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Recovery of the dibbler *Parantechinus apicalis* - review of progress and future research directions

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Recovery of the dibbler *Parantechinus apicalis* - review of progress and future research directions

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Cover image: Dibbler *Parantechinus apicalis* (© Perth Zoo)

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Summary

With no record of this small, carnivorous marsupial since 1904, the dibbler *Parantechinus apicalis* was presumed extinct until it was rediscovered in 1967 at Cheyne Beach on the south coast of Western Australia (WA), east of Albany. Between 1984 and 1988, populations were also discovered within the Fitzgerald River National Park (FRNP), on Boullanger and Whitlock Islands near Jurien Bay north of Perth, and Torndirrup National Park west of Albany. Currently, the FRNP, Boullanger and Whitlock Island populations remain the only known surviving indigenous populations. In parallel with the goal of conserving, discovering, and establishing additional dibbler populations, captive breeding for translocation has been critical in the recovery of the dibbler. The species' range has been extended, as have the number of known populations, and we now have a good understanding of the species' biology and ecology.

Despite the general lack of success of mainland translocations of the dibbler, two populations have been established on islands off the west (Escape) and south (Gunton) coasts of WA, and another on the mainland (Peniup). A third island population on Dirk Hartog Island in Shark Bay is in the early establishment phase. Dibbler numbers have fluctuated over time, with a declining trend observed on Boullanger Island, and more recently, Whitlock Island. The status of the mainland Peniup population is also uncertain. Although regular monitoring of the dibbler within the FRNP has been inconsistent in recent years, detection rates also appear to have declined. Managing each of these small, fragmented populations is likely to require ongoing intervention to maintain genetic diversity and promote long-term viability.

This review details the conservation management of the dibbler since the species' rediscovery (1967-2022), with particular emphasis on the scientific research and recovery actions that have been carried out by the Department of Biodiversity, Conservation and Attractions and other key stakeholders, with advice from the Dibbler Recovery Team. To support ongoing dibbler recovery, we identify some potential future research directions to assist with the conservation management of this species. These include: (1) improving genetic health through meta-population management; (2) selecting new translocation sites; (3) determining influences on population trends; (4) developing a robust monitoring strategy; (5) assessing climate change impacts and promoting resilience; and (6) understanding the influence of prescribed fire. We suggest that a separate structured decision-making process be considered to prioritise future research and conservation management actions that are most likely to benefit the dibbler.

1 Introduction

Conserving Australia’s unique mammal fauna is an ongoing challenge, with many of the human-induced threatening processes that have been implicated in the extinction of other species, continuing to persist. These drivers of decline, coupled with new threats such as climate change, are fuelling the loss of biodiversity at an unprecedented rate (Woinarski et al., 2015). Within Australia, a disproportionately high number of species extinctions have occurred since European settlement, with small-medium sized marsupials over-represented (Woinarski et al., 2019). Currently 61 species of terrestrial mammal are classified as threatened within Australia (IUCN, 2022). Given that 87% of Australia’s terrestrial mammal fauna are found nowhere else in the world (Woinarski et al., 2015), protection of Australia’s endemic species and their habitat is a key priority for conserving global biodiversity.

Aptly described as “a speckled-grey ball of action” (Friend, 1999), the dibbler *Parantechinus apicalis* (Gray, 1842) is a small (40-125 g) dasyurid marsupial, and like many other Australian mammal species, it has suffered a dramatic decline in distribution since European settlement. In fact, with no further records of the species after 1904, it was assumed to be extinct (Friend, 2004a). However, in 1967, the dibbler was rediscovered by chance, with two individuals captured during a survey for honey possums *Tarsipes rostratus* within the Arpenteur Nature Reserve near Cheyne Beach, east of Albany in Western Australia (WA; Morcombe, 1967).

The dibbler currently persists in small numbers (< 700 estimated mature individuals; Burbidge and Woinarski, 2016) within fragmented populations on the WA mainland and on five islands (Friend, 2004a) (Figure 1). Sub-fossil evidence suggests that the dibbler previously occupied a much larger area, extending from Shark Bay on the mid-west coast of WA to the Eyre Peninsula in South Australia, and inland to Peak Charles north of Esperance (Maxwell et al., 1996). Predation by introduced predators (red fox *Vulpes vulpes* and feral cat *Felis catus*), altered fire regimes, habitat destruction, and *Phytophthora* dieback disease have been implicated in the species’ decline (Maxwell et al., 1996; Woinarski et al., 2015). The dibbler is currently listed as Endangered under the Commonwealth *Environment Protection and Biodiversity Conservation Act 1999*, the Western Australian *Biodiversity Conservation Act 2016*, and by the International Union for Conservation of Nature (Burbidge and Woinarski, 2016).

Research to support the recovery of the dibbler began in 1995, with the implementation of a three-year research program by the then Department of Conservation and Land Management (CALM) (now the Department of Biodiversity, Conservation and Attractions; DBCA), supported by the Commonwealth Government’s Endangered Species Program (Start, 1997). A Dibbler Recovery Team was established in 1996 to assist with the coordination of dibbler recovery actions, and in 1998 a formal Interim Recovery Plan was put into effect (Start, 1998). Implementation of a full Recovery Plan commenced in 2004 (Friend, 2004a). Recovery actions focused on monitoring, protecting and searching for new populations; captive breeding to provide stock for translocation; the establishment of additional mainland populations; genetics to inform population management; and facilitating community involvement in recovery work (Friend, 2004a). Further research including dietary analysis (Lerch, 2015; John, 2018), disease investigation (Mathews et al., 2006; Moore, 2019; Bowry, 2021) and genetic studies (Mills and Spencer, 2003; Mills et al., 2004; Aisya, 2018; Thavornkanlapachai et al., 2019, 2021) has since been undertaken to improve knowledge to underpin dibbler recovery.

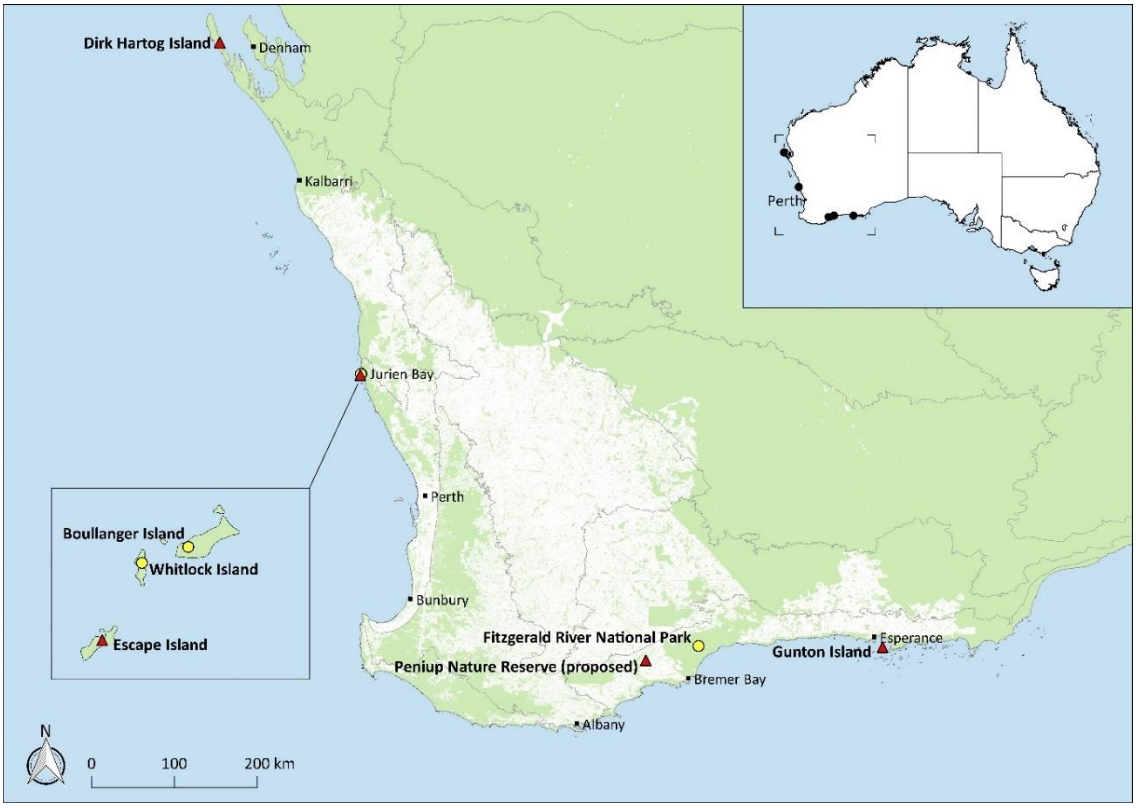

Figure 1 Location of known dibbler subpopulations within Australia; indigenous populations shown as yellow circles, translocated populations as red triangles. Remnant vegetation and Interim Biogeographic Regionalisation for Australia (IBRA) boundaries have been included for reference.

In 1997, a captive breeding program was established at Perth Zoo to provide founders for translocations (Appendix 1). Within a year of commencing this program, captive-bred dibblers were translocated onto Escape Island (10.5 ha) near Jurien Bay, 200 km north of Perth, with supplementations in 1999 and 2000 (Moro, 2003; Figure 1). In 2001, the first translocated population of dibblers was established on the mainland within the proposed Peniup Nature Reserve (6530 ha), 150 km north-east of Albany (Friend, 2001a). Additional mainland translocations to the Stirling Range National Park (SRNP) (Friend, 2004b), Waychinicup National Park (Waychinicup enclosure; Friend, 2010b), and Whiteman Park (Mawson, 2014) were conducted between 2004 and 2014, however these were unsuccessful (Moro and Friend, 2018). In 2015, a second new island population was established on Gunton Island (90 ha) in the Archipelago of the Recherche, west of Cape Le Grand near

Esperance (Friend, 2015). In 2019, dibblers were released onto Dirk Hartog Island (DHI; 63,300 ha) near Shark Bay, 730 km north of Perth (Moro and Friend, 2018).

This review summarises the research and conservation actions that have been carried out by DBCA and other stakeholders to recover the dibbler since the species was rediscovered. We also identify some potential future research directions to assist with the ongoing recovery of the dibbler.

2 Road to recovery

2.1 Monitor known populations

Dibblers are currently known from wild indigenous (FRNP, Boullanger Island and Whitlock Island), wild translocated (Escape Island, Peniup, Gunton Island, and DHI) and captive (Perth Zoo) populations (Figure 1). Monitoring of each wild population has been carried out routinely via trapping and/or the use of camera traps (Moro and Friend, 2018). Standardised dibbler monitoring was first initiated on the mainland within the FRNP in 1997. Twice-yearly trapping (autumn and spring) has been carried out (weather and resources permitting) along two permanent monitoring transects in the eastern (Moir Track) and western (Twertup Creek) FRNP as part of DBCA's *Western Shield* program (Friend, 2001a), but is now confined to the Twertup transect only. Additional monitoring transects and/or grids have since been established to monitor other sites (e.g., Hamersley-Moir trapping grid used for a population dynamics study in the eastern FRNP; Friend et al., 2007). Detection rates at sites within the FRNP appear to have declined. Within the western FRNP for example, dibbler captures have been low since a wildfire in 2008, and only two individuals have been trapped since autumn 2018 (unpublished data). Lack of captures in the past few years have coincided withbelow average rainfall (BOM, 2021b).

Other reintroduced mainland populations have likewise been monitored regularly using permanent monitoring grids (e.g., Peniup; see Friend and Utber, 2017), and camera traps have been useful for detecting the presence of dibblers and for monitoring predator activity (see section 2.2.1). Extensive camera monitoring has been conducted at Peniup, with baited camera traps deployed to complement routine Elliott trap monitoring (Friend and Button, 2019). The Peniup population has fluctuated over time, with numbers dropping to undetectable levels on at least two occasions (2008 and 2015; Friend and Utber, 2017). Periodic supplementations may have assisted in maintaining this population, though dibblers have not been captured in traps since autumn 2018 (Friend and Button, 2018) and the last known sighting on remote camera was in November 2019 (Thavornkanlapachai et al., 2021).

Dibbler populations on Boullanger and Whitlock Islands have also been monitored routinely (weather permitting) since 2005 with biannual surveys in autumn and spring (Friend and Button, 2016). Dibbler monitoring prior to this was variable and largely dependent on research work (e.g., bimonthly monitoring between 1997 and 1999 to examine male die-off; Mills and Bencini, 2000). Dibbler populations on Boullanger and Whitlock Islands have fluctuated over time (Friend, 2016). Since 2012, capture rates on Boullanger Island have shown a decreasing trend and numbers have remained low (Figure 2). A steady decline has also been observed on Whitlock Island since 2017 (Figure 3). Most recently, evidence of breeding, in conjunction with a high proportion of subadults captured, indicates some recovery of both populations, most likely in response to the significantly higher rainfall over winter 2021 (Friend, 2022).

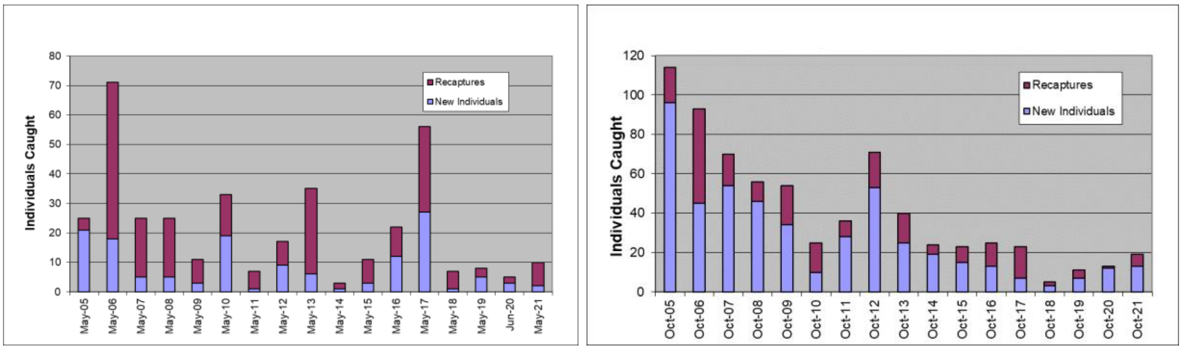

Figure 2 Number of individual dibblers captured on Boullanger Island during each monitoring period

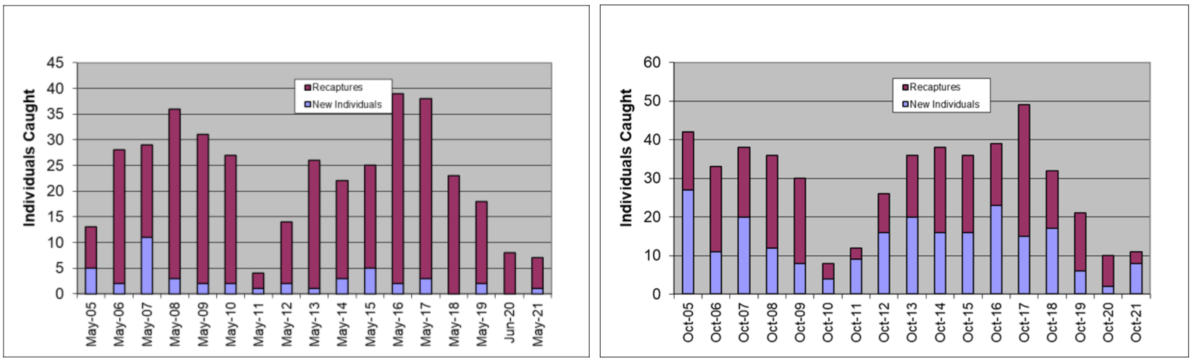

Figure 3 Number of individual dibblers captured on Whitlock Island during each monitoring period

Biannual monitoring was also planned for Gunton Island, however with adverse sea conditions often preventing island access, remote cameras have predominantly been used to monitor dibblers on the island (Friend, 2015). On Escape Island, adverse sea conditions or lack of resources have also limited routine monitoring (Moro and Friend, 2018); trapping protocols have also varied (e.g., timing and trap setting protocols; Friend, 2021a). On DHI, an intensive post-release monitoring plan was established prior to translocation, incorporating biannual trapping (autumn and spring) and the use of remote cameras to map extent of occurrence (see Moro and Friend, 2018). Due to the large size of the island and relatively small number of dibblers released however, there has been difficulty recapturing and detecting dibblers post-release (Cowen et al., 2020); though the use of ‘delayed release’ techniques (see Section 2.7), has improved post-release monitoring.

2.2 Protect known populations from threatening processes

Recovery of the dibbler is heavily reliant on habitat protection and mitigation of key threats (Friend, 2004a). On the mainland, dibbler habitat is characterised by dense vegetation, particularly mallee/*Banksia* spp. over heath up to one metre, which has been unburnt for at least 10 years (Baczocha, 1997; Moro and Friend, 2018). On islands, dibblers likewise prefer dense vegetation with a high percentage of cover (McCulloch, 1998). Several threatening processes including predation by introduced predators, and habitat loss through wildfire and *Phytophthora* dieback disease, are known to adversely impact dibbler populations (Friend, 2004a). Other potential threats such as competition with house mice *Mus musculus* on Boullanger and Whitlock Islands (Bennison et al., 2016), and climate change (Aisya, 2018; Moro and Friend, 2018) have been identified.

2.2.1 Predation by introduced predators

On the mainland, the introduced red fox is controlled by routine aerial and ground baiting using 1080 poison (sodium fluoroacetate; *Probait*®) as part of DBCA’s *Western Shield* program (Morris, 1998; Friend, 2004a). Within the FRNP, routine baiting has been undertaken since 1996 (Baczocha and Start, 1996). Regular fox control using 1080 was also implemented within Peniup and the adjacent Corackerup Nature Reserve at the inception of *Western Shield* (Friend, 2001a). In 2010, DBCA’s South Coast Region began investigating landscape-scale baiting with *Eradicat*® for the management of feral cats within the FRNP (Comer et al., 2020). Following non-toxic rhodamine trials to ensure dibblers did not consume baits (Friend, 2010a), annual aerial *Eradicat*® baiting commenced (Comer et al., 2020). In 2021, *Eradicat*® baiting was also initiated within Peniup (and Corackerup Nature Reserve) as part of Bush Heritage’s collaborative Fitz-Stirling Fauna Recovery Program (M. Drew pers. comm.; Hams, 2021). Targeted feral cat control has also been undertaken as required (e.g., strategic leg-hold trapping at Peniup coinciding with dibbler supplementations; Friend and Utber, 2017). Sand pads and remote cameras have been used to monitor predator activity and to guide targeted predator control (e.g., Waychinicup enclosure; Friend, 2010b). While introduced predators do not inhabit islands where dibblers are known to occur, monitoring for invasion is undertaken to prevent their establishment. This is particularly important for Boullanger and Whitlock Islands, given their proximity to the mainland and frequent visitation by the public (Friend, 2004a).

2.2.2 Wildfire

Dibblers have typically been associated with long-unburnt vegetation (16+ years; Chapman and Newbey, 1995), however, under a regime of fox control in the FRNP, dibblers have also been detected within vegetation only seven years post-fire (Baczocha, 1997). The FRNP Management Plan aims to protect long-unburnt vegetation via the strategic separation of cells using low-fuel buffers, which decrease the likelihood of large-scale wildfires (Moore et al., 1991). Peniup is likewise maintained in a largely long-unburnt state (Friend, 2004a), although to achieve this, strategic fuel reduction burns have been used (DBCA, 2020). Management of dibbler habitat in a heavy fuel-load state is not without risk. In January 2008, a wildfire burnt approximately 50,000 ha of the western FRNP, and dibbler capture rates declined significantly during subsequent *Western Shield* monitoring in this area (Friend and Button, 2009) and have remained low since (Friend, 2021a). Fire management of dibbler habitat on islands currently excludes fire and prohibits the deliberate lighting of fires. Compared with the mainland, the frequency and risk of fires on islands is considered low (CALM, 2001).

2.2.3 *Phytophthora* dieback

Mainland dibbler habitat is highly susceptible to dieback disease, principally caused by the soil-borne water mould *Phytophthora cinnamomi* (Miller and Dunne, 2005). Dieback disease is considered to be the greatest management concern within the FRNP, given the ability of *Phytophthora* spp. to alter the floristic structure and composition of vegetation (Moore et al., 1991). Changes to the understorey and litter layer, plant diversity and/or vegetation density, and the availability of food and/or shelter, are likely to negatively impact dibbler survival (Baczocha, 1997; CALM, 1999). Strict hygiene measures have been implemented within the FRNP to prevent the spread of disease, which currently impacts a relatively small area of the park (Moore et al., 1991; Friend, 2004a). Unfortunately, inaccessibility due to wet soil conditions can limit dibbler monitoring (Friend et al., 2007).

Peniup is considered *Phytophthora*-free, though monitoring for its presence (i.e., soil and plant tissue sampling) is routinely undertaken (Friend, 2004a). The heavily infected state of the SRNP (CALM, 1999) was considered a threat to the dibbler translocation at this location (Friend, 2004b) and may have contributed towards its failure. Hygiene restrictions under wet soil conditions disrupted routine ground baiting to control introduced predators (Friend and Button, 2005).

2.2.4 Competition with introduced house mice on islands

Competition between dibblers and house mice on Boullanger and Whitlock Islands has been the subject of past investigation. In 2007, a workshop held by the Dibbler Recovery Team agreed that the eradication of mice from the islands was desirable (Bennison, 2011). Dibblers and mice exhibit some dietary overlap (Stewart, 2006) and mouse removal experiments resulted in increased invertebrate biomass and litter layer depth (Dickman, 1999), and higher juvenile dibbler survival in mouse removal plots (unpublished data). Mice also pose a disease risk to native species (DEWHA, 2009). Experimental studies sought to determine the feasibility of eradicating mice from the Jurien Bay islands using poison baits (Bennison et al., 2016). Unfortunately, dibblers readily consumed non-toxic 10 mm rhodamine B baits (48% uptake on islands, 90% uptake in captivity). Evidence of bait consumption was also detected in other non-target species (e.g., King’s skinks *Egernia kingii*). Whilst poison baiting had the potential to eliminate mice from the islands (92% bait acceptance in mice; Bennison et al., 2016), baiting was deemed too hazardous for non-target species, including dibblers; though the temporary removal of dibblers from the islands could enable bait application (Bennison et al., 2016).

Subsequent trials using smaller 5.5 mm baits resulted in 100% bait uptake by mice on both islands, with reduced consumption by dibblers (12% uptake on Boullanger Island, 0% uptake on Whitlock Island, 0% uptake in captivity; Friend and Button, 2017b). A dibbler was however, observed taking a dead mouse on camera (Bennison et al., 2016), thus secondary poisoning via consumption of poisoned mice carcases was considered a risk (Friend and Button, 2014). As dibblers did not consume baits on Whitlock Island, and only one skink species likely to eat mouse baits (*Liopholis pulchra longicauda*) is known to inhabit the island, the Dibbler Recovery Team recommended that mouse eradication on Whitlock Island should be considered pending a risk assessment (DBCA, 2020). However, reduced dibbler captures coinciding with below average rainfall in recent years (BOM, 2021a; Friend, 2021a), may render any attempts to eradicate mice too high a risk on the dibbler population. A risk assessment would help to inform this approach, particularly if removing native species (including the dibbler) from the islands prior to baiting.

A project investigating genetic biocontrol of invasive rodents (GBIRd; https://www.geneticbiocontrol.org) considered the use of gene drive technology as an eradication method for mice on Boullanger and Whitlock Islands (Friend, 2018). However, given their close proximity to the mainland there was concern that “gene drive mice” may not be adequately contained (Wilson, 2018).

2.2.5 Climate change

With increasing temperatures and reduced winter/spring rainfall projected for Jurien Bay (Hope et al., 2015), and rising water levels and/or extreme sea-level events likely to reduce the carrying capacity of the islands (CoastAdapt, 2018), climate change may have a significant impact on island dibbler populations. Within 100 years, it is predicted that Boullanger and Whitlock islands will be reduced to two small, isolated islets; Boullanger Island is at greatest risk given its dominantly undifferentiated soft sediment shores (Clapin and Doak, 2018). Erosion is already evident at the eastern end of Boullanger Island (Friend, 2021b). Escape Island is also predicted to erode over time, but to a lesser degree (Clapin and Doak, 2018).

Population viability analyses undertaken by Aisya et al. (2022) indicated that drought (and reduced carrying capacity) has the potential to significantly reduce dibbler survival probability. Reduced dibbler captures have coincided with below average rainfall in recent years (BOM, 2021a; Friend, 2021a). According to Stewart (2006) dietary overlap between dibblers and mice on islands was evident and estimated biomass consumption was greater in mice compared with dibblers at most times of the year. Seasonal changes in rainfall regulated the level of competition between dibblers and mice on islands, with cool, wet winters offsetting competition between the two species (Stewart, 2006). A warming climate, however, is likely to improve mouse survivorship over the winter, and prolong their breeding season, favouring higher mouse densities.

2.3 Surveys to locate new populations

2.3.1 South coast

Since the opportunistic capture of two dibblers at Cheyne Beach in 1967, only seven more dibblers were captured within this small patch of *Banksia*-dominated heathland despite numerous surveys between 1970 and 1981 (Ride, 1970; Woolley, 1977; Woolley, 1980; Hopper, 1980; Woolley and Valente, 1982). A single dibbler was trapped near this locality in 1994 (Baczocha, 1997). After two deceased dibblers were found on properties in Jerdacuttup between Ravensthorpe and Hopetoun, multiple surveys were conducted in uncleared bushland adjacent to farmland between 1976 and 1978, however no dibblers were captured (Woolley, 1977; Woolley, 1980). Based on observations that habitat within the FRNP may support dibblers (Morcombe, 1969), trapping was also conducted at four sites within the southwest corner of the FRNP in 1978, but no dibblers were detected (Woolley, 1980). In 1984 the discovery of a deceased dibbler on Hammersley Drive (Muir, 1985), prompted a large-scale biological survey of the FRNP between 1985 and 1987, during which 17 dibblers were captured at eight different sites (Chapman and Newbey, 1995). Since then, dibblers have been recorded from several other sites within the park (Figure 4), however their occurrence is highly dynamic.

Three dibblers were captured in the Torndirrup National Park between 1987 and 1988 (Smith, 1990), but as no dibblers have been detected since, this population is presumed to be extinct (Baczocha, 1997; Friend, 2004a). Surveys within Cape Arid National Park during the early 1990s also failed to detect dibblers (Start, 1997). Further surveys were undertaken between 1995-96 based on sub-fossil evidence of the dibbler’s past distribution, and the species distribution modelling software BIOCLIM to predict suitable habitat, but new sites were only detected within the FRNP (Start, 1997). Sporadic surveys for new populations have since been undertaken, including in south and west coast conservation areas between 2006 and 2008 using Faunatech hair funnels, which again led only to the discovery of new sites within the FRNP (Friend et al., 2007).

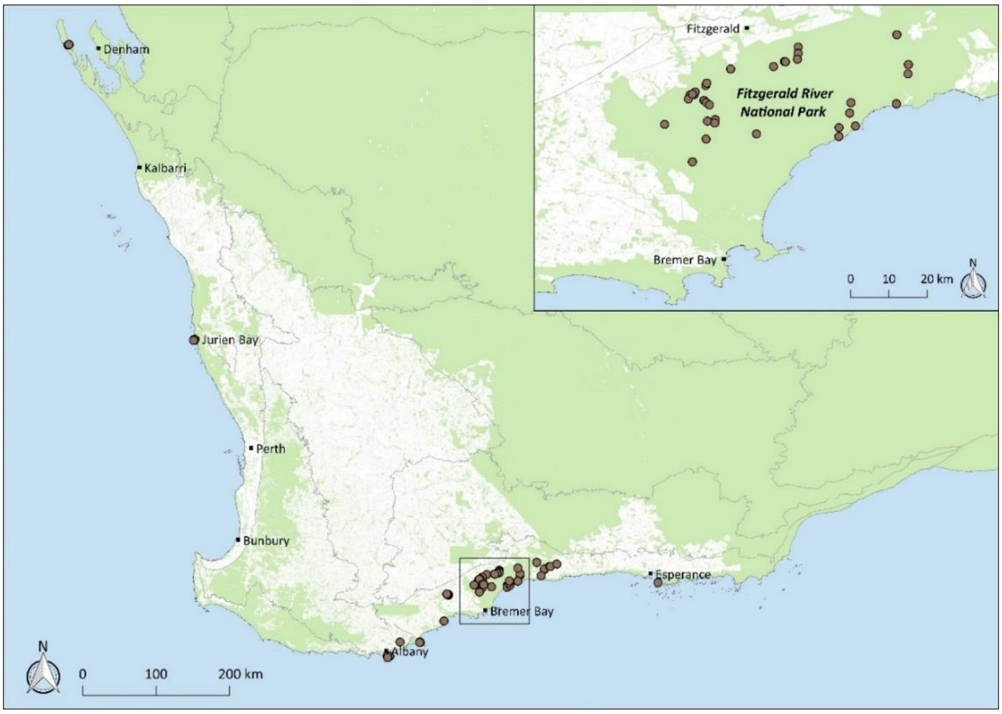

Figure 4 Map depicting reported dibbler sightings between 1967 and 2022. The inset (top right) shows locations within the Fitzgerald River National Park. Remnant vegetation and Interim Biogeographic Regionalisation for Australia (IBRA) boundaries have been included for reference.

2.3.2 West coast

In 1985, during an independent CALM (now DBCA) biological survey of islands between Lancelin and Dongara, small mammal tracks were detected on Boullanger Island off Jurien Bay. Follow-up trapping on Boullanger, Whitlock, Escape and Favourite Islands led to the discovery of dibblers on Boullanger and Whitlock Islands (Fuller and Burbidge, 1987). Large numbers of introduced house mice were also detected on both islands, as was the grey-bellied dunnart *Sminthopsis griseoventer* on Boullanger Island (Fuller and Burbidge, 1987). Surveys on the adjacent mainland around Mount Lesueur National Park and nearby coastal areas of Jurien failed to detect the species (D. Moro pers. comm.).

2.4 General biology and ecology

Initial field observations were documented by Morcombe (1967) following the species' rediscovery, and dibbler habitat was described as 'populations were discovered' (e.g., Muir, 1985; Chapman and Newbey, 1995). By studying dibblers in captivity, Woolley (1971, 1991) provided the first insight into dibbler reproductive biology (see below), and Lynam (1987) assessed inbreeding and juvenile dispersal on islands. Dickman (1986) was the first to investigate dibbler population dynamics on islands. Between 1986 and 1989 population size fluctuated from year-to-year and between seasons on Boullanger and Whitlock Islands, with maximum numbers observed in September/October following juvenile dispersal (Dickman and Lynam, 1989). A study on the microhabitat use and foraging behaviour of dibblers on Boullanger Island indicated dibblers tended to forage for prey > 2 mm on the litter surface or in trees (Dickman, 1988).

Further studies focused on monitoring known populations within the FRNP to investigate population dynamics and habitat use. Initial radio-tracking studies proved to be largely unsuccessful due to the low density and high mobility of dibblers, as they frequently moved beyond the short tracking range of transmitters (Baczocha and Start, 1996; Start, 1997). Greater success was achieved in a subsequent study, which investigated the movements and social organisation of FRNP dibblers using a combination of radio-tracking and trapping (Friend, 2001b). Despite difficulties with detection of radio-collars from an aircraft (i.e., due to whip antennas breaking off), intensive ground-based radio-tracking showed that dibblers occupied discrete home ranges within long-unburnt vegetation and had very restricted crepuscular activity (see section 2.6). Low recapture probability was also evident (Friend, 2001b).

From 1997, additional studies on the Boullanger and Whitlock Island populations investigated abundance and potential threats, and identified important resources (Start, 1997). Cheng (2000) estimated dibbler population size on Boullanger and Whitlock Islands between 1997 and 1999. While poor model fit limited interpretation of the results, females on Boullanger Island showed a significant decline over time. Demography and habitat use was studied in detail by McCulloch (1998), who found that Whitlock Island supported a higher density of dibblers with a larger proportion of older animals compared to Boullanger Island. Both populations had a bias towards females. Dibblers were sexually dimorphic (i.e., males were significantly larger than females) and Boullanger Island dibblers were significantly larger than Whitlock Island dibblers. Compared with mainland dibblers, island dibblers were smaller in size and had significantly smaller home ranges. Greater plant species richness and a larger number of seabird burrows were observed on Whitlock Island. Dibblers utilised seabird burrows for foraging and as a refuge (McCulloch, 1998).

A link between nesting seabird density and resource availability for dibblers on islands was subsequently identified, whereby seabird density was positively related to soil nutrient levels, which was in turn, positively related to plant productivity and thus abundance of invertebrate prey (Wolfe, 2004; Wolfe et al., 2004; Stewart, 2006).

2.5 Reproduction and captive breeding

Woolley (1971) determined that dibblers were monoestrous, breeding in autumn, with a gestation period of about 44-53 days. Young reached sexual maturity by 10-11 months of age, and both males and females survived to breed the following year. Between 1986 and 1988, complete and synchronous post-mating mortality was reported in male dibblers on Boullanger Island (Dickman and Braithwaite, 1992). A subsequent comparative study of Boullanger Island dibblers in captivity (Woolley, 1991) found that males survived the mating period. As mainland dibblers had also been reported to survive the breeding season (Fuller and Burbidge, 1987; Smith, 1990), it was concluded that post-mating mortality was not an obligatory event for the dibbler. In a further study on island dibblers, Mills and Bencini (2000) proposed a new category of life history strategy '*Strategy VII - facultative male die-off*' for species that display complete and synchronous male die-off in some populations in some years. To date the dibbler is the only species in this category.

Following the establishment of a captive breeding colony at Perth Zoo in 1997, knowledge of dibbler reproductive biology was further refined. In collaboration with CALM (now DBCA), the University of Western Australia, and the Marsupial Cooperative Research Centre, and in accordance with the Interim Recovery Plan, Perth Zoo embarked upon a comprehensive research program, focusing on the development of captive husbandry techniques to breed dibblers in captivity (Bradley et al., 1999). A complete series of morphometric measurements were obtained from offspring born between 1997 and 1998 to characterise the kinetics of pouch young growth and development (Mills et al., 2000). The mating behaviour of captive island dibblers was studied by Wolfe et al. (2000) and was found to be similar to other dasyurid species. A form of “silent oestrus”, in which small peaks in mating behaviour coincided with minor peaks in cornified epithelial cell counts during pro-oestrus, was detected in females (Wolfe, 2000). The gestation period of mainland females was calculated to be 45 days, a week longer than island females (38 days). Mainland females also entered oestrus 12 days earlier, and for five days longer, than island females (Mills et al., 2012). Males were spermatorrhoeic in February and early March, with peak testis volume to body weight ratio identified immediately prior to copulation (Mills et al., 2012). When the captive colony was first established, dibblers were paired with multiple partners, however genetic studies (H. Mills; unpublished data) to verify whether dibblers exhibit multiple paternity were inconclusive (Lambert, 2017).

Using island-caught founders, early success of the breeding program was evident with three of the four females giving birth and 19 young raised to independence during the first year (Start, 1997). Between 1998 and 2000 captive breeding and husbandry methods were refined, with a transition to mainland founders in 2000 (Lambert, 2020). While reproductive output has varied over the years (i.e., reduced breeding output in some years, and surplus animals in others), the captive breeding program has been regarded as highly successful (Friend, 2017). Perth Zoo staff have continually sought to optimise captive breeding techniques to maximise the health and reproductive potential of captive dibblers and reduce the need for replacement stock. Research to determine breeding longevity for example, showed that dibblers could breed for a second season, but fecundity was poor in 3-year-old females (Lambert, 2020).

Captive husbandry techniques have also been modified in response to reproductive-related morbidity and mortality events including cannibalism, mating-associated injuries, and mastitis. After three captive-born females cannibalised their entire litters in 1998, the adoption of measures to reduce stress (see Lambert, 2000), significantly lowered the rate of cannibalism thereafter. While mating-associated injuries are not uncommon, aggression due to incompatible pairings can lead to injury. Ensuring males are larger than females has reduced the likelihood of male injury due to incompatibility (Lambert, 2000; Wolfe et al., 2000). Likewise, to reduce the incidence of mastitis, a weaning protocol was implemented whereby half of the young are weaned initially, with a 4-5-day interval before weaning all but one of the second half of the litter, and then a few days after that, the last offspring is weaned away (Lambert and Frost, 2013); the last documented case of mastitis was in 2014.

To support the establishment of dibblers on Dirk Hartog Island, the captive breeding program again transitioned from mainland to island stock in 2019. However, reproductive output fell below expectations with only 50% of breeding pairs producing young in 2019 and 2020 (Lambert, 2020). Of those that did not produce young (n = 9), mating was not observed for six of these pairs. In 2020, Perth Zoo conducted a comparison of reproductive parameters between island and mainland provenance dibblers to identify potential causes of poor reproductive output. In 2020, three females had atypical oestrus events (i.e., two did not enter oestrus and the other came into oestrus a month early). Failure of females to enter oestrus has been documented in the past (e.g., 2004 and 2008) with no identifiable underlying cause (Lambert, 2020). There was a correlation between lack of observed matings and the absence of young in 2019 (3 out of 5 pairs) and in 2020 (3 out of 4 pairs). In collaboration with the University of Queensland, a dibbler fertility index is currently being developed through the evaluation of spermatorrhoea characteristics to determine if there is a male factor in breeding success (P. Mawson pers. comm.).

As dibblers are presumed to have a polygynandrous mating system, with both males and females having multiple partners during the breeding season

(Thavornkanlapachai, 2016), females were given access to two males rather than a single mate choice and paired with animals from different island founding stock to encourage mating during the 2021 breeding season (Lambert, 2020). Encouragingly, 70% of breeding pairs successfully produced young, though only 3/7 females produced the full complement of eight young, and one of these litters was cannibalised (Mantellato, 2021). During the 2022 breeding season, females were offered one of two males, alternating every 24-72 hours (L. Mantellato pers. comm.).

2.6 Behaviour and physiology

In 2009, physiological trials examined the thermal, metabolic, hygric, and ventilatory physiology of eight captive dibblers during their presumed inactive phase (i.e., day) (Withers and Cooper, 2011). It was expected that the dibbler would use torpor to conserve energy and reduce water requirements like other dasyurids (see Geiser and Körtner, 2010), however results indicated that dibblers were typical endotherms with no tendency for torpor. A subsequent study, which expanded upon this research by monitoring captive dibblers with infra-red motion detectors above enclosures and temperature loggers in nest-boxes (Bruning, 2010), identified consistent bouts of torpor at night. Dibbler activity appeared to be influenced by the activity of keepers (e.g., exiting nest boxes long after sunrise to forage), rather than following natural crepuscular rhythms, where foraging is confined to two short periods around dawn and dusk (Friend, 2001b). Comparative tracking studies of wild dibblers showed that all radio-collared FRNP dibblers were inactive by 10am. Within Peniup, 50% of dibblers (captive-bred origin) were still active at this time, with others observed out in the open during the day (Friend, 2001c). If dibblers develop atypical behaviour and lose their natural rhythms of activity in captivity (i.e., become accustomed to diurnal activity), there is concern it may negatively impact survival post-release (e.g., increased exposure to predators).

The effects of behavioural and physiological traits on reintroduction success of dibblers were investigated by Kealley (2016). Field release behaviour (i.e., movement) was monitored in 12 captive-bred subadult dibblers released at Whiteman Park, and a correlation between elevated beam behaviour (a measure of anxiety) and field movements was identified (i.e., calmer dibblers moved greater distances post-release). Dispersal distances (252 ± 55m; range 7-648 m) were consistent with those reported in dibblers released onto Escape Island (Moro, 2003). Kealley (2016) also identified a positive relationship between male parents and offspring, such that the behaviour of male parents was a useful indicator of open field behaviour (i.e., boldness) in offspring. Further assessment of behavioural criteria such as these, may be useful for predicting post-release survival and optimising the breeding and selection of individuals for translocation.

2.7 Translocations

To date, dibblers have been translocated to seven sites (Appendix 1); all except Escape and Gunton Islands are considered reintroductions. Translocations have typically involved captive-bred stock, however wild-to-wild translocations have occasionally been undertaken.

Attempts to establish mainland populations of dibblers via translocation have had little success, due to various factors including predation, insufficient food resources and wildfire. Following the translocation to the Stirling Range National Park (SRNP), recapture rates were extremely poor. Only four (out of 57) dibblers were recaptured after the 2004 release (Friend, 2010b). Two radio-collared dibblers released in January 2005 were predated by birds within 10 days of release (Friend, 2010b). Low recapture rates were also reported following the translocation of dibblers into the Waychinicup enclosure and Whiteman Park. At Waychinicup, 4 out of 6 collared dibblers were found dead post-release; two were predated by native fauna (raptor and heath goanna *Varanus rosenbergii*), and two died apparently of starvation (Moro and Friend, 2018). While there was evidence of breeding, with at least two generations identified, no dibblers have been captured at Waychinicup since July 2014 (Moro and Friend, 2018). Two months after dibblers were released into Whiteman Park, 60% of the park was burnt by a wildfire. Despite a second release into habitat unaffected by fire in 2015 and some dibblers being detected on remote cameras later that year, none have been trapped since. A study evaluating invertebrate availability (see Lerch 2015; Section 2.8) within the SRNP and Waychinicup enclosure questioned the adequacy of the abundance and diversity of invertebrates at these sites to support dibblers.

Within Peniup, following the first dibbler release in 2001, mortality rates of collared individuals were high. Of the 23 individuals collared, only 2 out of 17 whose fate was known survived the ten-week lifespan of the radio-collars (Moro and Friend, 2018), with 11 individuals predated by native birds and one taken by a fox or feral cat. Another dibbler died after its forelimb was caught in its collar and two were found dead with no apparent cause (Friend, 2001c; Friend, 2010b). Overall, wild-born founders had better survival than captive-bred progeny (Friend, 2001c). No further radio-tracking was used to monitor dibblers in subsequent releases due to concern that collars may compromise their host. A review of survival data between 2001 and 2009 indicated that recapture rates of released animals were low (i.e., 12-21%; Friend, 2010b). Access into Peniup to apply introduced predator control has been problematic due to wet weather and the lapse in winter ground baiting in 2005 is thought to have contributed to the population crash observed at this site in 2006 (Thavornkanlapachai et al., 2021).

Access to Escape and Gunton Islands is often limited, and while traps have been useful to estimate trap success and abundance on small islands (Moro, 2003) they require daily attendance. Cameras have been useful for detecting dibbler presence and dispersal across islands in the absence of trapping. For example, dibblers have been detected regularly by cameras deployed across Gunton Island (e.g., Friend and Button, 2017a; Friend, 2022).

Due to the large size of Dirk Hartog Island (DHI), and the relatively low number of dibblers released, monitoring to determine translocation success has proved challenging (Cowen et al., 2020). Since dibblers were first released in 2019, only two uncollared individuals have been recaptured during trapping in 2020 and 2021 (Cowen et al., 2020; Cowen et al., 2021); though dibblers have been detected on remote cameras within the release area on five occasions (S. Cowen pers. comm.). High native rodent abundance (*Pseudomys* spp.) may have reduced trap success (Cowen et al., 2020). In response to the poor success of radio-tracking in 2019 (i.e., collar slippage), an alternative design (similar to Sims et al., 2020) was trialled at Perth Zoo, and in the field in 2020 (Cowen et al., 2021). This resulted in a small improvement in attachment duration but still only one of seven collars (in part due to transmitter failure) was retrieved from a live animal 13 days post-release (Cowen et al., 2021).

To improve release site fidelity and monitoring efficacy in the short-to-medium term on DHI, a trial 'delayed release' incorporating individual open-air pens and artificial refuges (i.e., nest boxes) with supplemental feeding was undertaken in 2021 (S. Cowen pers. comm; Cowen et al., 2021). Preliminary results suggest an improvement in post-release monitoring, with evidence that at least 50% of the release cohort remained within the release area for at least 10 days following release (S. Cowen pers. comm.). Ongoing camera monitoring has demonstrated a significant increase in dibbler detections in the last nine months (S. Cowen pers. comm.). Trap conditioning is also being trialled at Perth Zoo to enhance re-capture probability (L. Mantellato pers. comm.). If effective, such methods could also be applied during any future mainland translocations to improve the likelihood of populations establishing within suitable habitat.

2.8 Diet

Examination of a dibbler specimen by Gray (1842) provided the first insight into the diet of dibblers, with evaluation of the stomach contents revealing predominantly insects, in particular small Coleopteran species (Whittell, 1954). Early captive studies reported that dibblers readily consumed *Banksia* nectar, insects, spiders, and raw meat (Morcombe, 1967). Subsequent studies, which examined the contents of faeces, confirmed that dibblers are predominantly insectivorous (Dickman, 1986; Fuller and Burbidge, 1987; McCulloch, 1998; Bencini et al., 2001; Miller et al., 2003; Rawlinson, 2003), though plant material (e.g., *Rhagodia baccata* berries; Bencini et al., 2001) and other small vertebrates including reptiles (Dickman, 1986; Bencini et al., 2001), birds (Dickman, 1986; Fuller and Burbidge, 1987; Bencini et al., 2001) and mice (Dickman, 1986; Stewart, 2006), have been documented. While several studies have found no evidence that dibblers actively feed on mice (Fuller and Burbidge, 1987; McCulloch, 1998; Bencini et al., 2001; Miller et al., 2003), both Dickman (1986) and Stewart (2006) reported the presence of mouse remains in faeces during pitfall trapping. The consumption of other vertebrates is more likely to be opportunistic (e.g., dibblers have been found alongside partially ingested mice in pitfall traps; Stewart, 2006). The ability of dibblers to vary their diet and select food based on availability (e.g., seasonal consumption of *R. baccata* berries when abundant; Bencini et al., 2001) categorises dibblers as opportunistic feeders (Miller, 2000; Miller et al., 2003; Rawlinson, 2003). The captive colony at Perth Zoo are offered a varied diet including invertebrates, Proteaceae and Myrtaceae blossoms, meat, fruit (including *Rhagodia* sp.), boiled egg, small carnivore mix and cat biscuits; the proportion of invertebrates is increased during the breeding season (Lambert, 2000).

While it is evident that invertebrates, in particular arthropods, form an integral part of the dibbler diet, the importance of invertebrate abundance was not immediately apparent. Lerch (2015) collected standardised leaf litter samples from four mainland sites and compared the abundance, size and diversity of invertebrates over 2 mm in length and found a strong correlation between site and invertebrate diversity and abundance. Within the FRNP, invertebrate abundance and diversity was greatest, followed by Peniup, Waychinicup and SRNP. Invertebrate abundance and diversity were significantly lower at both failed translocation sites (Waychinicup and SRNP) than in the FRNP. Prior to the translocation of dibblers to DHI, the diversity and abundance of ground-dwelling macro-invertebrates was determined, including temporal and spatial variation (John, 2018). Pitfall trapping and leaf-litter sampling detected 21 orders from seven classes of invertebrate, with a predominance of Hymenoptera taxa. Up to 24 orders of invertebrates were previously identified from pitfall traps and leaf-litter samples on the Jurien Bay islands (Stewart, 2006). Thus, from a dietary perspective, it was inferred that DHI had a suitably diverse invertebrate community that could support a self-sustaining dibbler population (John, 2018).

2.9 Disease

The incidence of disease in dibblers has been low with most reports originating from dibblers maintained in captivity, though comprehensive disease screening of wild populations has not been undertaken. Overall, there is a paucity of knowledge regarding health and disease in dibblers compared with other dasyurid species (Moore, 2019). The infectious agents identified in dibblers and other dasyurids have been summarised by Moore (2019). Appendix 2 lists the infectious and non-infectious agents recorded in dibblers.

Demodex spp. mite infection is the most prevalent pathological disease reported in dibblers. Demodicotic lesions i.e., erythematous skin nodules consisting of predominantly adult demodex mites, larvae and eggs (Lambert, 2000) with or without alopecia and crusting (Bowry, 2021) which typically affect the eyelids and snout (Eden et al., 2004; Bowry, 2021), have been documented in captivity (Lambert, 2000), with a few presumed cases (i.e., based on clinical appearance) identified in the wild. Perth Zoo recorded 13 cases of demodicosis between 1998 and 2019 and all were juveniles between 4-8 months of age (Mantellato and Lambert, 2019). Transmission was not immediate and not all individuals housed together were affected. There was no predilection for island or mainland stock, or male versus female (Mantellato and Lambert, 2019). Lesions often regressed spontaneously, or in response to treatment with topical ivermectin 0.1% (Eden et al., 2004; Mantellato and Lambert, 2019; Bowry, 2021). Infection is associated with inadequate immune system function (Mantellato and Lambert, 2019), hence the predilection for juveniles with an immature immune system. The species of *Demodex* infecting dibblers have recently been investigated by Bowry (2021). During this study, two species were identified, both of which were morphologically distinct from other species described in marsupials and house mice. Mice on Boullanger and Whitlock Islands also harbour *Demodex* mites (Mathews et al., 2006) with four species of *Demodex* from three polyphyletic clades molecularly confirmed by Bowry (2021).

While infections with other mite species have also been reported, other ectoparasite taxa (fleas, lice and ticks) have not been documented on dibblers. Woolley (1971) reported a heavy infestation of *Ornithonyssus bacoti* mites in a female and her young, which was managed with frequent cleaning of the nest box and cage. Mesostigmata mite infestations, including *O. bacoti* (Lambert and Frost, 2009), have also been reported in the captive population at Perth Zoo (Lambert, 2000). Resolution of infection was achieved by treating infected individuals with 2.5 g/L fipronil (*Frontline®*) and disinfecting nest boxes with a residual permethrin-based insecticide (Lambert, 2000). Mites may have been introduced into the captive colony via contaminated leaf litter (Lambert and Mills, 2006) or food (i.e., flies; Lambert, 2009).

Other sporadic cases of disease in captive dibblers include neoplasia (see Appendix 2), respiratory distress/pneumonia, heart disease, various eye conditions and trauma (Lambert, 2009; Moore, 2019). A case of haemochromatosis (iron storage disease (ISD); Clauss and Paglia, 2012) was recently reported in a captive-bred dibbler released onto DHI in 2020 (Cowen, 2021). While ISD is often associated with excess dietary iron (e.g., Brust, 2013), infectious disease, starvation and hereditary disorders may also cause ISD (Clauss and Paglia, 2012) and it is unclear what the underlying aetiology was in this case. Lastly, eight cases of mastitis have been documented in captive females 5-10 days post-weaning, three of which died (Moore, 2019). *Pseudomonas aeruginosa* was implicated in one of these cases (Vitali, 2015). Modified weaning protocols (see section 2.5) have reduced the incidence of mastitis in captivity.

In 2003, non-invasive disease surveillance for gastrointestinal (GIT) parasites was undertaken with the aim of screening dibblers and mice from different populations to identify and compare the parasites infecting both species and determine whether there is the potential for inter- and/or intraspecific parasite transmission during translocation (Mathews et al., 2006). Faecal samples collected from captive (island origin), source (Whitlock and Boullanger Islands) and translocated (Escape Island) populations were screened for the presence of GIT parasites. Twelve parasite taxa were identified in dibblers from the source population, whilst 22 taxa were found in mice (Mathews et al., 2006). For dibblers, the captive cohort had the highest number of parasite taxa (n = 16), and the source population had a higher number (n = 12) than the translocated population (n = 8). Coccidia and nematodes were detected in the source and captive populations but were absent in the translocated population. Several novel bacterial parasites that were acquired in captivity, were also lost following translocation (Mathews et al., 2006). *Bacillus cereus* was identified in captive dibblers and is a potential pathogen (Moore, 2019). *Salmonella bovis morbificans* PT24, a potential pathogen and zoonotic risk, was also identified in mice

(Mathews et al., 2006). This study provides important baseline health data, which can be used to inform disease risk and the need for parasite management during translocation.

For the dibbler, disease and parasite considerations were first incorporated into translocation protocols in 2014 and were reliant on strict quarantine protocols and health monitoring by keepers and veterinary staff at Perth Zoo to detect disease (Mawson, 2014). Dibblers entering the captive colony were quarantined for a minimum of 30 days, during which time faecal samples were collected for parasitological analysis and animals received a full health assessment (including physical examination and blood collection) under anaesthesia (Moore, 2019). Dibblers have not been treated prophylactically for any disease condition, only in response to clinical disease (Lambert, 2000). Prior to translocation, however, dibblers have received treatment to remove parasites if they were identified (Mawson, 2014). In the absence of any benefit to host health, the prophylactic use of antiparasitic drugs requires careful consideration, particularly for species with low genetic diversity, which may be more susceptible to disease (Wait et al., 2017). The dibbler is also host to two dependent host specific *Demodex* species (Bowry, 2021), which puts these (presumably endangered) parasites at risk of co-extinction.

Prior to the introduction of dibblers onto DHI, a qualitative disease risk analysis (DRA) was undertaken by Moore (2019). While the translocation was regarded as a low disease risk scenario, the risks of disease transfer from native sympatric species and house mice on DHI to dibblers was not evaluated. Based on this DRA, targeted evaluation of gastrointestinal health across all dibbler populations, and research to further evaluate *Demodex*, trypanosomes and viral disease was recommended (Moore, 2019). Incorporating disease risk analyses into future translocation protocols has also been proposed (Matthews et al., 2006; Moore, 2019).

2.10 Genetics

Originally described as *Phascogale apicalis* (Gray, 1842), the dibbler was later reclassified as *Parantechinus apicalis* (Tate, 1947) and now represents the sole species of this genus (Jackson and Groves, 2015). Studies of phallic morphology support this grouping (Woolley, 1982). Despite morphologic (i.e., substantially smaller size of island dibblers; Start, 1998) and physiologic (e.g., shorter gestation period in island dibblers; Mills et al., 2012) disparity between island and mainland populations, no subspecies of the dibbler have been formally described. Genetic differentiation between island and mainland populations is indicative of two major lineages (Mills et al., 2012; Thavornkanlapachai et al., 2019) prompting their separate conservation management.

The dibbler persists within small, fragmented populations and is at risk of extinction through loss of genetic diversity resulting from random genetic drift and increased inbreeding (Weeks et al., 2015). Evidence of developmental instability and a greater risk of inbreeding depression was first identified within the Whitlock Island population due to its small size and therefore greater probability of matings between close relatives (Lynam, 1987). Pouch young on Whitlock Island were less numerous, but larger in size, than Boullanger Island pouch young (McCulloch, 1998). A subsequent study that examined genetic variation at seven microsatellite loci (Mills et al., 2004) reported low levels of heterozygosity and high levels of inbreeding in the Boullanger and Whitlock Island populations (particularly Whitlock) compared with the much larger FRNP population. Significantly lower numbers of pouch young on Whitlock Island also raised concerns of inbreeding depression (Mills et al., 2004). Both island populations displayed evidence of genetic bottlenecking, whereas the FRNP population did not, with Boullanger and Whitlock Islands retaining only 46% and 28% of the genetic diversity of the FRNP population, respectively (Mills et al., 2004).

Wilcox (2003) examined the effects of translocation on genetic diversity across five microsatellite loci by comparing translocated (Escape Island) and source (Boullanger and Whitlock Islands) populations five years after translocation (samples collected up to mid-2003). Genetic variation and the percentage of polymorphic loci was significantly lower on Escape compared with Boullanger Island, though both populations had the same number of alleles per locus. In contrast, genetic diversity was significantly higher on Escape compared to Whitlock Island with more alleles per locus, but the same number of polymorphic loci. The Escape Island population showed significant genetic divergence from Boullanger, but not Whitlock Island. Given that the captive colony was established with a small number of wild-caught dibblers from Boullanger and Whitlock Islands (n = 8; two pairs from each island; Moro, 2003), and the first translocated cohort was relatively small (n = 26), a combination of founder events and genetic drift most likely explain the divergence (Wilcox, 2003). In contrast, a more even contribution from Whitlock and Boullanger Island dibblers to the Escape Island population was identified by Thavornkanlapachai (2016) using more recent samples (2002-2012), while Aisya (2018) reported that the Escape Island population was genetically most similar to Boullanger Island (samples 2013-2018).

Loss of genetic diversity is common in translocated and captive populations due to founder effects and when ongoing population size is small (Weeks et al. 2015). Dibblers reintroduced to Peniup (captive-bred stock of FRNP provenance) have suffered a 10 to 16-fold reduction in effective population size compared with the source population, including the loss of several rare alleles (Thavornkanlapachai et al., 2021). Genetic relatedness was also higher in captive and reintroduced populations compared with the source population. While some loss of genetic diversity coinciding with genetic bottlenecks occurred after establishment, the sufficient size of the captive founder colony (n = 26) and rapid population growth during the establishment phase, limited loss of genetic diversity (Thavornkanlapachai et al., 2021). As this population is small and isolated however, the long-term viability of this population remains uncertain with loss of genetic diversity expected to occur over time due to random genetic drift (Thavornkanlapachai et al., 2021).

There is also concern for the long-term viability of island populations. Aisya (2018) showed that Escape Island had the highest allelic and genetic diversity of the three Jurien Bay island populations. However, all three populations were estimated to have low genetic diversity (microsatellite expected heterozygosity H_E = 0.05

Whitlock, 0.28 Boullanger, and 0.39 Escape; allelic richness A_R = 1.07 – 2.0). While population viability analysis (PVA) predicted high homozygosity values (> 0.9, Whitlock Island), inbreeding coefficients (F_{IS}) were slightly negative. For Whitlock Island (F_{IS} = -0.06), this contrasts with previous studies (Mills et al., 2004; Thavornkanlapachai, 2016) that reported positive F_{IS} values; though analysis of pairwise relatedness (*r*) showed that Whitlock Island dibblers were, on average, highly related to each other (*r* > 0.8), indicating that inbreeding should also be high (Aisya, 2018). However, the low variability present at genetic markers in this population is likely impacting these results. PVA modelling also predicted that population size and genetic variation will continue to decline over the next century, with Whitlock Island predicted to have the lowest genetic diversity, highest level of inbreeding and greatest extinction risk (Aisya, 2018).

The FRNP mainland population has retained comparatively high levels of genetic diversity within all sites (Thavornkanlapachai et al., 2019). Dibblers from seven sites within the FRNP were screened for genetic diversity at 17 microsatellite loci, with heterozygosity and allelic diversity levels double that found in island populations (H_E = 0.68 to 0.71; A_R = 3.4 – 4.1). Two FRNP subpopulations were identified (eastern and western), dissected by the Fitzgerald River. Significant genetic differentiation between sampling sites and evidence of fine-scale genetic structure of up to 200 m in females, indicated that dibblers exhibit female philopatry and male-biased dispersal at this location (Thavornkanlapachai et al., 2019). Lynam (1987) also reported male-biased post-weaning dispersal on Boullanger Island.

The genetic structure and diversity of the Gunton Island population has not been examined since the population was first established in 2015, however, given that this population was founded using FRNP mainland stock it is likely to have retained relatively high genetic diversity. Nevertheless, given that the majority of genetic diversity within the species resides in the FRNP population it is vital that this component of genetic diversity is maintained; other than Peniup, Gunton Island is the only other population containing FRNP genetics.

DNA Zoo (https://www.dnazoo.org/) is in the process of sequencing the entire dibbler genome, which may help inform genetic management of the species in the future (P. Kaur pers. comm.).

2.11 Community involvement

Community groups and volunteers have made a significant contribution towards the recovery of the dibbler, providing hands-on assistance and enabling research to be carried out in a cost-effective manner (Friend, 2004a). With the establishment of the Dibbler Recovery Team in 1996, a representative from the Jurien Bay community was appointed to help facilitate dibbler conservation initiatives (e.g., dibbler monitoring on islands) and raise community awareness within the area. Assistance from the Jurien Marine Parks staff and volunteers is still provided today (Friend, 2022).

Invited to join the Dibbler Recovery Team in 2002, the Malleefowl Preservation Group (MPG) has provided support to DBCA through the provision of volunteers and by supporting funding applications for the implementation of dibbler recovery actions within Peniup, a key MPG malleefowl monitoring site (Friend, 2002). The MPG for example, assisted with obtaining National Heritage Trust funding through a cross-regional project with the two NRM regional groups that cover the dibblers range (Friend and Button, 2007), to help implement Dibbler Recovery Plan actions (e.g., population monitoring; Friend, 2008).

In 2015, a representative from the Friends of the Fitzgerald River National Park (FOTFRNP; https://www.fitzgeraldfriends.org.au) was invited to join the Recovery Team. The FOTFRNP promote and support ecological studies and conservation management within the FRNP and have provided volunteer assistance with dibbler surveys and monitoring in collaboration with DBCA. After successfully obtaining a State NRM Community Action Grant, the FOTFRNP assisted with monitoring of the DBCA-established dibbler grid in the eastern FRNP in collaboration with the Esperance Indigenous Green Army team and other volunteers between 2016 and 2020 (Chapman, 2018).

3 Future research directions

While considerable progress has been made to safeguard the dibbler and reduce the likelihood of extinction, the conservation status of the dibbler has not been downgraded since recovery initiatives were first implemented over 25 years ago (Start, 1997). The fragmented nature of populations means that ongoing management to maintain genetic representativeness and promote long-term viability is likely to be required. For the species to be resilient to threats such as disease and climate change, and to improve the capacity for future recovery, the remaining genetic variability needs to be maintained. To support dibbler recovery into the future, we make some suggestions for further research below. A separate structured decision-making process would help to prioritise future research and conservation management actions (e.g., Pritchard et al., 2022), such as threat mitigation, that are most likely to benefit the dibbler.

3.1 Improving genetic health through meta-population management

Given recent population declines and evidence of low genetic diversity and inbreeding of dibblers on Boullanger and Whitlock Islands (Mills et al., 2004; Aisya, 2018; Thavornkanlapachai et al., 2019), augmented gene flow between island populations may be warranted (Whiteley et al., 2015). Supplementing Boullanger Island with Escape Island stock for example, has been proposed to boost this population (Mills et al., 2019). The Escape Island population, which was established by mixing Boullanger and Whitlock Island stock, has the highest genetic diversity of the three Jurien Bay Island populations (Aisya, 2018). Admixture between island populations could have contributed to alleviating the effects of inbreeding depression (‘genetic rescue’) in the Escape Island population, however, other factors such as the greater size (and carrying capacity) of the island, absence of the house mouse or higher seabird nutrient input is also likely to explain the significant population growth observed.

As the FRNP population is larger and more genetically diverse than island populations, supplementing island populations (or establishing new translocated populations) with mainland stock has been proposed by Aisya (2018) to increase genetic diversity and reduce inbreeding risks. Rick (2021) is further evaluating population genetic diversity and structure of island and mainland populations using SNP-based methods, which deliver much greater power for genetic analyses, and investigating the feasibility of genetic rescue between the mainland and island populations. Given the highly divergent nature of island versus mainland populations (Mills et al., 2004), the risk of outbreeding depression would need to be carefully considered. Morphologic and physiologic disparity may also inhibit interbreeding, or reduce fecundity (Aisya, 2018). A trial under controlled conditions at Perth Zoo could be undertaken to assess whether interbreeding is possible, and if so, determine the risk of outbreeding depression (i.e., run trials to at least the F2 generation; K.

Ottewell pers. comm.). In practice, genetic rescue can be achieved with introduction of only small numbers of individuals to contribute novel diversity while reducing the risk of ‘genetic swamping’ (Weeks et al., 2017; Madsen et al., 1999).

Determination of the genetic status of Gunton Island, seven years on from when it was established, would also help to inform future supplementations from mainland stock. In addition, the development and implementation of a population management strategy would help to manage sub-populations as part of a defined meta-population to effectively maintain the species’ genetic diversity and representativeness over the long term.

3.2 Selecting new translocation sites

With no additional dibbler populations discovered on the mainland, and the long-term viability of the reintroduced Peniup population in question, the FRNP population remains the stronghold of the species with its relatively high genetic diversity and distinct eastern and western clusters (Thavornkanlapachai et al., 2019). Establishing additional populations is important for preserving this element of genetic diversity and securing the persistence of the species. Due to difficulties encountered in establishing new mainland populations, the further use of island sites for dibblers in the short-term should be given strong consideration. A structured decision-making exercise that prioritises potential mainland and island locations would facilitate this selection process.

Suitable dibbler habitat within the species’ former range is known to exist (Friend, 2004a), however the selection of an appropriate site is dependent upon satisfying several other host environment criteria. For the dibbler, invertebrate abundance is one of the most important determinants of habitat suitability (Lerch, 2015). The ability to implement sustained introduced predator control and appropriate fire management is also crucial for population persistence (Thavornkanlapachai et al., 2021). Identifying sites with reliable access is preferred to ensure monitoring and predator management can be consistently undertaken. Other critical requirements for the successful establishment of populations need to be investigated, including the importance of fine-scale habitat attributes such as the abundance of refuge sites (e.g., burrows) and leaf litter layer depth for foraging. Vegetation cover is also important for the post-release survival of captive-bred stock with a greater propensity for diurnal activity (e.g., as observed on DHI; S. Cowen pers. comm.). The influence of climate change should also be considered when selecting translocation sites (see below).

3.3 Determining influences on population trends

Statistical appraisal of dibbler population fluctuations in relation to changes in environmental attributes by interrogating the existing monitoring data for the Jurien Bay islands would help to better understand the influence of climate and other attributes on these populations. In addition, investigation of the monitoring data available for mainland dibbler populations on the south coast may assist in understanding the status of these populations. These analyses would also help inform ongoing monitoring requirements for all the dibbler populations (e.g., frequency and timing) – see below.

3.4 Developing a robust monitoring strategy

Population-level monitoring is required to measure the effectiveness of recovery actions, as well as to demonstrate that success criteria for translocations have been met, and that source populations have not been impacted by removing animals for translocations. The need for a cost-effective monitoring strategy, with standardised methodology, appropriate for each dibbler population has been identified by the Dibbler Recovery Team to provide robust estimates of abundance and determine long-term trends. The value of remote camera monitoring for populations with unreliable access needs to be evaluated, given that live trapping permits assessments of the health of individuals and enables the collection of samples for genetic analyses to inform ongoing management. In addition, developing eDNA approaches by sampling of the soil/leaf litter to detect DNA of the dibbler could be another non-invasive monitoring method to provide information on the species' presence.

3.5 Assessing climate change impacts and promoting resilience

There is some evidence to suggest an association between below average rainfall and population declines of the dibbler (Friend, 2021a), which may indicate susceptibility to a drying climate. In addition, with increasing temperatures and reduced winter and spring rainfall predicted for south-western WA (Hope et al., 2015), the frequency and intensity of extreme weather events (e.g., wildfire) is likely to increase. Climate change is also likely to interact with other threats to dibblers such as dieback disease. Sea levels are also predicted to rise (Hope et al., 2015), reducing the carrying capacity of islands that support dibblers. As such, the long-term viability of some island populations may be compromised.

To promote adaptive capacity and resilience to climate change, genetic diversity should be maximised (Sgrò et al., 2011). Emerging genomic approaches such as targeted gene flow to identify favourable traits and promote adaptive capacity to climate change (see Kelly and Phillips, 2015) may benefit the dibbler. Likewise, the development of genetic markers to screen for variation in genes of interest, for example, ensuring there is sufficient diversity at Major Histocompatibility Complex loci, which is important for immunological fitness (Smith et al., 2010), may also assist with conserving functional gene diversity and enhancing adaptive capacity.

Undertaking a climate change vulnerability assessment (e.g., Foden and Young, 2016), may assist with identifying and mitigating the potential impacts of climate change on dibblers.

3.6 Understanding the influence of prescribed fire

With climate change predicted to promote a harsher fire-weather climate in the future (Hope et al., 2015), minimising the risk of high intensity fires destroying whole habitats and populations, while still retaining areas of long unburnt habitat preferred by dibblers, requires considered fire management planning. Knowledge regarding the influence of prescribed fire on dibblers and their habitat, including recolonisation/habitat recovery post-burn, and the availability of invertebrates and refuge sites post-burn will help to inform fire management.

4 Conclusion

The recovery of the dibbler is contingent on habitat protection and mitigation of key threats together with the implementation of other targeted conservation actions. Captive breeding for translocation has been critical in the recovery of the dibbler, effectively increasing the species' range and number of known populations, and reducing the likelihood of extinction. Recovery into the future will be reliant on management actions that are informed by genetic analyses to promote long-term viability of dibbler populations. We have provided some suggestions regarding future research directions to inform the recovery of the dibbler. Many of the recovery actions outlined in the 2004 Dibbler Recovery Plan are still likely to be applicable to the management of the species today, though a structured decision-making process could be considered to prioritise future research and conservation management actions that are likely to provide the most benefit to the dibbler.

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Appendices

Appendix 1 Dibbler translocations carried out between 1998-2021. Bold typeface indicates founding year with the number of founding stock in parentheses. Non-bold typeface signifies additional supplementations. Note: *Wild-to-wild translocation; **Includes 2 females with 8 pouch young; ***Includes 3 females with 8 pouch young.

Translocation site	**Source site**	**Outcome**	**Year (no. released)**
Escape Island	Captive colony (Boullanger and Whitlock Island origin)	Successful	**1998 (n = 26)** 1999 (n = 41) 2000 (n = 19)
Peniup proposed Nature Reserve	Captive colony (Fitzgerald River National Park origin) Fitzgerald River National Park (wild) Gunton Island (wild)	Successful	**2001 (n = 41)** 2002 (n = 46) 2003 (n = 43) 2006 (n = 6) 2007 (n = 3)* 2008 (n = 24) 2009 (n = 34) 2010 (n = 41) 2014 (n = 6)** 2017 (n = 69) 2018 (n = 8)*
Stirling Range National Park	Captive colony (Fitzgerald River National Park origin)	Failed	**2004 (n = 57)** 2005 (n = 62) 2006 (n = 38) 2007 (n = 40)
Waychinicup enclosure	Captive colony (Fitzgerald River National Park origin)	Failed	**2010 (n = 20)** 2011 (n = 74)*** 2012 (n = 84) 2013 (n = 58)**
Whiteman Park	Captive colony (Fitzgerald River National Park origin)	Failed	**2014 (n = 55)** 2015 (n = 9)
Gunton Island	Captive colony (Fitzgerald River National Park origin)	Successful	**2015 (n = 29)** 2016 (n = 47) 2017 (n = 38)***
Dirk Hartog Island	Captive colony (Whitlock and Escape Island origin)	Establishing	**2019 (n = 26)** 2020 (n = 31) 2021 (n = 37)

Appendix 2 Infectious and non-infectious agents recorded in dibblers

Infectious			**Agent**	**References**
Parasites	Endoparasites	Protozoa	Coccidian oocysts	Mathews et al., 2006
			Isospora sp.	Moore, 2019
			Klossiella sp.	Moore, 2019
			Toxoplasma gondii	Attwood and Woolley, 1982; Canfield et al., 1990
			Trypanosoma sp. DBA1	Averis et al., 2009
		Nemotodes	*Trichuris* sp.	Mathews at al., 2006
			Unidentified strongyle eggs	Mathews at al., 2006; Moore, 2019
			Woolleya sprenti	Moore, 2019
		Cestodes	*Rodentolepis* sp.	Moore, 2019
		Pentastomids	*Armillifer* sp.	Ladds, 2009
			Waddycephalus sp.	Ladds, 2009
	Ectoparasites	Mites	*Demodex* spp.	Eden et al., 2004; Moore, 2019; Bowry, 2021
			Ornithonyssus bacoti	Woolley, 1971
			Order Mesostigmata (*Ornithonyssus* spp.)	Moore, 2019
			Family Trombiculidae	Moore, 2019
Bacteria		Gram-negative	*Enterobacter* sp.	Mathews et al., 2006
			Escherichia coli	Mathews et al., 2006
			Flavobacterium sp.	Mathews et al., 2006
			Klebsiella oxytoca	Mathews et al., 2006
			Proteus vulgaricus	Mathews et al., 2006
			Serratia marcescens	Mathews et al., 2006
			Unidentified organisms	Mathews et al., 2006
		Gram-positive	*Bacillus cereus*	Mathews et al., 2006
			Bacillus sp.	Mathews et al., 2006
			Enterococcus faecalis	Mathews et al., 2006
			Enterococcus sp.	Mathews et al., 2006
			Staphylococcus aureus	Mathews et al., 2006
			Staphylococcus haemolyticus	Mathews et al., 2006
			Staphylococcus saprophyticus	Mathews et al., 2006
			Staphylococcus sp.	Mathews et al., 2006
			Streptococcus sp.	Mathews et al., 2006
			Pseudomonas aeruginosa	Moore, 2019
			Salmonella spp.	Moore, 2019

Non-infectious	**Agent**	**References**
Neoplasia	Carcinoma (small intestinal)	Moore, 2019
	Chondrosarcoma (lung)	Moore, 2019
	Diffuse lymphoid or myeloid leukaemia	Moore, 2019
	Islet cell adenoma	Attwood and Woolley, 1980
	Lymphosarcoma	Attwood and Woolley, 1980; Canfield et al., 1990
	Small cell lymphoid neoplasia (inguinal area)	Moore, 2019
	Undiagnosed neoplastic tumour (stomach)	Moore, 2019