 32 ha during the same period (Fig 4.6 and 4.3). The difference between males and females was not significant across the whole 12 day period ($F_{1,5} = 3.865$, $P = 0.106$), but was significant for the first two four-day periods (eight days) which increases sample sizes ($F_{1,16} = 6.144$, $P = 0.025$) (Fig 4.6).

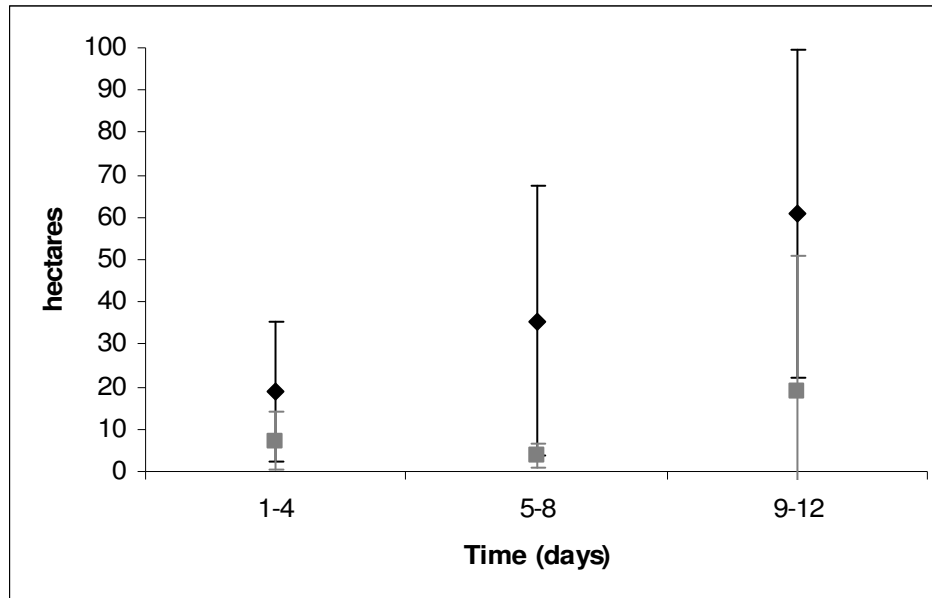


Figure 4.6: Home range over four day periods (\pm sd). \blacklozenge males, \blacksquare females. Home range calculated as Minimum Convex Polygon. Sample sizes vary and reflect the various analyses. Days 1-4 & 5-8: $n_m = 13$, $n_f = 5$. Days 9-12: $n_m = 3$, $n_f = 4$.

Differences between first and second phase birds

There was no overall significant difference in the area explored by first versus second phase birds in the first two four day periods (eight days), although there was a significant interaction between release phase and time since release ($F_{1, 16} = 5.811$, $P = 0.028$).

During the first four day period bristlebirds from both release phases explored similar areas 16 ± 15 ha and 15 ± 18 ha but over the next four day period second phase birds had home ranges of 42 ± 40 ha, much larger than the first phase releases of 19 ± 23 ha (Fig 4.7 and 4.3).

The 50% UD area was only calculated for the first four day period as only enough data points were collected during this time for its calculation. There was no difference between males and females over this period ($t_{29} = 0.54$, $P = 0.6$), males having a 50% UD of 4 ± 3 ha and females of 5 ± 6 ha. There was also no difference between first and second phase releases with 50% UD of 5 ± 5 ha and 3 ± 2 ha respectively ($t_{29} = 0.77$, $P = 0.45$).

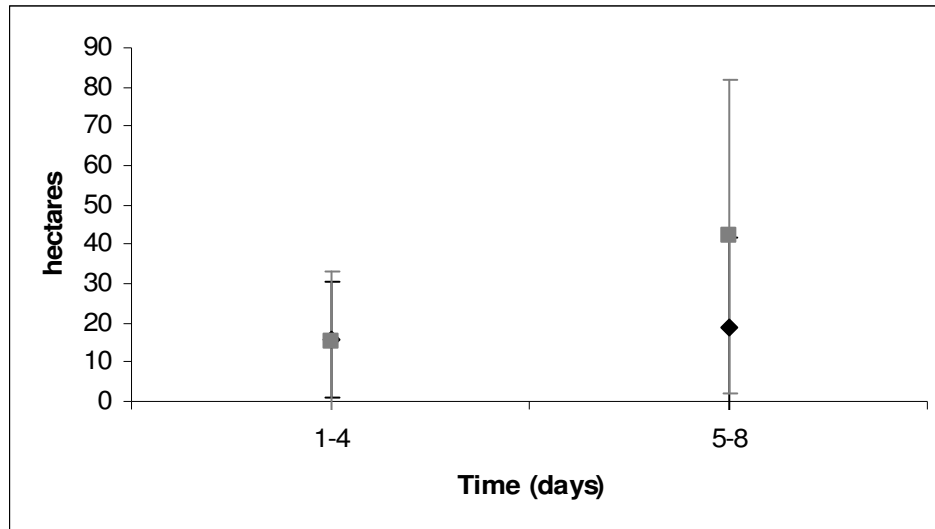


Figure 4.7: Home range over four day periods (\pm sd). \blacklozenge 1st phase birds, \blacksquare 2nd phase birds. Home range calculated as Minimum Convex Polygon. Sample sizes: $n_{1st} = 12$, $n_{2nd} = 6$.

Distance between positions on consecutive mornings

All bristlebirds

From the morning of the second day to the morning of the ninth day, there was a significant increase in the average distance between bristlebird positions on consecutive mornings ($F_{6, 24} = 2.583$, $P = 0.045$) from 199 ± 133 (sd) to 676 ± 496 m (Fig 4.8). However, analysing the data from the morning of the second day to the morning of the fifth day revealed no significant changes over time (Fig 4.8).

Differences between sexes

There was no difference in the distance between positions on consecutive mornings of males compared to females over the first three morning periods ($F_{1, 10} = 0.464$, $P = 0.511$). Males ranged from 190 ± 120 to 405 ± 384 m and females from 264 ± 238 to 146 ± 130 m.

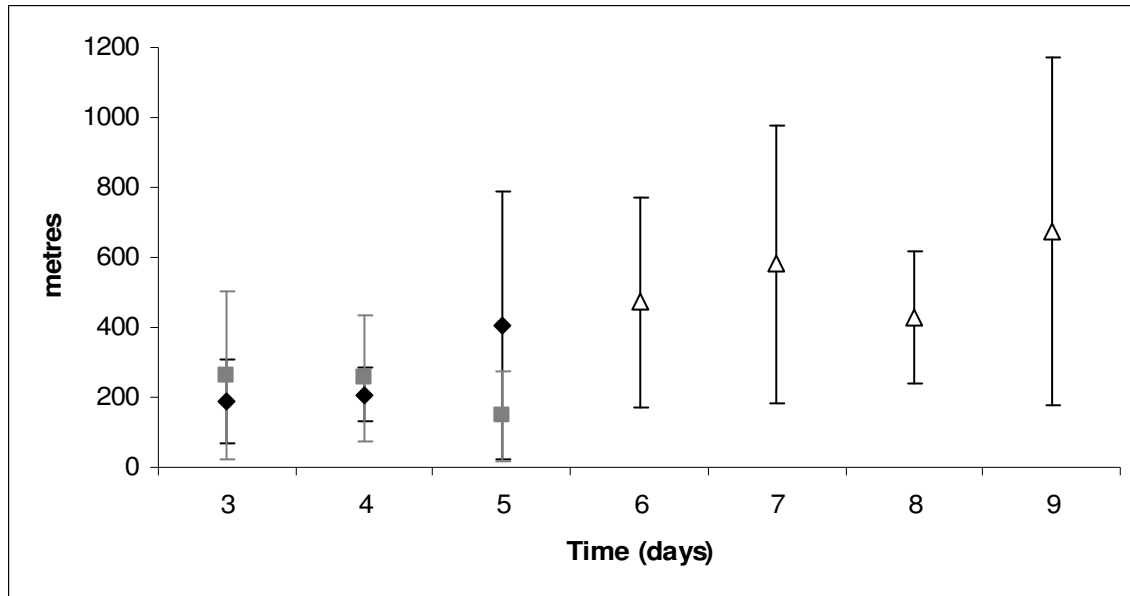


Figure 4.8: Distance between consecutive mornings positions (\pm sd). \blacklozenge males, \blacksquare females, \triangle all birds. Sample sizes vary and reflect the various analyses. Days 3-5: $n_m = 6$, $n_f = 6$. Days 6-9: $n = 5$.

Differences between first and second phase birds

First phase bristlebirds decreased the distance between positions on consecutive mornings from 264 ± 226 to 229 ± 191 m over the first three morning periods while second phase birds increased from 182 ± 139 to 331 ± 337 m. However, there was no significant difference between first and second phase birds ($F_{2, 20} = 2.854$, $P = 0.081$).

Average daily distance moved per hour

All bristlebirds

During the first five days there was no significant effect of time on the hourly distances moved by the released bristlebirds ($F_{4, 40} = 0.613$, $P = 0.656$). Bristlebirds moved an average of 136 ± 78 (sd) m/hr and ranged between 12 and 471 m/hr.

Differences between sexes

There was a significant interaction between time since release and sex ($F_{4, 40} = 5.195$, $P = 0.002$) caused mainly by a significant divergence in the male and female trajectories between day 3 and day 4 ($F_{1, 10} = 51.193$, $P = 0.019$) (Fig 4.9). There was no overall significant difference between the sexes ($F_{1, 10} = 0.206$, $P = 0.66$).

Differences between first and second phase birds

There were no differences in the hourly movements of first phase bristlebirds compared to second phase birds. First phase birds averaged 136 ± 85 m/hr over the first 5 days and second phase birds averaged 137 ± 63 m/hr.

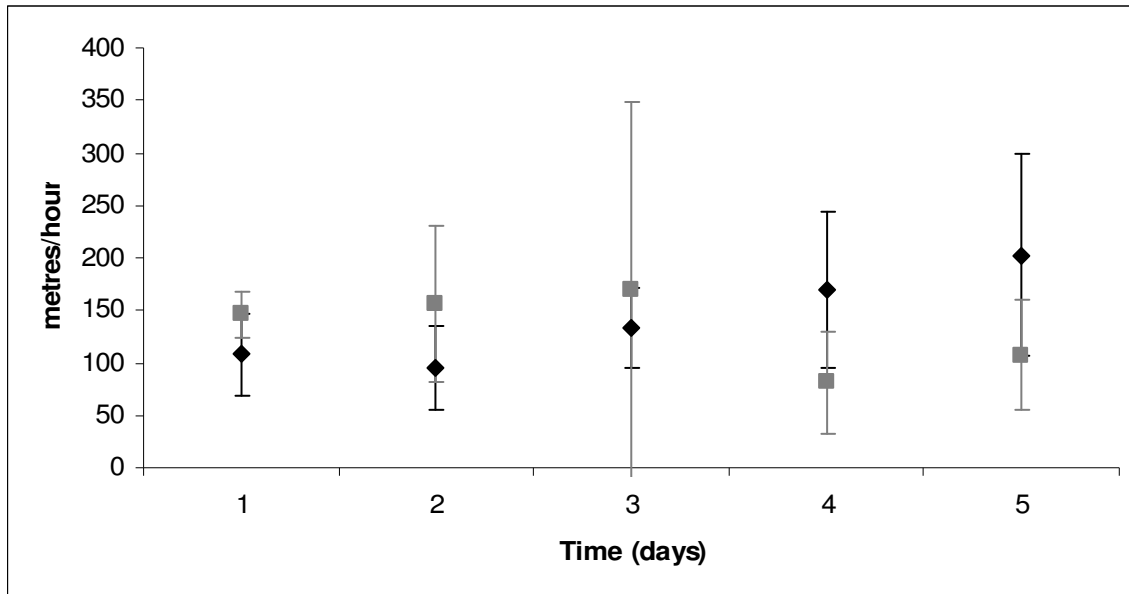


Figure 4.9: Average daily distance moved per hour (\pm sd). \blacklozenge males, \blacksquare females. Sample sizes: $n_m = 8$, $n_f = 4$.

Radio-tracking error

Radio-tracking error was positively related to distance from the receiver ($F_{1, 32} = 21.198$, $P < 0.001$), although it was variable ($r^2 = 0.4$) (Fig 4.9). Tracking error averaged 100 ± 81 m and ranged from 14 to 373 m. However, if four outliers (Fig 4.10) are removed from

the 34 error measurements, the new average becomes 82 ± 58 m and ranges from 14 to 231 m.

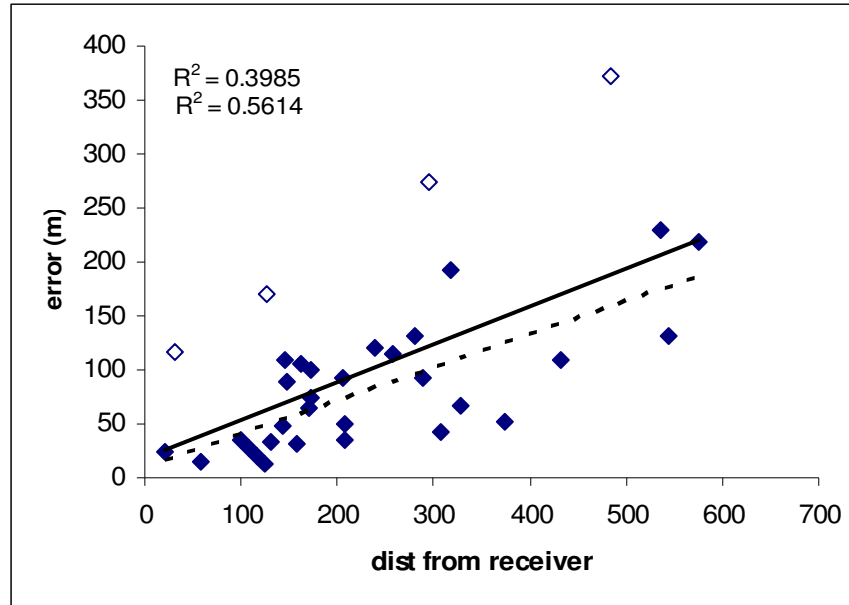


Figure 4.10: Regression of error distance against the distance between transmitter and receiver. \diamond represents outliers. Two regression lines, solid line for \diamond and \blacklozenge , dashed line for just \blacklozenge .

Discussion

It was expected that translocated bristlebirds would move less than bristlebirds in their native environment, due to the availability of vacant habitat. However, home ranges were larger in translocated bristlebirds than those in their native habitat. Productivity of the habitat may affect the home range size and behaviour of translocated individuals (van der Ree *et al.* 2001), although Gibson (1999) found a similar abundance of dietary items in the release and source environments used in this study. Interestingly, there was little difference between the rate of movement between translocated and non-translocated individuals. The average hourly movements of bristlebirds were similar between

translocated and non-translocated individuals. Both translocated and non-translocated bristlebirds were moving at a similar rate but covering very different amounts of habitat. Bright and Morris (1994) examined the dispersal of Dormice (*Muscardinus avellanarius*) for 10 days after translocation and found that hard-released individuals moved further and for longer periods per night than those that were resident in a site with similar characteristics. Presumably non-translocated bristlebirds know where resources are within their home range and their movements are based around those resources. With translocated individuals, they may make more random exploratory movements around the habitat, presumably to find resources.

It is unclear whether bristlebirds had begun to settle in the host environment during the first two weeks. Home ranges were large compared to bristlebirds in native habitat, although this does not necessarily mean bristlebirds were still dispersing. Home ranges of these translocated bristlebirds seem to be increasing over time, although this increase was not statistically significant. The location of bristlebirds between consecutive mornings got progressively further apart over the first nine days and they also moved away from the release point, increasing this distance over the first five or six days after release (Fig 4.1). These results suggest that bristlebirds had not begun to settle within the first two weeks after translocation but were staying in the vicinity of the release locations (Fig 4.3). Translocated Hihi in New Zealand (Castro *et al.* 1994), Noisy Scrub-birds (Danks 1994) and Southern Emu-wrens (Pickett 2003) in Australia were all settled in their release habitat within three months, longer than the tracking period in this study.

Male bristlebirds were consistently further from the release point than females, had larger home ranges in the first eight days and females seemed to stop moving away from the release point before males. This is similar to many results from mammal translocations (Davis 1983; Short and Turner 2000) with males dispersing more than females, but bird translocations seem to be very variable in responses where reported (Castro *et al.* 1994; Armstrong and Craig 1995). Discussions on male and female dispersal have generally postulated male-biased dispersal in mammals, where males disperse further than females, and female-biased dispersal in birds (Greenwood 1980; Wolff and Plissner 1998; Dale

2001). It is expected that females would disperse more if males defend territories or conversely that males will disperse more if females settle quickly to avoid high costs associated with dispersal and mate searching. The rate at which the translocated bristlebirds were travelling and dispersing showed no differences between males and females. The average movement per hour and initially the distance between consecutive morning positions was the same for both males and females. By five days after release, females were beginning to reduce their distance travelled between consecutive morning points compared to males. Therefore, it seems that males were exploring a larger area and in a general direction taking them away from the release point, whereas females explored smaller areas closer to the release point. This was all done while males and females were moving at a similar rate. These results suggest that females may be undertaking a more intensive exploration of areas whereas males are undertaking a more extensive exploration of the area (Fig 4.3).

Alternatively, males or females may have been searching for the presence of the opposite sex. Brush-tailed Phascogale males dispersed significantly less when released into areas with established female home-ranges (Soderquist 1994). In some translocations of Noisy Scrub-birds in Western Australia, to test the suitability of the habitat, males were released up to a year before females. A year after females were released, there was evidence of breeding which suggested that females had dispersed to within the vicinity of the males (Danks 1994). In line with predictions, males may be exploring widely to secure a high-quality home range, as proposed in Greenwood's (1980) resource defence theory, whereas females may be responding to male cues in the environment and settling quicker, reducing resource and predation costs associated with dispersal and mate searching (Dale 2001).

Expectations about the dispersal of first and second phase birds were based on whether vacant habitat or conspecific cues were the main stimulus for settlement. Second phase bristlebirds moved further from the release point and moved greater distances each day than first phase birds. This accords with the general observation that translocated individuals released into an area of habitat, as a second phase of a translocation, disperse

further and faster than those released initially. When translocating Dormice, Bright and Morris (1994) conducted two releases two months apart. Those released in the second stage moved significantly further in a night than those released two months earlier. Castro *et al.* (1994) found that New Zealand Hihi delayed in an aviary before release dispersed further and faster than those released immediately after transportation. This might suggest that vacant habitat can stimulate dispersal and settlement.

Notably, second phase bristlebirds stopped travelling away from the release point in much the same time as first phase bristlebirds, only they had travelled further. If the rate at which the second phase bristlebirds moved was faster than the first phase bristlebirds then it might be concluded that these birds were actively pushed away by those released initially, even though bristlebirds are known to have highly overlapping home ranges (Baker 2001). There was no difference in the rate of movement between first and second phase bristlebirds, suggesting that more recently translocated bristlebirds were probably not being actively pushed away. It has been suggested that resident conspecifics may provide the cues to good quality habitat and may attract newly arrived individuals and stimulate them to settle (Smith and Peacock 1990; Stamps 1991). The presence of the first phase bristlebirds may have stimulated settlement in the more recently translocated bristlebirds once they had traversed to vacant habitat beyond the home ranges of first phase bristlebirds. Second phase bristlebirds were consistently between 200 and 400 metres away from where the first phase bristlebirds were last recorded (Fig 4.2). This distance approximates the diameter of a large (circular) home range (Baker 2001). Support for this conspecific attraction has been observed with translocated Hihi moving from vacant habitat at the release site to areas near resident conspecifics within three days of release (Castro *et al.* 1994).

There was one predation event recorded during the radio-tracking period, where a bristlebird was taken by a bird of prey. This is in stark contrast to most translocations of Australian mammals (Short *et al.* 1992; Fischer and Lindenmayer 2000) which were predominately killed by introduced predators. Translocations of birds in Australia have not suffered from significant predation (Danks 1998; Fischer and Lindenmayer 2000;

Soorae and Seddon (eds) 2000; Clarke *et al.* 2002), with the exception of the Malleefowl (*Leipoa ocellata*) (Priddel and Wheeler 1997). Translocated individuals perhaps have a higher risk of predation from the increased stresses associated with capture, transportation and an unfamiliar environment.

Significant error was measured from the transmitters. Error was calculated after transmitters were assumed to have fallen off the bristlebirds, suggested by failing to detect any movement from a transmitter for more than a day and a diminished detectability of the signal. It is thought that this method of estimating error caused an overestimation due to the diminished transmitter signal strength once transmitters had fallen off the birds. The signal was thought to have diminished as transmitters were almost always found on the ground in the leaf litter. It is thought that this position created more interference to the signal in this dense vegetation and hence it was harder to get an accurate bearing. Bristlebirds could often be detected moving whilst bearings were being taken which gave confidence in the accuracy of the radio-tracking procedure.

Despite the error, translocated bristlebirds still moved over much larger areas than those in their native home ranges. This suggests that birds in novel habitats may be predominately searching more widely for resources rather than being cautious and dispersing little in the new environment. The requirements for food and shelter may be more important than the perceived risks associated with a novel environment. Male bristlebirds dispersed more than females (Fig 4.1 and 4.3), opposing the general trends found for bird dispersal (Clarke *et al.* 1997). This may be a combination of males searching for high-quality home ranges (Greenwood 1980) and females limiting their dispersal due to the costs associated in mate searching (Dale 2001). Females possibly stop dispersing when they find males or male cues in the environment. Conspecific attraction was important in reducing dispersal and aiding settlement. First phase bristlebirds did not disperse widely, as was predicted for birds released into vacant habitat. Second phase bristlebirds also dispersed very little, just beyond the first phase dispersal (Fig 4.2 and 4.3). It is suggested that a combination of conspecific attraction and the presence of vacant habitat drives dispersal and settlement in translocated birds.

There may be benefits in staging translocations if individuals released initially help to stimulate the settlement of individuals released in later stages. Initially, a trial release can be made where the loss of individuals may not be too significant, but enough individuals are released for survival until the next release stage. If the habitat seems suitable, more releases can be made to augment the translocated population with the understanding that the initial birds may stimulate settlement in the new releases. Danks (1994) used a variation of this technique with Noisy Scrub-birds. In some translocations males were released a year before females to test the habitat. Males were used as they are highly territorial and they are easier to catch, they were also thought of as more disposable than females. Presumably these males will also help to stimulate settlement in females. Caution should be used with this technique unless there is a significant understanding of the breeding system of the species.

Translocated bristlebirds travel further than bristlebirds in their native habitat, however, their dispersal was much less than juvenile dispersal recorded for the closely related Rufous Bristlebird (Seymour *et al.* 2003). To date, juvenile dispersal has not been studied in the Eastern Bristlebird. This is important to understand when planning release sites for translocated birds as it may help to answer the following questions: How much habitat is available in the release site and how isolated is the release site from other known populations? Some future research should be directed at determining whether there is a relationship between the amount of available habitat at the release site and the extent of exploration in translocated individuals.

There are also implications for understanding the extent of dispersal in the management of current populations of bristlebirds. There are currently a number of isolated pockets of bristlebirds, disjunct from main populations and in some instances with very little connecting habitat (Bain and McPhee 2005). The conservation of these populations of bristlebirds may be difficult. If these populations are lost or further fragmented then individuals cannot be expected to colonise these isolated areas of habitat away from the presence of other bristlebirds. Dispersing birds may be lost to the population as these

fragments of habitat may become sinks. This investigation of bristlebird dispersal suggests that any fragmentation of habitat is likely to cause a decline in species where the presence of conspecifics influences dispersal.

Impact of removals on an Eastern Bristlebird population

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This chapter has been written as a manuscript and as such it will contain some repetition in the introductory section from Chapter 1 and some overlapping in the methods section with other data chapters. All species are reintroduced.

Chapter 5

Impact of removals on an Eastern Bristlebird population

Introduction

The removal of individuals from a population may occur for a number of reasons including predation, environmental stochasticity, catastrophe and anthropogenic activity. The impact on a population will depend on the magnitude and nature of the removal, and on the life history of the species. An increased loss of individuals from a population would be expected to decrease local population densities and may increase local extinction rates. As such, an understanding of the influence of a loss of individuals from a population has importance to conservation and understanding population dynamics. For animals that maintain territories, often the loss of individuals does not change local population densities. In a removal experiment off the coast of Wales Harris (1970) found that territorial behaviour, and not an equilibrium between mortality and juvenile recruitment, was maintaining a population of Oystercatchers (*Haematopus ostralegus*) at between 49 and 52 breeding pairs for over 5 years. In a similar removal experiment on Crested Tit (*Parus cristatus*) and Willow Tit (*Parus montanus*) by, six months after removals the population had returned to a density matching that in a control population (Cederholm and Ekman 1976). The suggestion was that territorial behaviour was the main factor determining the number of birds in both populations, which is backed up by other studies (Wesolowski 1981; Sherry and Holmes 1989; Monkkonen 1990).

A range of studies have improved our understanding of the response of bird species when a territory is vacated. Of 59 removal studies in territorial species reviewed by Newton (1992), 46 % reported over 75 % replacement of removals, of which 17 % reported

greater than 100 % replacement. Only 12 % of studies reported no replacement. The replacements were often said to be coming from a pool of non-territory holders or 'floaters' (Newton 1992). Floaters live in the same areas as territory holders, they just move around with much larger home ranges than the territory holders (Pedersen 1988) and may be sexually active or not (Newton 1992). Direct evidence for the presence of floaters is lacking, although many studies claim their existence (Pedersen 1988; Sherry and Holmes 1989; Monkkonen 1990). Most projects suffer from restrictions in experimental design such as the unknown origin of replacements or a failure to monitor the future of floaters replacing the removed territory holders (Newton and Marquiss 1991; Newton 1992). When they have been identified and followed, floaters were mainly comprised of both young and old males (Sherry and Holmes 1989; Newton and Marquiss 1991; Newton 1992; Sternberg *et al.* 2002). Therefore, conclusions need to be careful about territoriality limiting breeding numbers in populations which only have a male surplus, as it may be due to a limitation of breeding females and not territorial behaviour. Thompson (1977) found that there were many more male Yellow-breasted Chats (*Icteria virens*) than were breeding and when breeding males were removed, non-breeding males paired with females. It was the availability of females and not territoriality that determined the number of breeding pairs.

Territoriality necessarily influences the distribution of birds within a population and vacation of territories can influence sizes and numbers of territories in an area. Knapton and Krebs (1974) found with Song Sparrows (*Melospiza melodia*) that if all territory holders were removed at once, the replacement territories were smaller due to more territories being established in the same area at the same time. However, if territory holders were removed one at a time, the replacement territories were the same size as others before the removal. In a separate experiment on Willow Warblers (*Phylloscopus trochilus*), the removal of individuals caused an influx of new arrivals along with the expansion of territories by some territory holders that were not removed (Arvidsson and Klaesson 1984).

The study of removals and the response of populations to removal is important to understanding limiting factors in a population including territoriality, recruitment and resources (Sherry and Holmes 1989). Furthermore, the translocation of individuals to establish or re-establish populations is a common conservation strategy, which is growing in popularity (Griffith *et al.* 1989; Fischer and Lindenmayer 2000). Part of many of these projects involves the removal of individuals from wild populations. Understanding the impact of the removal on the source population is important for assessing how viable translocation from the wild is for the conservation of a species.

In translocation projects using wild caught individuals as founders, the monitoring of source populations is often neglected, or not reported. In a recent examination of 23 translocations of wild animals published since 1983 (see Table 1.1, chapter 1), only four of these discuss any monitoring of the source population. This paucity of monitoring of wild source populations during translocation projects, or a lack of reporting of results is alarming, as most wild sourced translocations involve threatened species whose wild populations are often very small. There is both an obligation and necessity to monitor the impact and recovery in a source population (McCarthy 1994), as original wild stock should always have the highest priority (Kleiman *et al.* 1994).

As part of recent conservation efforts, the Eastern Bristlebird (*Dasyornis brachypterus*) was translocated at Jervis Bay, NSW. This translocation was a reintroduction of the species into part of its former range. Founding individuals for the translocation were sourced from a wild population over three years. With concern for the source population from which bristlebirds were removed and in an effort to understand more about the species, a monitoring study was established in the source population. This study provided an opportunity to investigate the impact of a sustained removal of individuals from a bristlebird population. This was used to test the hypothesis that removals will cause a drop in density and cause a change in the distribution of bristlebirds across the habitat. The aims were: (1) To monitor bristlebird numbers in the source population. (2) To investigate changes in the distribution of bristlebirds in response to the removals. (3) To investigate changes in the population structure as a result of the removals.

Methods

This study was conducted at Bherwerre Peninsula, Jervis Bay (150°45', 35°04') on the south coast of NSW, where the translocation program involved the removal of bristlebirds from a site over three years.

Two sites in the source population were investigated, a removal site and a control site. The design initially incorporated 2 removal and 2 control sites although a large wildfire prevented this replication. At the remaining removal site, bristlebirds were caught and removed using mist nets and call playback along service trails. The source population encompasses approximately 4 300 ha, with the removal and control sites 2 km apart and encompassing approximately 500 ha and 400 ha of habitat respectively. There were three removal periods, 2003, 2004 and 2005, removing 16, 20 and 8 birds respectively. Bristlebird activity peaks around the breeding season (September to February). Removals were planned approximately two months after the breeding season to maintain reasonable capture success while trying to avoid removing bristlebirds caring for juveniles.

The removal and control sites were surveyed annually in October from 2002 - 2005. Two additional surveys were also conducted during January in 2004 and 2005. Survey methods followed closely those of Baker (1997) and were conducted along two transects, 4620 m long in the removal site and 4500 m long in the control site. Surveys consisted of walking at 2 – 4 km/h, mapping the number and positions of bristlebirds seen or heard. It is well established that bristlebirds can be reliably mapped by a competent observer (Bain and McPhee 2005). Surveys were repeated at the same time of the year to try to avoid any seasonal changes in detection probability of the birds (MacKenzie and Kendall 2002).

Surveys were repeated on four separate mornings within four weeks. The survey recording the maximum number of bristlebirds was used for data analysis. The transect in the removal site was divided into discrete sections of 500 m, a minimum of 480 m apart. Due to the transect arrangement, the control site was only divided into two sections, 1865

m and 1600 m long, that were 500 m apart. Each of these sections were considered replicates within each of the sites. It was assumed that these sections were independent as it is expected that bristlebirds would not move over 500 m in the 15 to 30 minutes taken for an observer to traverse that distance. The maximum distance bristlebirds have been recorded moving in one hour is 330 m (Baker 2001). Repeated measures analysis of variance was used to examine changes in bristlebird numbers before and after the removals.

To investigate changes in the size of territories, nearest-neighbour data were recorded, with bristlebirds found as duetting pairs (calling to one another) considered as one bird for analysis. Repeated measures analysis of variance was used to investigate any differences in the average nearest-neighbour distances from year to year. The distributions of nearest-neighbour distances were fitted to a Poisson distribution, to examine if they were distributed randomly (Zar 1984). The goodness of fit of the Poisson distribution was examined using Kolmogorov-Smirnov one-sided exact tests. Kolmogorov-Smirnov two-sample tests were then used to compare the distribution of nearest-neighbour distances from bristlebirds between control and removal sites and before and after the removals.

To further understand the effect of the removal of birds on the population, the sex ratio and morphometrics of removed birds were measured over the three years. The sex ratio of bristlebirds caught over the three years was examined with a chi-square test. The expected ratio was obtained using a contingency table. Morphometrics of removed bristlebirds were analysed across the three years using analysis of variance.

Results

Numbers

Before any removals there were 2.6 ± 2.4 (sd) bristlebirds/500 m in the removal site and 3 ± 1.2 bristlebirds/500 m after three years of removals (Fig 5.1). No impact was found from the removals on the number of bristlebirds in either of the sites over the length of the study ($F_{5, 20} = 0.812$, $P = 0.555$). The removal and control site varied very little from each other and no significant differences were found between them ($F_{1, 4} = 0.35$, $P = 0.861$). However, between Oct 2003 and Jan 2004 there was a significant increase of 1.4 bristlebirds/500 m in both the control and removal sites ($F_{1, 4} = 11.391$, $P = 0.028$).

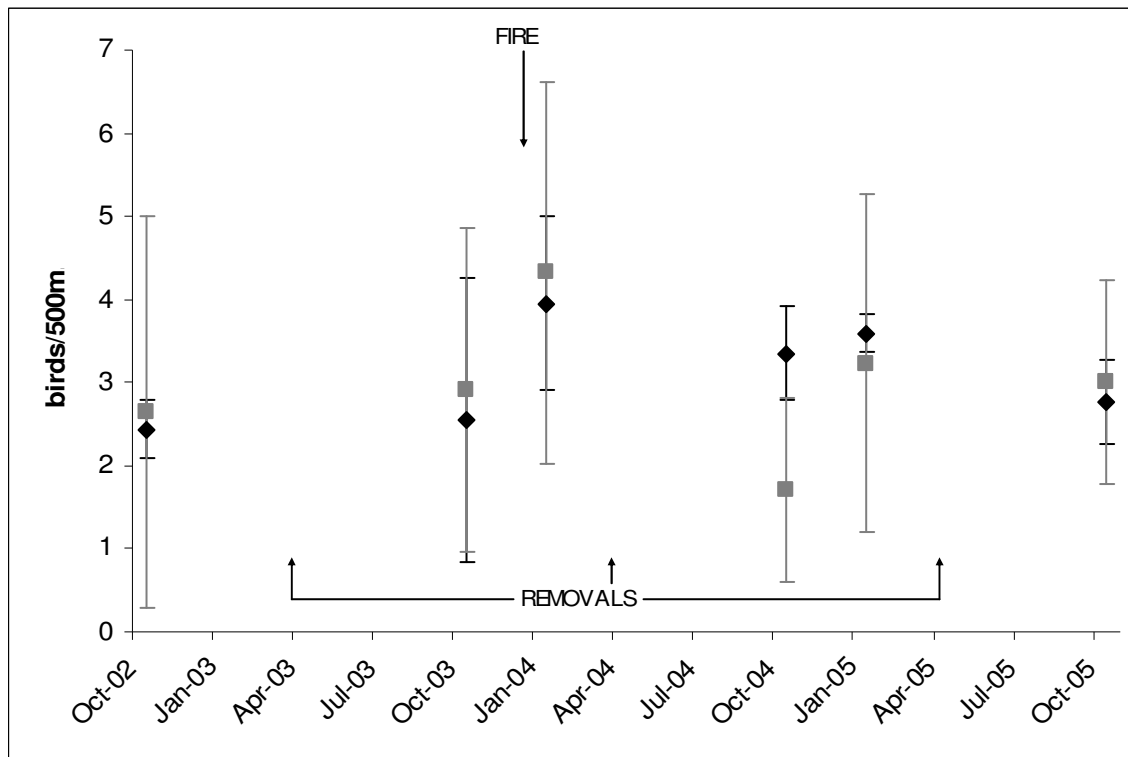


Figure 5.1: Average number (\pm sd) of Eastern Bristlebirds surveyed per 500 metres over the three years of the study. ■ removal site, ◆ control site. Time of removals and fire are indicated.

Average nearest-neighbour distances

Nearest-neighbour distances in the removal site increased from 223 ± 80 (sd) to 236 ± 97 m and the control site decreased from 184 ± 73 to 161 ± 56 m over the three years of removals (Fig 5.2). However, these changes over time were not statistically significant ($F_{3,90} = 2.225$, $P = 0.091$). The interaction between time and site was not significant either ($F_{3,90} = 2.212$, $P = 0.092$). The overall difference between the control and removal sites was nearly significantly different ($F_{1,30} = 4.049$, $P = 0.053$). There was a significant change over time in average nearest-neighbour distances from 2002 to 2003 ($F_{1,30} = 5.071$, $P = 0.032$) with the removal site decreasing by 60 m and the control site by 8 m (Fig 5.2). However, the interaction between time and site suggests that there were no differences between the sites during this period ($F_{1,30} = 2.736$, $P = 0.109$). From 2003 to 2004 there was another significant change through time ($F_{1,30} = 4.829$, $P = 0.036$) with the removal site increasing by 55 m and the control site by 6 m. Again, the interaction between time and site shows the difference between the sites was not significant during this period ($F_{1,30} = 3.856$, $P = 0.059$).

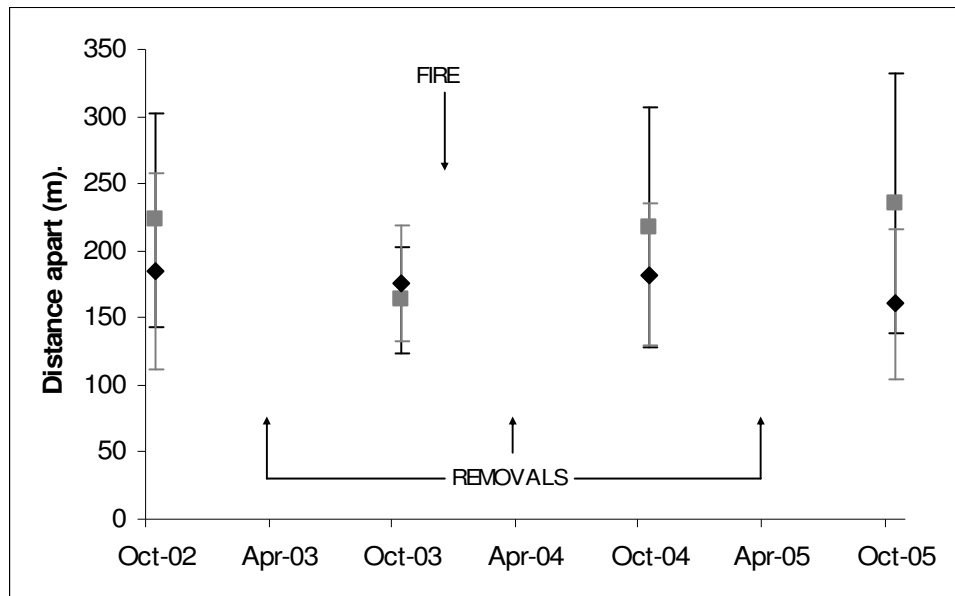


Figure 5.2: Average nearest-neighbour distances (sd) of Eastern Bristlebirds.

■ removal site, ♦ control site.

Distribution of nearest-neighbour distances

At all times in both of the sites, the distribution of nearest-neighbour distances was significantly different from a Poisson distribution (Fig 5.3, Table 5.1) suggesting that they were not randomly distributed. Examination of Fig 5.3 suggests that the distributions were close to uniform, except in 2004 and 2005 in the removal site where the distribution has become bimodal. To examine if the distribution changed as a result of the removals, the distributions of nearest-neighbour distances were compared against each other for change over time or between sites. There was a significant difference between years 2002 and 2003 ($P = 0.01$, Table 5.2), and between 2003 and 2005 ($P = 0.01$, Table 5.2) in the removal site. There were no other differences between time periods or between sites (Table 5.2). The results show that in the removal site in 2003, bristlebirds were closer together than pre-removal in 2002 or after all three removals in 2005 (Fig 5.3).

Table 5.1: Goodness of fit of Poisson distribution. Lambda represents the average. D is the Kolmogorov-Smirnov one-sided test statistic. Significance values are also given. Removals occurred between each of the years.

<i>Site and year</i>	<i>Lambda (average)</i>	<i>D</i>	<i>P value</i>
Removal 2002	223	0.446	<0.0001
Removal 2003	163	0.342	0.0056
Removal 2004	218	0.545	<0.0001
Removal 2005	236	0.409	0.0005
Control 2002	184	0.466	<0.0001
Control 2003	176	0.357	0.0036
Control 2004	182	0.451	<0.0001
Control 2005	161	0.4	0.0007

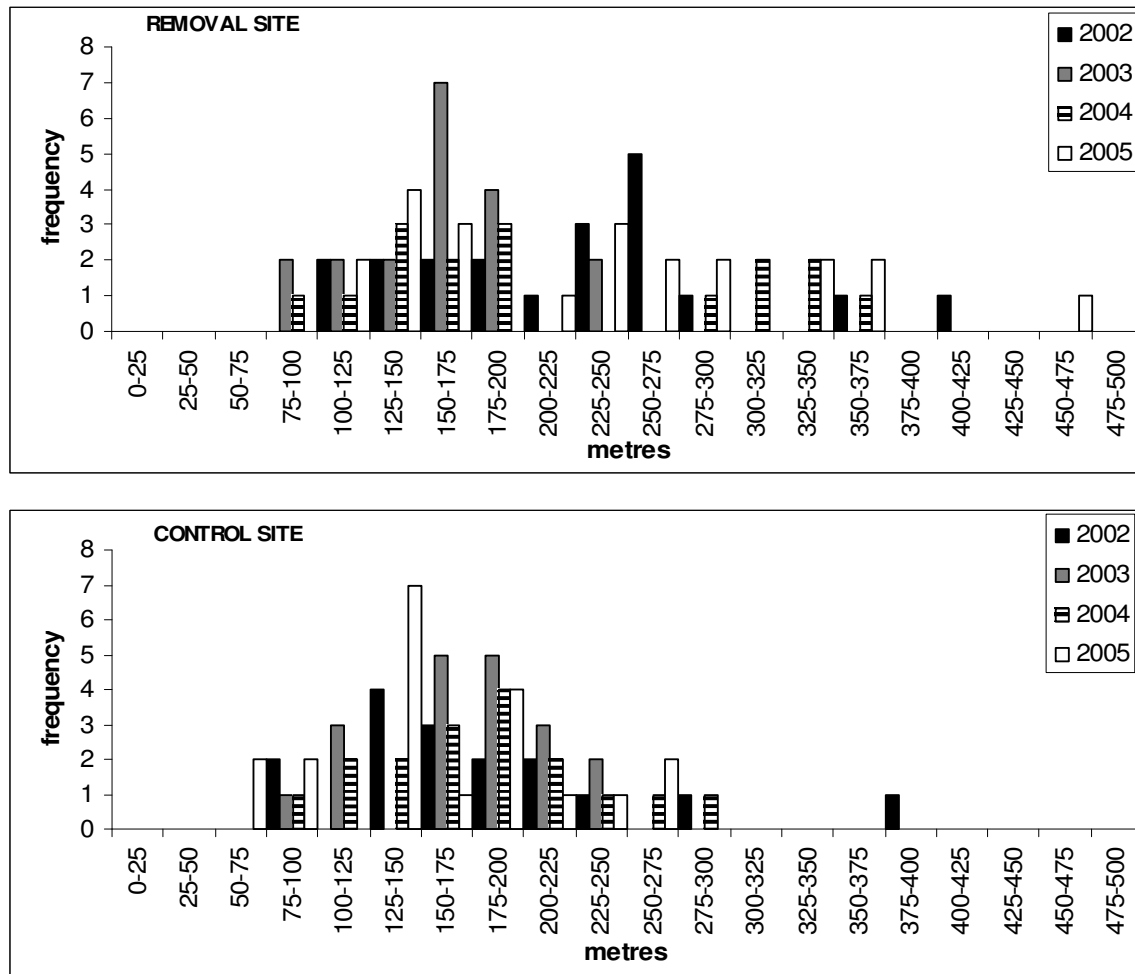


Figure 5.3: Distribution of nearest-neighbour distances. Removals occurred between each of the years.

Table 5.2: Comparisons of the distributions of nearest-neighbour distances. Distributions are shown in Fig 5.3.

	<i>nln2D*</i>	<i>nln2D_{crit}</i>	<i>P value</i>	<i>Distributions</i>
<i>Removal site</i>				
2002 v 2003	188	160	0.01	Different
2002 v 2004	72	140	>0.1	Same
2002 v 2005	74	176	>0.1	Same
2003 v 2004	114	133	>0.1	Same
2003 v 2005	203	169	0.01	Different
2004 v 2005	76	150	>0.1	Same
<i>Control site</i>				
2002 v 2003	50	133	>0.1	Same
2002 v 2004	25	124	>0.1	Same
2002 v 2005	56	140	>0.1	Same
2003 v 2004	38	141	>0.1	Same
2003 v 2005	129	160	>0.1	Same
2004 v 2005	87	146	>0.1	Same
<i>Removal v Control</i>				
2002 v 2002	116	140	>0.1	Same
2003 v 2003	76	171	>0.1	Same
2004 v 2004	86	124	>0.1	Same
2005 v 2005	174	176	0.06	Same

*Two-sample Kolmogorov-Smirnov statistic

Sex ratio and bird size

The sex ratio of removed bristlebirds did not change over the three years ($\chi^2 = 0.19$, $df = 2$, $P > 0.1$) (Table 5.3). The wing lengths of male bristlebirds varied significantly during the three removals ($F_{2, 24} = 3.869$, $P = 0.035$). A Tukey HSD post hoc test revealed the significance was due to a decrease in male wing lengths from 76.4 ± 1.8 (sd) mm in 2003 to 73.5 ± 2.6 mm in 2004 ($P = 0.027$). There was no other variation in the morphometrics measured in either males or females between any of the years.

Table 5.3: Number of males and females removed and average wing lengths (\pm sd).

<i>Year</i>	<i>Males</i>	<i>Wing length (mm)</i>	<i>Females</i>	<i>Wing length (mm)</i>
2003	8	76.4 (1.8)	6	72.6 (3.2)
2004	10	73.5 (2.6)	10	70.8 (2)
2005	4	74 (1.8)	4	71.3 (1.3)

Discussion

The loss of individuals from a population may be expected to exacerbate local extinction rates or lower reproductive output. However, following the removals of bristlebirds from the wild population at Bherwerre Peninsula, there was no observed impact on the numbers of bristlebirds surveyed in any of the subsequent years. Removed bristlebirds were presumably replaced each time. A common suggestion in territorial species is the presence of floaters, which replace individuals lost to the population (Newton 1992).

The distribution of bristlebirds was expected to be uniform in this territorial species and when birds were removed, this uniformity may be expected to be disrupted. Bristlebirds were uniformly distributed in all years in the control site. In the removal site the distribution of the nearest-neighbour distances becomes bimodal in 2004 and 2005, suggesting bristlebirds are starting to clump in their distribution through the habitat, possibly as numbers contract.

After the first removal of 16 bristlebirds, there was a decrease in the average nearest-neighbour distances in both the control and removal sites. This decrease was statistically significant at the removal site but not the control site. Bristlebirds were closer together after this first removal than in any other time throughout this study. Following the removal, there were new birds recruited into the calling, or presumed breeding population, as no impact was observed in the numbers of birds. This decrease in nearest-neighbour distances may imply a small increase in bristlebird numbers. Knapton and Krebs (1974) found that, when territory-holding Song Sparrows were removed at one time, there were more replacements than removals and territory sizes were smaller.

The second removal, in 2004, was the largest with 20 birds removed. There was a decrease in the number of bristlebirds surveyed following this removal, although this was not statistically significant. In contrast to the first removal, there was an increase in average nearest-neighbour distances at the removal site. Removed individuals were replaced, but perhaps not to the extent that they were in the first removal. During the

removal of Willow Warblers, Arvidsson and Klaesson (1984) found there was incomplete replacement of the removed birds and that the remaining individuals expanded their territories to encompass some of the vacant area. In the present study, the increase in average nearest-neighbour distances suggests that there may be fewer bristlebirds than before this removal.

There was a significant increase in bristlebird numbers in the removal site just prior to this second removal. This increase in bristlebirds at this site was attributed to a wildfire and may have provided the necessary individuals to mask the full impact of the removal. The wildfire, in December 2003 three months before the second removal, significantly compromised the original design of this study. The fire burnt approximately half of the control site and less than 25 % of the removal site. In a study investigating the post-fire recovery of bristlebirds after this fire at Jervis Bay (Chapter 6), I found increases in bristlebird numbers in unburnt vegetation adjacent to burned vegetation. These increases were assumed to be birds moving from burnt habitat escaping the fire. The movement of birds away from a fire and from the resulting burnt habitat is documented in literature (Wooller and Calver 1988; Burbidge 2003). These displaced birds generally return when conditions become more suitable (Woinarski and Recher 1997; Burbidge 2003). The movement of bristlebirds in response to this fire has confounded the results of this removal study.

There was no observed impact on bristlebird numbers after the third removal, although the distribution of nearest-neighbour distances was significantly further apart after this removal when compared to the distribution in 2003, after the first removal. This may be evidence of bristlebirds spreading out across the habitat, or of the significant increase in bristlebirds that followed the first removal. The distribution in Fig 5.3 shows that more bristlebirds were further apart in 2005 than in any other year. Similar results were found to this study in American Redstarts (*Setophaga ruticilla*) when Sherry and Holmes (1989) removed 15 males. These were replaced by another 15 males of mixed ages. In another removal experiment, Cederholm and Ekman (1976) observed no differences in density between control and removal sites six months after the removal of Crested Tit and

Willow Tit. All these populations must have some mechanism counteracting the loss of individuals in the population.

The origin of the replacement bristlebirds is unknown, and this is common in removal experiments (Newton and Marquiss 1991). Whether replacements are territory-holders moving from poorer to better habitat, territory-holders expanding their territories, or floaters in the population is difficult to test and rarely examined (Newton 1992). A number of studies have shown that replacements after removals did come from floaters in those populations (Sherry and Holmes 1989; Danks 1994; Sternberg *et al.* 2002). There was a possible increase in bristlebirds after the first removal, followed by a slight decrease in numbers. However, there was no significant change in the numbers of bristlebirds surveyed over the three years of this study. There must be some mechanism stabilising the bristlebird population around these removals.

Replacements may have been dispersing juveniles. Removals occurred approximately two months after the bristlebird breeding season, to avoid stress to the population during nesting. Removals may have been occurring at the same time as some fledged juveniles were dispersing through the habitat. The removal of bristlebirds may have provided vacant territories for juveniles to occupy whether from the removal site or outside. I calculated approximately 200 hectares of habitat within the direct vicinity of the transect (within 250 m either side). At an average density of two bristlebirds per 5 hectares (Pyke *et al.* 1995; Baker 2001) and using Baker's (1997) population growth rate of 14% per year, it would be expected that the population within this area could increase by about 12 birds each year. Given that there is more habitat beyond 250 m from the transect it is plausible that the replacements were dispersing juveniles. However, apart from a significant drop in male wing lengths from 2003 to 2004 of only 2.8 mm, there were no other changes in the size of morphological measurements taken from removed bristlebirds that may have suggested that they were younger birds. Monkkonen (1990) found that during removal experiments, over two years there was a significant decrease in the tarsus and wing length in more than 9 bird species studied. It seems doubtful that all replacements were dispersing juveniles as it is unlikely that a species renowned for its

low fecundity (Higgins and Peter 2002) should produce new recruits at or above the natural rate of increase (Baker 1997) for three years.

Density-dependent changes in fecundity may be alluded to by this removal of individuals and the apparent rise in recruitment. However, there are unlikely to be density-dependent changes in fecundity as overall there was no change to density across the site. The period between the removals and the surveys did not encompass the breeding season and this effect would be expected to lag one year behind the removals which was not the case.

Floaters are common to many territorial species and if juvenile dispersal is unlikely to account for all replacements in this study, it lends support to the possibility that some replacement bristlebirds came from a non-calling surplus of individuals. In removal experiments on nine bird species in Finland, Monkkonen (1990) found that, in all plots, the removal of individuals subsequently resulted in a higher number of birds recorded. The combination of juvenile dispersal and the presence of floaters may have together reduced the impact of removals on this bristlebird population.

The sex ratio in the birds that were removed did not change, suggesting that there was no change in the sex-ratio in the population. If one sex was represented more than the other within the floaters, then replacements may be expected to vary the sex-ratio in the population. After the removal of American Redstarts, Marra *et al.* (1993) found there was a significant shift in sex ratio, with the mainly male floaters excluding females from mutually acceptable territories. A possible difference with bristlebirds may be that pairs are assumed to live in territories and the removed bird may be replaced by the same sex.

The role of floaters in the dynamics of a population is not completely clear. In some species, floaters seem to be a common feature of populations; whereas in other species, floaters were only present in some seasons, years or areas (Newton 1992). Having a non-territorial sector in the population may be a useful technique for population persistence in an intermittent or temporally variable habitat. This may be especially true where dispersal is limited, such as with the semi-flightless bristlebird. Floaters would be able to replace

individuals lost during catastrophes or environmental stochasticity, if they themselves survived. If the presence of floaters in a population is due to a limitation of available habitat (Sherry and Holmes 1989), then these populations may be good candidates for translocation. The potential impact to a population from the removal of individuals would be expected to be less with an available source of recruits such as floaters.

There are implications for the conservation of bristlebirds of the potential presence for floaters in a population. It would be valuable to determine if floaters are truly present in the population at Jervis Bay and whether they are present in all bristlebird populations, or whether it is a function of population size and available habitat. If there are floaters present, is this a standard state of this population, or perhaps suitable habitat or some other parameter is limiting population growth at Jervis Bay.

There are implications for monitoring with the presence of floaters in a population. It is almost certain that not all birds will be surveyed during monitoring, resulting in a misrepresentation of the size of the population. However, when monitoring populations of endangered species, a measure of the effective population size, encompassing only those individuals in the breeding pool, may be a more meaningful measure of population status.

Post-fire recovery of Eastern Bristlebirds

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This chapter has been written as a manuscript and as such it will contain some repetition in the introductory section from Chapter 1 and some overlapping in the methods section with other data chapters. All species are reintroduced.

Chapter 6

Post-fire recovery of Eastern Bristlebirds

Introduction

Fire has been present in Australia since the Tertiary Period (Kemp 1981) and much of the flora of Australia has evolved characteristics that allow survival after this disturbance (Gill 1981). Much of Australia's fauna has also evolved with the influence of fire resulting in a range of responses from avoidance to exploitation (Keith *et al.* 2002; Whelan *et al.* 2002). The variation in response to fire among Australia's fauna can be explained by a combination of life history, biology and fire regime (Whelan *et al.* 2002), although escape behaviours are expected to become less effective as the size, frequency or intensity of fire increases.

Many species, including some bird species, take advantage of post-fire conditions. Loyn (1997) found that birds that feed in open ground, such as the Flame Robin (*Petroica phoenicea*), Scarlet Robin (*Petroica multicolor*), Buff-rumped Thornbill (*Acanthiza reguloides*) and Superb Fairy-wren (*Malurus cyaneus*), thrived for three years post-fire to levels greater than before fire by exploiting low shrub regrowth. Some aerial insectivores and raptors respond immediately to take advantage of fire, hunting disturbed insects or vertebrates (Woinarski and Recher 1997).

The response to fire of some species can vary as the inter-fire period changes, although it may not be sufficient to describe the variation in responses within a species. Post-fire trends cannot be viewed as repeated responses, as long-term fire histories will be

different and presumably more important than the inter-fire period (Bradstock *et al.* 2005). This variation in response to fire was recorded in the Splendid Fairy-wren (*Malurus splendens*), which had no direct mortality following a fire after a six-year fire-free period (Rowley and Brooker 1987). However, two more fires in the ensuing three years each caused a decline in Splendid Fairy-wren numbers (Russell and Rowley 1993).

There is a range of species that are sensitive to fire because they are unable to avoid the direct effects or they depend on resources that are removed by fire such as dense vegetation. For example, Rufous Bristlebirds (*Dasyornis broadbenti*) were not recorded in the area of a single large fire in eastern Victoria until 2 years post-fire (Reilly 1991). The Western Bristlebird (*Dasyornis longirostris*) took 2 - 6 years post-fire to recolonise moister areas and up to 14 years to recolonise some drier areas (Smith 1987; Burbidge 2003). Another fire-sensitive species, the Noisy Scrub-bird (*Atrichornis clamosus*) took 4 - 10 years to begin recolonising burnt areas (Danks 1997).

Both individual wildfires and altered fire regimes have been identified as threats facing many species in Australia's bird fauna (Garnett and Crowley 2000). Of Australia's 155 threatened bird species, 47% have wildfire and/or inappropriate fire regimes listed as current threats to their long-term survival (Garnett and Crowley 2000). The Eastern Bristlebird (*Dasyornis brachypterus*) is listed as threatened in all jurisdictions of its range: nationally endangered under the *Environment Protection and Biodiversity Conservation Act 1999*, endangered in New South Wales under the *Threatened Species Conservation Act 1995*, endangered in Queensland under the *Queensland Nature Conservation Act 1992* and threatened in Victoria under the *Victorian Flora and Fauna Guarantee Act 1988*. The main threats have been identified as habitat loss and fragmentation, introduced predators and inappropriate fire regimes (Garnett and Crowley 2000; DEC 2004).

The bristlebird is a cryptic, ground-dwelling, insectivorous and semi-flightless passerine. It is also described as fire-sensitive, based primarily on the work of Baker (1997; 2000; 2003). From various studies Baker (2003) summarised that fire was implicated in the

extinction of at least 12 populations of bristlebirds in the last three decades. Fire temporarily removes dense understorey vegetation, which is the bristlebird's preferred habitat. Being semi-flightless and cover-dependent, the bristlebird is not expected to be able to colonise new areas readily or to recolonise areas following disturbances (Smith 1977; Baker 2000) such as fire.

Various studies of fire responses of bristlebird populations have shown an array of results. Bristlebirds were known from 11 sites in 1978 near Mallacoota in Victoria but by 1994 as a result of fires, bristlebirds were only found at 1 of those 11 sites (Clarke and Bramwell 1998). Lamb (1993) surveyed, in 1992, 88 bristlebird territories that had been defined in 1989 near the NSW/QLD border and found that only 36 % of them contained birds. Declines in these northern populations have been attributed to inappropriate fire regimes (Hartley and Kikkawa 1994; DEC 2004). In the central populations, after five fires within 13 years at Barren Grounds Nature Reserve, bristlebird densities increased with increasing time since last fire from zero birds to approximately 2 birds per 5 hectares 11 - 16 years post-fire (Baker 1997; 2003). At Booderee National Park, Pyke *et al.* (1995) found no significant difference between the numbers of bristlebirds detected in habitat 0-7 years post-fire and habitat 13-14 years post-fire. At Nadgee Nature Reserve in 1972 and 1980 large-scale wildfires burnt most of the bristlebird habitat. By seven years post-fire, bristlebirds had colonised only a few of the areas burned in 1980 (Woinarski and Recher 1997). Baker (1997) proposed the Nadgee population was, at that time, recovering from near extinction after the fire in 1980 because the two fires had left refuges that were few and far between.

In late December 2003, a wildfire burned through approximately 3000 ha of Booderee National Park and adjacent parts of NSW on Bherwerre Peninsula at Jervis Bay on the New South Wales south coast. It was estimated that Bherwerre Peninsula at that time had in excess of 700 bristlebirds (D. Bain and J. Baker, unpublished data). Bristlebirds were being surveyed here prior to the fire as part of other research. This fire provided an opportunity to examine the direct effects of a summer wildfire on bristlebirds, because two transects had been surveyed two months prior to the wildfire and parts of them were

affected to different degrees by the fire. Thus, there was the opportunity to estimate bristlebird numbers after fire in a range of locations of varying fire intensity and different distances to unburned vegetation. The aims of this study were: (1) To determine the immediate effects of fire on the bristlebird population. (2) To determine the effect of fire within the first 13 months post-fire.

Methods

Post-fire survey

Within 2 weeks of the fire, call playback was used to try to elicit calls from bristlebirds throughout the burnt area. Ninety-two playback points were surveyed in 3000 ha of burnt vegetation (Fig 6.1). Areas known to have supported bristlebirds before the fire were targeted for playback. At each call playback point, a tape of bristlebird 'duets' was played for two minutes, followed by a five-minute listening period. All bristlebirds heard or seen were recorded, and their location estimated. Burnt habitat was assessed in a radius of 150 m from the playback point and was grouped into one of four categories:

1. High-intensity burn: canopy and understorey completely burnt.
2. Moderate-intensity burn: canopy scorched/burnt but some leaves remaining and possibly green; understorey completely burnt.
3. Low-intensity burn: canopy with unburnt patches; understorey generally burnt with occasional patches unburnt.
4. Burnt edge: any burnt habitat within 150 m of the fire boundary.

Data were analysed using a chi-square analysis, to test the hypothesis that the frequency of occurrence of bristlebirds after the fire was independent of fire intensity. Expected frequencies were calculated using a chi-square contingency table.

Before and after surveys

Two transects affected by the 2003 fire, one 4.5 km, the other 3.4 km long had been surveyed two months prior to the wildfire. These transects were subsequently surveyed 1, 9 and 13 months after the fire to assess changes in bristlebird numbers. Surveys involved a standard listening technique used previously in bristlebird research (Baker 1997; Bain and McPhee 2005). Transects were walked slowly, 2-4 km/h, and the locations of all bristlebirds heard or seen were mapped. The direction and distance of a bristlebird call can reliably be mapped by a competent observer (Bain and McPhee 2005). Both transects were within 800 m of the edge of the burnt area, with neither penetrating into the middle of the burnt area (Fig 5.1). Transects were each walked on four separate mornings within four weeks, with the direction of travel reversed each time. The survey with the maximum number of records from the four was used for data analysis. Vegetation was grouped into three categories: burnt, burnt edge and unburnt. The burnt edge category contained burnt vegetation along the fire boundary. The number of bristlebirds per 500 m was recorded for each vegetation category and were averaged across the transects. There were two burnt and unburnt replicates and three replicates in the burnt edge category. A repeated measures, general linear model was used to test the hypotheses that bristlebird numbers changed over time and that the nature of the change depended on the location of the birds in relation to burnt vegetation. The burnt category was adjacent to the burnt edge category which was adjacent to the unburnt category. The burnt edge category consequently was not independent of the other two categories, which is required for analysis. Therefore, the burnt edge category was not included in this analysis but analysed alone for any influence of time since fire on bristlebird numbers.

It is acknowledged that only one pre-fire survey period was measured. However, the variability of bristlebird numbers before the fire was expected to be low. Numbers of bristlebirds surveyed along just one of these transects only varied between 2.4 and 2.5 birds/500 m in the two years prior to this study (D. Bain, unpublished data).



Figure 6.1: Bherwerre Peninsula, Jervis Bay: The area of the 2003 fire is shaded, the thick black lines are the two transects and the grey dots are the 92 call playback locations.

Results

Post-fire survey

Bristlebird occurrence in the second week after the fire was dependent on fire intensity (proportion of sites with birds $P < 0.008$, $\chi^2 = 11.8$, $df = 3$). The frequency of survey sites with birds present was much lower in intensely burnt areas than less intensely burnt areas. In the areas burnt with high-intensity only 4 % of sites recorded bristlebirds but in the areas burnt with low-intensity, 50 % of sites recorded bristlebirds (Table 6.1).

Table 6.1: Number of sites with Eastern Bristlebirds observed two weeks post-fire and the number of bristlebirds expected from a contingency table. Surveyed column indicates the total number of sites surveyed in each fire intensity category.

<i>Vegetation category</i>	<i>Sites with birds</i>	<i>Expected</i>	<i>Surveyed</i>
High-intensity	1	6	24
Moderate-intensity	11	9	35
Low-intensity	8	4	16
Burnt edge	3	4	17

Before and after surveys

There was a contrasting impact of the fire on bristlebird numbers in burnt compared to unburnt vegetation over the whole 15 months (Fig 6.2). This interaction between time since fire and vegetation category was significant ($F_{3,6} = 8.064$, $P = 0.016$). Over the time-span between 2 months before the fire to one month after, burnt vegetation showed a marked decrease in bristlebird occurrence from 2.18 ± 1 (sd) to 1.35 ± 0.5 birds/500 m. Unburnt vegetation showed a large increase from 0.58 ± 0.8 to 2.35 ± 0.5 birds/500 m. This interaction between time since fire and whether the vegetation was burnt was significant during this period ($F_{1,2} = 40.119$, $P = 0.024$). From 2 to 9 months there was very little change in both burnt and unburnt vegetation. By months 9 to 13 numbers in the burnt vegetation had increased to more than pre-fire numbers and numbers in the unburnt vegetation had returned towards pre-fire numbers. The interaction between time since fire and vegetation category during this period was influential over the whole study but was not significant ($F_{1,2} = 7.553$, $P = 0.111$). There was no significant effect of time since fire on the vegetation category 'burnt edge' ($F_{3,6} = 0.963$, $P = 0.469$). This vegetation category showed the most variability in bristlebird numbers over the 15 months (Fig 6.2).

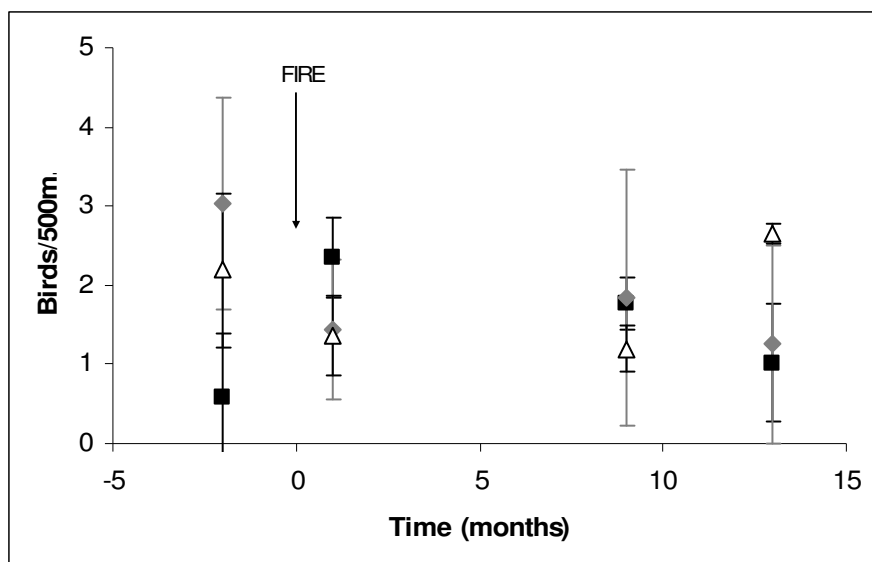


Figure 6.2: Number of Eastern Bristlebirds (\pm sd) observed per 500 metres in the months around the fire. ■ unburnt habitat, ◆ burnt edge habitat, △ burnt habitat.

Discussion

Fire intensity influenced the frequency of occurrence of bristlebirds at Booderee National Park. Two weeks after the fire, the more intensely burnt areas had fewer bristlebirds than the less intensely burnt areas. In forests in south-eastern New South Wales, Smith (1989) found a similar relationship for 10 bird species associated with dense understorey shrubs. With these species, such as the White-browed Scrubwren (*Sericornis frontalis*) and Brown Thornbill (*Acanthiza pusilla*), he found fewer records in the more intensely burnt areas. A similar relationship has also been described for Western Bristlebirds which survived a mild-intensity fire but were killed in more intense wildfire (Burbidge 2003).

Fire decreased bristlebird numbers in the burnt and burnt edge sites. This probably reflects the mortality and displacement of birds that often occurs in recently burned habitat (McFarland 1988; Smith 1989; Burbidge 2003). The concomitant increase in density in unburnt habitat is important because it suggests that some bristlebirds moved

from burnt to unburnt habitat, presumably escaping the fire and subsequently remaining away from the burnt habitat for some months. Burbidge (2003) found that Western Bristlebirds survived some milder fires and subsequently set up home ranges in nearby unburnt vegetation. The findings of the present study suggest that this may also happen after higher-intensity fire.

Prior to the fire, bristlebird densities varied between the sites. In particular the unburnt sites had very few bristlebirds. This may be interpreted as differences in habitat quality, although all sites were located in a continuous stretch of habitat which was considered, superficially, to differ very little between sites. The low number of replicates in this study almost certainly contributed to the variation in bristlebird densities. Caution should be applied when interpreting such variable data with low replication.

From 9 to 13 months post-fire, the frequency of occurrence of bristlebirds in the burnt areas, away from the edge, increased to more than pre-fire levels and numbers in unburnt sites declined back towards pre-fire numbers. Although the differences between burnt and unburnt vegetation over this period were not significant, the return towards pre-fire numbers of bristlebirds may represent the return of displaced individuals back to pre-fire home ranges, a build-up of populations, a spreading of remaining bristlebirds across the habitat or juvenile dispersal. The spreading of bristlebirds across the habitat may be due in part to the species territorial behaviour (McNamara 1946; Chapman 1999; Higgins and Peter 2002) as territorial behaviour is known to influence bird density tending it towards even distribution across similar habitat (Newton 1992). Alternatively, dispersal may be associated with juvenile birds. If available unburnt habitat was filled, as there were more birds than before the fire (Fig 6.2), then the initial drop in bristlebird occurrence in burnt vegetation may be serving as vacant habitat for juvenile dispersal.

The numbers of bristlebirds in the burnt edge category was the most variable of the three vegetation categories at each of the time periods (Fig 6.2). As vegetation in the burnt edge category lay between the burnt and unburnt vegetation, this variation may reflect movement of bristlebirds between burnt and unburnt vegetation following the fire.

There are other examples of birds returning to previously occupied sites soon after a fire. Woinarski and Recher (1997) reported Silvereyes (*Zosterops lateralis*) foraging in burnt habitat within minutes of an intense fire that burned a relatively small area. In a low to moderate intensity fire in heathland near Sydney, birds avoided flames by moving to adjacent unburnt habitat then immediately returned after the fire had passed (Recher and Christensen 1981). At a longer time frame, marked White-browed Scrubwrens (*Sericornis frontalis*) and Inland Thornbills (*Acanthiza apicalis*) were re-caught in the same areas in the months after fuel reduction burns in Karri and Jarrah forests (Wooller and Brooker 1980; Wooller and Calver 1988). In the present study, the changing patterns of bristlebird abundance suggest that bristlebirds took over 9 months to return to original areas post-fire, perhaps spending the winter and non-breeding season surviving in areas other than their pre-fire home ranges.

The apparently short-term impact of the fire on bristlebird occurrence on Bherwerre Peninsula after one year is different to the results from Barren Grounds Nature Reserve (Baker 1997; 2003), which suggested that fire can devastate bristlebird populations and recovery of the population may take over ten years. However, Pyke *et al.* (1995) found that fire-age did not have a significant effect on bristlebird occurrence at Booderee National Park, although notably sample sizes were small and results were confounded by the proximity of unburnt vegetation.

As with the Pyke *et al.* (1995) study, the two transects used in this study were close to the fire boundary due to the opportunistic nature of this study. The greatest distance between a surveyed bristlebird and the fire boundary throughout the whole study was only 900 m. Bristlebirds are capable of travelling at least 300 metres per hour during daily activities (Baker and Clarke 1999). Post-fire, there were numerous lightly burned and unburnt refuges apparent throughout the study sites. Complex interactions between topography, fire and fuel conditions can result in small refuges being left unburnt (Whelan 1995). The proximity of unburnt habitat and the presence of refuges suggest that bristlebirds in this study may have been able to escape the fire. Different results may have been

recorded had this study included areas further from the fire boundary, although results suggest that in the less intensely burnt areas many bristlebirds survived. The need to examine the proximity and influences of unburnt areas when making conclusions about post-fire avian activity has been suggested previously (Pyke *et al.* 1995; Baker *et al.* 1997; Baker 1997).

The fire in Booderee National Park was the first large fire in 32 years. In 1972/73 about half of the park burned and since then, there has been a series of only small fires; many of which were prescribed hazard reduction burns (Pyke *et al.* 1995; Taws 1998). In contrast, there have been five large fires, each burning 10-50% of available habitat at Barren Grounds Nature Reserve during 1979-1991 (Baker 1997). It is possible that the difference in bristlebird response between Barren Grounds and through to the Booderee studies is the high fire frequency combined with the area of each burn at Barren Grounds. Similarly, after large-area fires in 1972 and 1980 burnt most of the bristlebird habitat in Nadgee Nature Reserve, bristlebirds had only colonised a few of the burnt areas after 7 years (Woinarski and Recher 1997).

Several studies have reported that there was little impact of a single fire on populations of some small bird species, after a fire-free period. For example, the Splendid Fairy-wren showed no direct mortality due to a large wildfire in 1985 after 6 fire-free years, and all birds remained on site with none moving from burnt to unburnt vegetation (Rowley and Brooker 1987). However, after a series of small repeated fires in the years following the 1985 wildfire, Russell and Rowley (1993) found that the numbers of birds had declined from 0.9 to 0.6 birds/ha. Individuals of two small cover-dependent passerine species, the Brown Thornbill and the White-browed Scrubwren managed to survive a single high-intensity fire and continued to use the same areas as before the fire (Cowley 1974).

There is another site-specific factor that may have contributed to the observed recovery of the bristlebird population at Booderee National Park compared to Barren Grounds and Nadgee Nature Reserves. Fire opens up the habitat by removing most of the vegetative cover, so many researchers have suggested that this can cause an increase in predation

(Recher and Christensen 1981; Brooker and Brooker 1994; Loyn 1997). Since 1999 at Booderee National Park, and much of the Bherwerre Peninsula, there had been an intensive baiting program for European Foxes (*Vulpes vulpes*). In particular, baiting occurred immediately after the fire in 2003 (Nick Dexter pers. comm.). Fox control may have influenced fox predation in Booderee but presumably it would have had no effect on native predators such as snakes, goannas and birds of prey or on some other feral predators such as cats and rats. Carcasses of animals are common after wildfire, providing an initial food resource for animals prepared to scavenge and may initially reduce predation pressures on fire survivors. Consequently, bristlebirds in the post-fire environment at Booderee may have been under less predation pressure than in the areas of previous studies on fire and bristlebirds. The interaction between feral predator control and post-fire responses of cover-dependant species such as the bristlebird requires investigation.

There were differences in impacts to bristlebird occurrence as a result of different fire intensities. Different fire intensities can arise from differences in vegetation and landscape features along with climatic factors. There have also been differences in the reported impacts to and recovery of bristlebird populations after fire from studies at Barren Grounds Nature Reserve and Jervis Bay. The differences in fire frequency and fire extent between Barren Grounds and Jervis Bay, coupled with differences in feral predator control, are likely to have contributed to the different responses to fire of these two populations. The results of this study support a statement by Bradstock *et al.* (2005) that the response of a species to fire will be highly variable and strongly context-dependent. Post-fire trends can not be viewed as repeated responses, even in two sites which have the same time elapsed since last fire, as fire histories of the sites are almost certainly going to be different along with the intrinsic landscape features and vegetation features of the habitat (Bradstock *et al.* 2005).

The response of bristlebirds to fire in this study was unexpected. Within a year, post-fire bristlebird numbers were similar to pre-fire levels. It was shown that bristlebird populations can survive fire and that they do better in less intensely burnt areas.

Bristlebirds may survive fire if in proximity to unburnt vegetation and refuges and following it are able to survive in recently burnt vegetation. Familiarity of habitat, available food resources and feral predator control all had a potential, but untested, influence on the bristlebird response to this fire. The results suggest that cover-dependant birds may escape fire by temporarily moving to nearby unburnt vegetation and then return when conditions are more suitable. The results strongly support the notion that avian responses to fire are context-dependant.

Management issues

Bristlebirds live in fire-prone vegetation and fire management is of high priority in many of these areas for property protection and conservation. The general recommendations for bristlebird management is currently fire exclusion (Baker 1997). However, in many areas this is not satisfactory for other fire management goals such as property protection, or indeed for other species of fauna or flora which may prefer more open vegetation. The results of the present study indicate that fire may be used strategically in bristlebird habitat to satisfy management goals for other species or for bushfire mitigation without serious threat to bristlebird populations. If prescribed burns can be achieved by small-area fires, which leave unburnt refuges either within the burn area or nearby, then fire management and bristlebird conservation should be achievable. This is not suggesting that fire in bristlebird habitat is necessary, but that fire may be strategically utilised if the environmental or social conditions require it.

Chapter 7

General discussion

The reintroduction

The popularity of translocations has provided the stimulus for a number of reviews that have examined aspects of translocations that are associated with success and aspects of translocation that have received too little attention. These include (i) the completion of a feasibility analysis prior to commencement, (ii) evaluation of success against criteria stated prior to commencement and that are adaptable to a range of projects, (iii) increased financial accountability for use in future planning, (iv) the use of experimental techniques to further the understanding of processes and species, and (v) the publication of translocation projects whether they are a success or failure to allow easy access to information. In this research I have used a single translocation program, the reintroduction of the Eastern Bristlebird (*Dasyornis brachypterus*) to part of its former range to advance two objectives (i) a better conservation outcome for this particular endangered species, and (ii) an assessment of four of these five key components of translocation programs. The principal aim of this thesis was to assess the potential and success of translocation for the conservation of small Australian passerines using the bristlebird as a case study and was focused on the following four aspects.

Key components of translocations

Feasibility analysis

Aspects of translocations that have been correlated with success include high habitat quality of the release site, reintroduction into part of the former range of the species, large number of individuals released and the use of a wild source population. A feasibility analysis, incorporated in a Translocation Proposal for the NSW Department of Environment and Conservation (Whelan and MacKay 2002) indicated that the proposed release area was in part of the former range of the bristlebird and contained a large area of good habitat. Bristlebirds were to be removed from a wild source population for the reintroduction. Other aspects of the feasibility analysis such as financial and community support were also favourable. The outcomes of the analysis suggested that reintroduction of the bristlebird was a potential conservation strategy for the species.

In completing these prerequisites, support was gained from land managers and the local community, securing a source population and suitable host environment. Alternate sources of birds or a less suitable host environment may have led to a less successful program as a result of increased time, travel and stress to birds and costs. The feasibility analysis included estimated costs and sufficient funds were subsequently sourced prior to the commencement of the program to ensure the program did not fail due to lack of funding. The feasibility analysis was critical in the preparation for this successful reintroduction program.

Criteria for success

The efficacy of the translocation was investigated by assessing the success of the reintroduction. The reintroduced population has survived for three years and has produced young within that time. The reintroduction of the bristlebird has achieved all

criteria for success in the release environment in the mid-term including breeding in the host environment and recovery in the source population.

The criteria developed to evaluate the success of this reintroduction were specifically designed to be adaptable to many projects or species for comparison. Criteria were developed for both the source population and the reintroduced population. They were based on a timescale of short, mid and long-term periods that were measured in generation time or time till first breeding. This timescale allows a more informative comparison of projects that may have been carried out some time apart. A measurement of time based on the species physiology is perhaps also more useful when comparing a range of species than a less biologically meaningful measurement of time such as years, although years will be a meaningful surrogate to generation time in a lot of cases. The use of generation time will allow comparisons between very long-lived animals such as primates and short-lived animals such as small mammals, particularly if investigating the potential genetic and demographic impacts to populations that may result from translocation.

Financial accountability

This translocation cost much less than other published accounts of translocation programs and was the cheapest conservation technique examined with the goal of increasing the security of the species.

The cost of this translocation exercise was broadly broken down into four categories: Personnel costs, materials and equipment, consumables and reporting. This reintroduction was much cheaper than other published accounts of the economic costs of translocation projects (Kleiman *et al.* 1991; Cohn 1993). I also compared the cost to other conservation strategies for the bristlebird, captive breeding and the purchase of private land for the preservation of habitat. Translocation of wild bristlebirds to habitat on reserved land is the cheapest potential option for intensive management of the species, although this is not

true in all parts of the bristlebird's distribution. In the north of its distribution, there are not enough wild birds to sustain the removal of individuals for translocations. Here, more expensive measures are underway with a captive breeding program in its early stages (DEC 2004). These expensive and intensive conservation strategies should be viewed as last resorts. Management of threatened species should begin by allocating resources to managing threats in current locations and if possible securing habitat for natural dispersal.

Experimental approaches

Sexing the bristlebird

An investigation into the potential for sexual dimorphism in the bristlebird and any potential for a field-based sexing technique found that bristlebird weight and head-bill length can be used to sex bristlebirds in the hand. There has previously been a small size dimorphism suggested between the sexes in bristlebirds (Chaffer 1954), although they have been widely considered a sexually monomorphic species (Higgins and Peter 2002; DEC 2004). Bristlebirds were sexed genetically and then weight and head-bill length were analysed to develop univariate criteria and a multivariate discriminant function for sexing bristlebirds in the field. When these two techniques were used in conjunction, the results agreed with the molecular sexing results in 80 % of cases and differed in only 7 % of cases.

These techniques for developing sexing criteria have been applied to a range of species (Pyke and Armstrong 1993; Rogers 1995) and can be useful for conservation. This technique has applications to bristlebird management as it provides a relatively quick and inexpensive method to investigate sex ratios in bristlebird populations and aid in the selection of individuals during further translocation projects. The ability to immediately sex an individual in the hand potentially provides savings in time and money during any manipulations of populations.

Released individuals

The post-release dispersal of reintroduced bristlebirds was investigated for any differences in dispersal behaviour between males and females and between bristlebirds released with and without conspecifics present and differences to bristlebirds in native habitat. Translocated bristlebirds moved through larger areas than those in native habitat presumably due to unfamiliarity with the environment. It was suggested that males were searching for high quality home ranges and females were settling quickly when male cues were found in the environment. It was also suggested that released bristlebirds were searching for vacant habitat although at the same time other bristlebirds provided some stimulus for settlement. In the conservation of current populations of bristlebirds, individuals can not be expected to colonise isolated or tenuously connected habitat away from the presence of other bristlebirds. Dispersing birds may be lost to the population as fragments of habitat become sinks. Any fragmentation of habitat is likely to cause a decline in bristlebird numbers.

Some bristlebirds travelled up to 4 km from the release site, substantially further than previous observations of bristlebird movements in their native home ranges (Baker 2001). However, Danks (1991) found in a similar species, the Noisy Scrub-bird (*Atrichornis clamosus*), that translocated individuals dispersed only half the distance of some young adults dispersing in their native habitat. Therefore, it may be reasonable to assume that if there is suitable available habitat near to existing populations, then bristlebirds have the ability to disperse into those areas. However, due to the semi-flightless nature of the bristlebird any dispersal will require continuous habitat.

Source population

The removals in the source population were investigated using a simple before and after design with control and treatment sites. The sustained removal of between 2 and 3.5 % of the population each year for three years did not have an observed detrimental impact. The origin of the replacements was unknown and the speed of the recovery after removals was unexpected. One suggestion was that the replacements were coming from a non-calling (undetected), non-territory holding surplus in the population, commonly called

floaters. This surplus has been commonly reported in territorial species (Newton 1992). There are two implications if this population contains floaters. It may be acting as a mechanism for population persistence in an unpredictable environment. Following disturbances, there is an immediate source of recruits to replace any individuals killed, although this will only be effective in connected habitat that is affected to varying degrees by the disturbance. Alternatively if floaters are particular to the population on Bherwerre Peninsula then perhaps lack of available or connected habitat is limiting population expansion in this area. Investigations of other populations and the actual source of the replacements would be required to accurately determine the structure of bristlebird populations.

Two management issues arise as a consequence of a non-calling surplus in populations of bristlebirds and other species. It must be understood that monitoring using current aural survey methods will only be detecting calling individuals, and will not deliver a true measure of the population size. Although they will still give perhaps a more meaningful measure, that of the effective population size if it is assumed that calling birds are the reproductive territory holders. Populations that contain floaters may be good candidates for use as source populations for translocations. Whether the reason for the presence of floaters is restricted habitat or a particular population dynamic, these populations should be better able to cope with the removal of individuals than populations without such a surplus.

Recovery after fire

In December 2003, three months before the second translocation, approximately half of the bristlebird habitat on Bherwerre Peninsula was burned in a hot summer wildfire. This fire provided the opportunity for a natural experiment as some sites that had been monitored in the previous year were burned and some were not. From some previous reports on fires and bristlebirds (Baker 1997; Woinarski and Recher 1997; Baker 2000), initial expectations of the impact from the fire were not very optimistic.

The effects of fire intensity on bristlebird occurrence immediately after fire were investigated and bristlebird recovery over the first 13 months post-fire was monitored. There were more bristlebirds found in the less intensely burnt areas immediately after the fire. Bristlebirds decreased in burned areas but, unexpectedly, by 13 months post-fire densities of bristlebirds had returned to, or beyond, pre-fire numbers. Many bristlebirds were assumed to have escaped the fire by temporarily moving to unburnt areas, as densities of bristlebirds increased in unburnt sites immediately following the fire and then slowly declined during the next year.

The unexpected recovery of bristlebirds was attributed to the context of this particular fire. Bradstock *et al.* (2005) stated that the response of a species to fire will be highly variable and strongly context-dependent. There are four specific factors related to this particular fire that may have contributed to the fast recovery of bristlebirds: 1) This was the first fire in 32 years. 2) There were high numbers of bristlebirds recorded on Bherwerre Peninsula. 3) The proximity of unburnt habitat and refuges for many of the bristlebirds allowed them to survive the fire. 4) Fox baiting was being undertaken regularly prior to the fire and was carried out immediately after the fire.

Current fire management for bristlebirds involves fire exclusion, reducing the extent of fires in bristlebird habitat, long inter-fire periods and if possible the prevention of converging fire fronts (Baker 1997; 2003; DEC 2004). These prescriptions will not always be satisfactory for other fire management goals. However, if prescribed burns are required and can be achieved by small-area fires, which leave unburnt refuges either within the burn area or nearby, then fire management and bristlebird conservation may not be mutually exclusive. As the responses of species to fire are context-dependent, then any broad prescription for fire for a particular species needs to be undertaken in an adaptive management framework to help understand site specific factors.

The value of translocations

In-situ management of species will not be replaced by translocation. It is an effective tool that can be used to aid in conservation and supplement, but not replace existing management. The costs of translocation programs can be high and the logistics and practicalities can often be difficult and all need to be considered when evaluating the potential of translocation for a particular species.

Translocation may not be useful for all species. This reintroduction has raised some aspects of bird species that may make them amenable to translocation. Species that interact regularly with other individuals in the population may translocate well if a large enough number are translocated. Any social interaction in the host site may provide the necessary stimulus to remain in the area if the habitat is suitable. However, this does not mean that more solitary species will not translocate successfully. Species that are sedentary in habit, that do not disperse long distances or that are faithful to a specific area at least at some times during the year may translocate well if site faithfulness can be reinforced. Species with broad diets may also translocate well due to their dietary flexibility. This translocation potential may be extended to insectivorous bird species over herbivorous or nectivorous species. The latter may be more particular about specific plant food species than insectivorous species might be over specific insect prey species. This difference may lead to insectivorous bird species being more flexible in their potential release areas. A broad range of species characteristics have been covered here, suggesting that properly evaluated and planned translocation programs have a good potential for success in a wide range of species.

Translocation is a valuable tool for reducing the threats to many threatened species. However, the technique requires an in-depth, long-term commitment from land managers and can be expensive compared to many less intensive threat abatement techniques. Despite these issues, there are many threatened Australian bird species that have translocation proposed as a potential recovery action. These include the Yellow Chat (Dawson) (*Epthianura crocea macgregori*), Mallee Emuwren (*Stipitus mallee*), Western

Ground Parrot (*Prezoporus wallicus flaviventris*), Helmeted Honeyeater (*Lichenostomus melanops cassidix*), Southern Cassowary (*Casuarius casuarius johnsonii*) and the Regent Honeyeater (*Xanthomyza phrygia*) (Garnett and Crowley 2000). No doubt there are many other bird species with translocation earmarked as a potential recovery action. This increase in popularity of translocations in Australian conservation highlights the need to make sure these programs are planned well, that they investigate specific aspects of the species or techniques experimentally, critically assess success throughout the program and publish the outcomes of the program. This will benefit not only individual projects but more broadly this area of conservation.

Conclusion

This study indicates that translocation can be a useful tool in the conservation of the bristlebird and other Australian passerines. The establishment of the Beecroft population has the potential to spread the risk of extinction and has expanded the area of occupancy of the species. However, translocation is intensive. It can be expensive and it involves a lot of stress to individuals, possibly resulting in their death or impacts to the source population (although this was not the case in this program). It is recommended that translocation should not replace in situ conservation and threat abatement for species, which should continue regardless of any translocation programs. If translocation becomes the norm in conservation, then there is the potential that increased numbers of translocations may give rise to a whole new suite of conservation issues to manage in the host environment. These may include impacts to ecosystem dynamics, altered fire management, disease introduction and genetic issues. Already on Beecroft Peninsula, an issue of traffic management has arisen from a road fatality of a translocated bristlebird. It is recommended that traffic advisory signs be placed along the road to educate drivers.

This reintroduction has not finished. Ongoing surveys by staff at Beecroft Peninsula will monitor the reintroduced population. If the population keeps increasing as expected, there

will be an opportunity to investigate bristlebird population growth following disturbance, complementing previous work in this area (Baker 1997). In the event of a decline in the population, the monitoring may aid in understanding reasons for such an event. Furthering the understanding of population growth and recovery will be important for the future management of bristlebirds following large disturbances.

The criteria developed for sexing the bristlebird may be used to begin documenting the sex ratio in various populations of bristlebirds. This will have applications when investigating population declines and will be useful for future translocations of the species when attempting to establish additional populations in new areas.

The DNA sampled from the bristlebirds translocated in this study (stored with the School of Biological Sciences at the University of Wollongong) could be used to describe the genetic variation in the source population. Combining these results with DNA sampled from the descendants of the founding individuals may allow investigations into the incidence and severity of any inbreeding. This could be used to test the hypothesis that this reintroduction contained sufficient genetic diversity to establish a viable population without inbreeding effects. There has been little investigation into the genetic consequences of translocation, both in founding populations and source populations. The genetic material from the source population may also be used for a comparison with bristlebirds in all other populations to characterise the level of genetic diversity present amongst the isolated populations of this endangered species. This will be critical in the development of future translocation programs of the bristlebird and the ongoing captive breeding of the species.

Although outcomes of this translocation program have been interpreted as successful, limitations still exist. With increased funding, more rigorous monitoring strategies could have been adopted such as utilising automated remote surveying equipment and improved radio-tracking techniques. This may have given a better understanding of processes operating in the release population. More individuals could have been released, given the knowledge that the source population was not compromised with the current levels of

removal. The reduction of potential factors such as Allee effects brought about by such a low population density in the release environment may have been alleviated. It must also be acknowledged that this reintroduction was a trial, designed specifically to understand whether reintroduction could be used as a viable management technique for the eastern bristlebird. However, future translocations should not solely rely on the outcomes of this program but also be designed with appropriate monitoring and experimental techniques to further our understanding of this increasingly popular conservation technique.

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