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Long term monitoring of recruitment dynamics determines eradication feasibility for an introduced coastal weed

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Abstract

Bitou bush (*Chrysanthemoides monilifera* subsp. *rotundata*) is a Weed of National Significance in Australia and has impacted a significant portion of the eastern coastline. Its discovery in Western Australia was, therefore, a cause for concern. Assessment and control of the isolated and well-defined population began in 2012. To assess the feasibility of eradication in Western Australia as a management outcome for bitou bush, we applied a rigorous data-driven quantification and prediction process to the control program. Between 2012 and 2018 we surveyed over 253 ha of land and removed 1766 bitou bush plants. Approximately 97 person-days were spent over the six years of survey. We measured the seed bank viability for five years starting in 2013, with the 2017 survey results indicating a decline of mean viable seeds/m^2 from 39.3 $\pm$ 11.4 to 5.7 $\pm$ 2.2. In 2018 we found only ten plants and no newly recruited seedlings in the population. No spread to other areas has been recorded. Soil core studies indicate that the soil seed bank is unlikely to persist beyond eight years. Eradication of the population in Western Australia, defined as five years without plants being detected, therefore remains a realistic management goal. The information generated from the documentation of this eradication program provides invaluable insight for weed eradication attempts more generally: novel detection methods can be effective in making surveys more efficient, all survey methods are not entirely accurate and large plants can escape detection, bitou bush seeds persist in the soil but become effectively undetectable at low densities, and migration of seed was unquantifiable, possibly compromising delimitation. Continued monitoring of the Western Australian population will determine how much of a risk these factors represent to eradication as the outcome of this management program.

Keywords
Control effort, management, monitoring, population decline, seed bank, seed viability, survey, weed control, weed eradication.

Introduction

The management of invasive alien plants costs billions of dollars and represents a significant challenge for a range of stakeholder groups, including agriculture, conservation and tourism (Pejchar and Mooney 2009; Shackleton et al. 2019). Depending on the characteristics and context of the invasion, the suitability of management approach and the likelihood of different outcomes can vary substantially (Blackburn et al. 2011). If the situation is appropriate then a desirable goal is the enduring eradication of the target population, but the reality is that this end point is particularly difficult to achieve (Wilson et al. 2017).

Eradication means the elimination of every single individual of a species from a defined area to a level beyond which recolonization is unlikely to occur (Myers et al. 1998). In the case of plants, eradication can be a reasonable management goal if the area of spread is small, if the target species is easily identified among other vegetation, if the infestation is detected early enough, and if sufficient resources are available for the full eradication program, especially in the final stages when target abundance is very low (Wilson et al. 2017). Unfortunately most species targeted by eradication programs have different characteristics, and eradication is only feasible in a limited number of circumstances (Wilson et al. 2017). It is, therefore, particularly important to share lessons learnt from eradication attempts if we are to improve outcomes for these management programs.

Soil seed bank longevity is often a key determinant of eradication success or failure, but is rarely measured. Panetta (2007) estimated that weed eradication programs often required 10 or more years to achieve their objective, or at least attempt the objective of complete mortality of any remaining seed bank. Very few weeds have been successfully eradicated world-wide, largely due to the seed bank outlasting monitoring and control efforts. For example, in Australia Panetta (2004) mentions that *Eupatorium serotinum* in southern Queensland required 18 years, to eradicate despite covering a relatively small area. However, in Western Australia, the annual chenopod herb, *Bassia scoparia*, was eradicated following a ten year monitoring and control program (Dodd and Randall 2002). The success of this program was largely due to the weed having a short-lived seed bank, mostly one year, with up to three years possible (Dodd and Randall 2002).

We have previously reported on the discovery of one of Australia's Weeds of National Significance, *Chrysanthemoides monilifera* (L.) T. Norl. subsp. *rotundata* (DC.) T. Norl. (Asteraceae; hereafter, bitou bush), as a well-established population in Kwinana, a major port and industrial area near Perth, Western Australia (Weiss et al. 1998, Scott and Batchelor 2014). Our initial surveys in 2012 documented a restricted extent for this bitou bush incursion, aside from the potential for seed export from the site (Scott and Batchelor 2014). The Kwinana bitou bush population has existed, as verified from aerial photography, since at least 1995 (Scott et al. 2019). The Kwinana population is unusual because the nearest populations of bitou bush are found more than 3,000 km to the east as an introduction in eastern Australia, or more than 8,000 km to the west as a native species of southern Africa (Scott and Batchelor 2014). This specific context makes it feasible to consider eradication as a management goal for bitou bush in WA, rather than local extirpation where there is a possibility of recolonization (Wilson et al. 2017).

Here we document the changing dynamics of the bitou bush infestation in Western Australia as an eradication program is implemented for its control. To address knowledge gaps in past eradication programs and to inform future control programs, we included information on both plants and the seed bank. We use this to make an ecologically informed assessment of the feasibility of eradication of bitou bush from WA, including for how long the infested area should be monitored. More generally, we use the example of bitou bush to illustrate how plant population biology data can be used to provide greater context, and thus improve the likelihood of success for the popular but rarely achieved objective of eradication for plant invasion management programs.

Materials and methods

Population surveys

The focal population was located in the Kwinana region of Perth, Western Australia – an industrial port area with mixed land-use, including fragments of low-quality native vegetation, and high levels of industrial goods traffic on and off-shore (32°12.652'S, 115°46.018'E). Scott and Batchelor (2014) provides details on the context of the focal population, including local climate, landscape and features that influence delimitation and detection. Initial surveys of the population started in September 2012 (Scott and Batchelor 2014), continued through 2013 and were repeated each year in March and April from 2014 to 2018 (Figures 1, 2). Additional site visits were made in spring (October-November) of each year, mainly to observe if there were newly germinated plants. The main infestation area only was surveyed in 2014 (the main or "core" infestation is defined in Scott and Batchelor (2014) Figure 1b). Port and industrial activity (e.g. construction of paved areas for heavy equipment, roads, new industries) meant that surveyed areas were not identical each year (Figure 1).

Surveys were done on foot, with personnel making traverses 5–10 m apart (depending on the density and height of vegetation). Particular interest was paid to areas under obvious bird perches (trees, fences, buildings and lights posts), likely locations for new seedlings resulting from bird dispersal of the fleshy coated seeds of bitou bush (Gosper 1999). Surveys were timed for March and April (i.e. end of summer) when the lush green of the bitou bush foliage is easily distinguished from surrounding vegetation. At this time the annual and perennial grasses and other annual vegetation has died back following the dry hot summer that is typical of the Mediterranean climate of this region. Records were kept of the search effort, measured in person days (where a

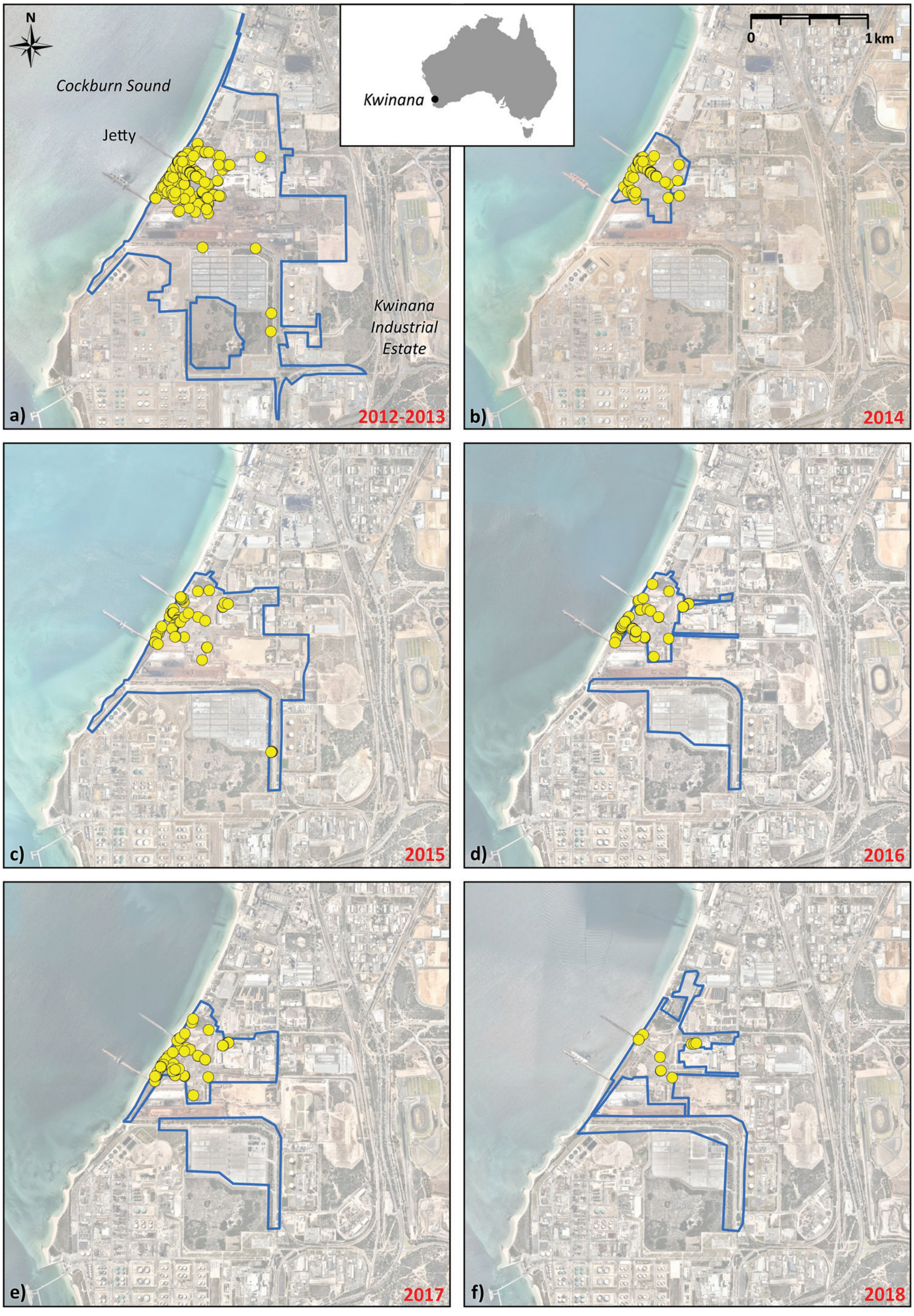

Figure 1. Location of each bitou bush (yellow dot) and outline of the survey area (blue) for each year (between 2012 and 2018) in the Kwinana Industrial Estate, Western Australia. Background images generated by Nearmap were taken in February of the designated year.

Figure 2. Bitou bush soil-core plant locations in the Kwinana Industrial Estate, Western Australia. Three zones were sampled: Kwinana Bulk Terminal (KBT) beach (red), Foreshore Rd (blue) and Horse Beach (green), in addition to a selection of other scattered plants. The background aerial photograph dates from 8 Jan 2012 (reproduced with permission of Western Australian Land Information Authority; CL05 – 2013).

day is considered to be 8 AM to 4 PM). A total of 97 person days have been expended on the survey (see Suppl. material 1).

The location of each bitou bush was mapped using a differential GPS (Hemisphere R100 Series Receiver utilizing the Australian Maritime Safety Authority DGPS beacon system for differential correction). All spatial information was managed within ArcInfo 10.3 (https://www.esri.com/).

Buffer zones

As the majority of plants discovered were found within a 500 m radius of a dilapidated wooden jetty, this area was defined as the main infestation (Scott and Batchelor 2014). A buffer zone of a further 500 m was calculated in ArcMap 10.3 based on the maximum observed dispersal distance (as shown in figure 3 in Scott and Batchelor (2014). Any reproductive plants detected in this buffer zone would determine the need to expand the buffer zone until such point as the population was delimited, or at a point where the population was determined to be beyond the scope of eradication. To provide further confidence in survey scope decisions, additional transect areas were surveyed beyond the defined monitoring regions, either along the coast or based on potential areas of seed dispersal by birds. These outer areas were surveyed by vehicle where the landscape was open, or on foot where there was dense vegetation or lack of vehicle access. Further details of the buffer zones and initial surveys are provided in Scott and Batchelor (2014).

Aerial photography

To check for the possibility of missed plants in ground surveys, the survey area was examined using recently obtained very high resolution aerial photography (5.8–7.5 cm per pixel) in 2016 and 2018. These photos were available from Nearmap (https://www.nearmap.com.au/) via the University of Western Australia. Potential bitou bush plants that may have been overlooked by the ground surveys were identified from the photos and ground-truthed during additional surveys in 2016 and 2018.

Plant measurements

For each bitou bush plant we measured: size (maximum and minimum plant canopy diameter (m), maximum height (m), basal stem diameter (mm)), and reproductive output (number of capitula in flower, number of capitula with seed). A capitulum is an inflorescence consisting of a head of closely packed stalkless flowers typical of the Asteraceae. We noted the habitat type and whether there were proximate perch points for birds (e.g. trees, fences, other structures). Plants were classed as seedlings if the cotyledons were still attached and green. Seedlings were counted rather than measured for size.

Plant removals

We killed the plants once measurements were taken. This was mostly achieved by uprooting the plant by hand. Some larger plants (~40) were removed with the assistance of earth-moving machinery and/or killed by herbicide (arranged by the landowners). Follow up surveys confirmed the death of all plants sprayed with herbicide or that had been hand-weeded.

Bitou bush is relatively shallow rooted, but care was required to make sure all stem material was removed from the soil, as the plant is capable of regrowth from stems cut at ground level or by layering from buried stem sections (Weiss et al. 2008). Some layered plants required multiple annual visits to confirm that all layered stem sections had been found. Regrowing layered sections were not counted as new individuals. All uprooted plants were left *in situ* to mark their original location so any regrowth could be readily found in future surveys.

Most of the new plants found in 2013–2017 were seedlings germinating from the soil seed bank under the canopy area of plants removed in 2012. These new plants were all small enough to be removed by hand when detected. However, seedlings can be small and hidden in other vegetation and consequently not detected in the year of germination, but readily detected when larger in the following year. Even so, one or two plants per year remained previously undetected (e.g. hidden in bushes or in difficult to access parts of the port infrastructure) and only detected when flowering occurred. All these plants were removed by hand.

Seed bank studies

We selected 15 plants within three locations in the main infestation area (Beach, Foreshore Road, Horse Beach) as sites for detailed seed bank studies (locations shown in Figure 2). All selected plants had evidence of flowering and seed production in 2012 and were > 3 m in canopy diameter. In 2013, 10 soil cores (10 cm wide by 15 cm deep) were taken from underneath the canopy and 10 soil cores were taken 1 – 2 m from the perimeter of the plant canopy. Each soil core was sieved on site, outside of the canopy area, to minimise seed bank disturbance. Initially, large leaf litter was removed with a >1 cm sieve, then fine sand was removed using a < 3 mm mesh sieve. The remaining material from each soil core was taken in a paper bag to the laboratory where it was sorted under 10× magnification (Magilamp) to partition debris from seeds and seed fragments. Seeds were then stored in vials for further quantitative processing.

We dissected intact seeds found in each soil core to assess their viability. We counted seed fragments (the seed coat naturally splits into three portions on germination or decay) and noted whether these seed parts had evidence of rodent damage (i.e. gnawing on the seed coat) or other evidence of seed predation (holes in the seed). We counted seed fragments because this provided confirmatory evidence that the soil core was taken under the original canopy (note that the plants were killed in 2012 and the canopy extent was known). Seed viability was assessed by the condition of endosperm in intact seed: firm, pale green/white in a viable seed, or dried, blackened or decayed in an unviable seed. Each soil core was assessed separately and aggregated by plant number for analysis.

We used the same method to measure the seed bank in 2014 and 2015, although the number of sampling sites decreased for various reasons: vehicle traffic destroyed one sample site at Horse Beach and five were lost by the removal of top soil along Foreshore Road and Horse Beach. Soil cores 1 m from the perimeter of the plants were not taken after 2013 owing to no viable seeds being found in any of these cores (totals in 100 cores: 6 seed fragments, 4 intact but dead seeds). In 2016 and 2017, the number of soil cores was increased to 20 per sample site. We increased the sampling effort in this way to compensate for the decreased frequency of viable seeds found previously in soil cores during 2014 and 2015. Sampling the seed bank was not carried out in 2018 because the target sampling areas were now highly disturbed (estimated at between 1 to 11% of the target area was cored and the rest trampled) and so few seeds were present that an unrealistic sample size of cores would be required to detect their presence, if there were seeds left at all. Instead specific attention was paid on the annual survey in the second half of 2018 to record seedlings present.

Soil cores were also collected from six plants on the fringe of the infestation to test whether isolated plants had a history of reproduction. Two methods were used: either 10 soil cores were taken under the canopy, as described above, or the collection and sieving of two or up to five samples of a 25 × 25 cm square of leaf litter and a 25 × 25 cm square of soil to 5 cm depth, also taken under the canopy.

Results

Annual surveys

Initial surveys in July 2012 covered three land holdings and identified 117 plants (Suppl. material 1: Table S1). Of these plants 138 were >1 m in canopy diameter, and the largest plant was over 11 m in diameter. The more detailed survey between September and December 2012 found 903 plants, of which most were seedlings and small plants <1 m in diameter, over an area that covered 253 hectares and 6 landholdings (Figure 1a). In 2013 the area within the main infestation was re-surveyed, capturing a further 365 plants, mostly seedlings located within the canopy area of plants removed in 2012. It required 59 person days, spread over two years (due to staff availability; Suppl. material 1: Table S1) to survey, locate and kill all plants, which is why the results for 2012–2013 are combined.

In 2014 the main infestation area was re-surveyed (Figure 1b) and subsequently the main and buffer area was re-surveyed each year (Figure 1 c–e) yielding in total

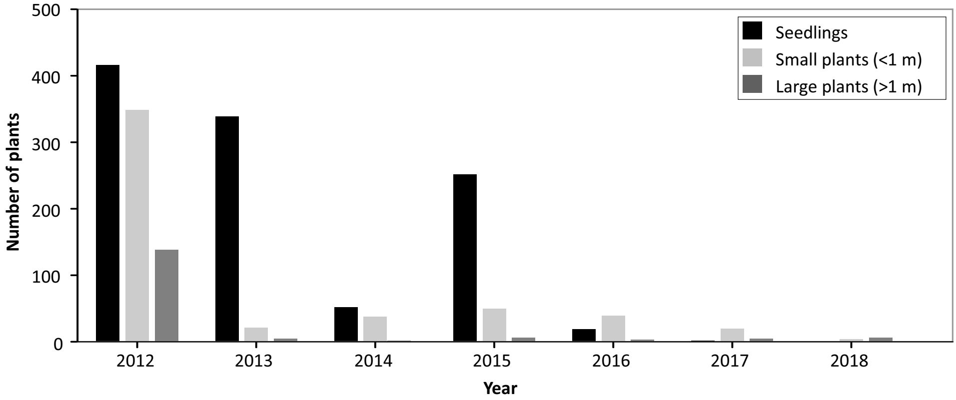

Figure 3. Comparison of number of bitou bush plants and seedlings found at Kwinana between 2012 and 2018. Plant size categories refer to maximum canopy diameter.

123 plants and 273 seedling plants over the four year period. Only 10 plants were found in 2018. Most plants found each year were small or newly emerged seedlings, but large plants that were greater than a year old were found, indicating that they were missed in surveys from previous years (Figure 3).

Of the 1756 individuals, all but five were found in the main infestation area (i.e. within 500 m of the putative population centre; Figure 1). Of these five, two plants were found within the first 500 m buffer from the main infestation: (i) a large (4 m canopy diameter) flowering plant in a nature strip, some 225 m from the nearest plant, and (ii) a small seedling in a reticulated garden bed, 284 m from the nearest flowering plant. Finding these two plants required expansion of the survey area a further 500 m. One plant was found in this third buffer zone (i.e. 1000 to 1500 m from the population centre), but as this plant was in a nature corridor (area where original vegetation including trees had been retained), and therefore an obvious dispersal route due to birds, the survey was extended along the nature corridor. This survey found two more plants. The discovery of these last two plants required a further redesign of the delimitation area so another 500 m and 1 km buffer specifically around these two plants was delimited and surveyed (see Scott and Batchelor (2014) for maps showing the sequentially increasing buffer areas).

The five individuals outside the main infestation area were found in the 2012 survey. In 2015, four plants were found growing in close proximity to where one of the five plants was killed in 2012 (location of plant number 624, on the southern end of the distribution; Figure 3 in Scott and Batchelor (2014). Upon closer inspection, all four plants had regenerated from the layering of stem material removed in 2012. None of the four plants had any evidence of seed production based on the absence of seed coat fragments in the soil. A large number of seedlings were found within the main infestation area in 2015, indicating conditions were more favourable for seed germination relative to other years (Figure 3). See also Suppl. material for results of live versus dead plants.

Survey effort and area

The number of plants found per unit of effort steadily decreased over the sampling period, which up to this point in the program has come to a total of 97 person days of surveys (Suppl. material 1). Extrapolating from the declining plant detection records indicates that there should be few plants found beyond 2019 (Figure 4a). The number of plants also decreased with the area searched each year (Figure 4b), similarly suggesting that few plants will be found after 2019 if there are no further additions to the soil seed bank.

The average size of plants found each year steadily decreased from 2012 to 2016 (Figure 5). In 2017, unseasonal summer rain (> 100 mm in February) and mild summer temperatures were likely responsible for the unusually high increase in the annual growth (diameter and height) of plants found (Figure 5). Over all survey years from

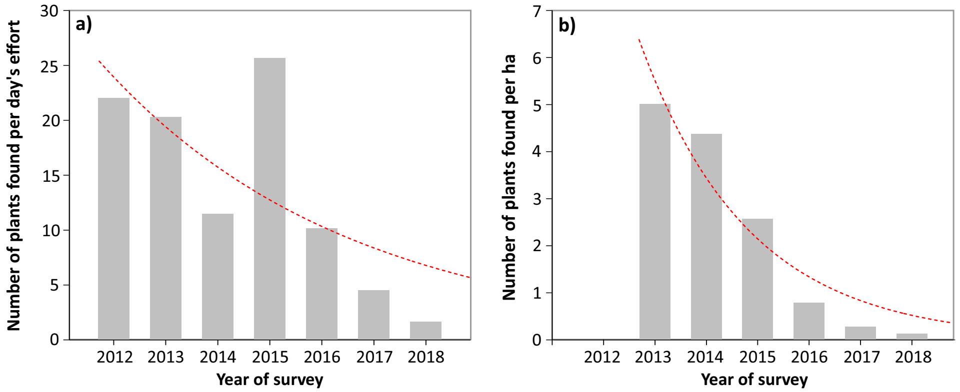

Figure 4. Number of bitou bush found with exponential decay trends represented (**a**) per person day effort ($Y = 29.54^{(-0.21X)}$, $R^2 = 0.54$, df = 6, $p < 0.05$) and (**b**) per ha searched ($Y = 14.16^{(-0.47X)}$, $R^2 = 0.91$, df = 5, $p < 0.01$).

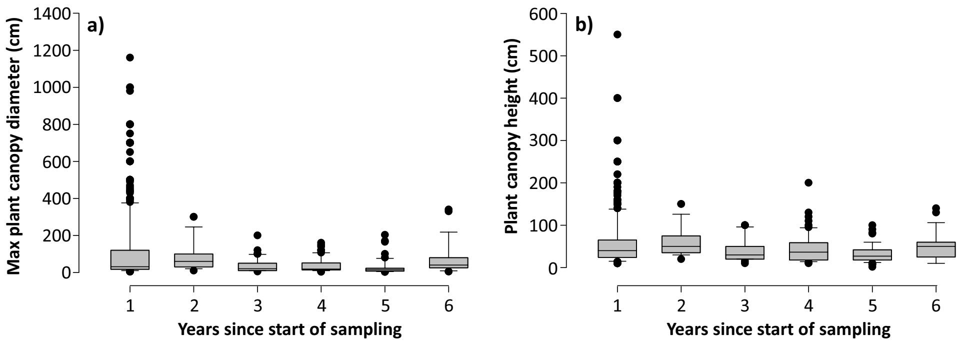

Figure 5. Canopy diameter (**a**) and height (**b**) for bitou bush plants in the Kwinana population between 2012 and 2017. Box plots depict median values with 10th, 25th, 75th and 90th percentiles. Results from 2018 are excluded as there were few individuals recorded and not all were measured.

2012 to 2017, nearly 70% of all small plants (canopy area < 1 m diameter) and seedlings were found directly underneath or within 1 m of the location of a large plant (a plant with canopy diameter > 1 m; Figure 6). Of the remaining 30%, approximately 25% were found more than 1 m from a previously removed large plant and 6% were > 10 m from a previously removed large plant, indicating likely vector dispersal (Figure 6). The maximum distance between a seedling and large plant was 284 m. These measurements do not factor in the distance between the large plants found south of the main infestation area, as they were all either large or layered plants that had not produced seed (based on seed bank assessment).

Capitula in flower or seed

The timing of plant surveys to coincide with optimal detection meant that bitou bush was not usually in the peak of flowering. Bitou bush produces flowers throughout the year (Scott 1996), with a peak in June to September. We were unable to determine the flowering phenology and reproductive output of plants because the primary objective was to kill plants as soon as found, preventing further seed set.

Aerial photography surveys

Examination of aerial photography in 2016 occurred after the majority of foot surveys had been completed. We focused on areas to the north of the study area that had eye-height vegetation difficult to survey. Plants that could potentially be bitou bush were identified and located *in situ*. All were *Schinus terebinthifolia* Raddi. In 2018 a survey of aerial photography was made of the entire survey area (Figure 1f) and adjacent areas

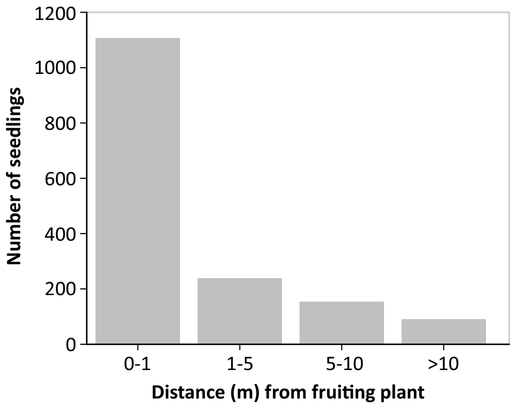

Figure 6. Distance of seedlings from the centre of the nearest large bitou bush canopy (> 1 m in maximum diameter). Average distance over the population 7.7 ± 0.7 m. Maximum distance was 284 m.

(369 ha in total). This virtual survey systematically grid checked photos of the study area over two hours. Ten potential bitou bush were identified. Ground searches found that none were bitou bush, being instead *Acacia*, *Eucalyptus* or *Schinus* plants.

Plant removals

All plants discovered during the six surveys between 2012 and 2018 – a total of 1766 individuals - were killed as part of the control program (Suppl. material 1: Table S1). Seedlings that most likely germinated in the 12 months preceding the annual surveys comprised 61% of the plants. Many plants were still quite small (i.e. < 1 m in maximum canopy diameter), but these were often observed with flowers, despite probably germinating between 12 and 24 months preceding the survey. Overall 9% and 4% (61) of plants respectively had canopy diameters greater >1 m and > 3m. Most of the plants were weeded by hand, though a number were sprayed by herbicide (arranged by landholders) or removed by heavy machinery.

Seed bank studies

Seeds were consistently found in soil cores taken under the plant canopy (Figure 7). However, no intact seeds were found in the soil cores that were taken 1 m from the perimeter of the plant canopy in 2013, indicating that most seeds remained within the canopy area or vectors transport the seeds beyond this short distance (potential vectors are listed in Suppl. material 1: Table S2). Of the nine original plants that were able to be sampled for seed bank studies throughout the project, the viable under-canopy seed bank averaged 39.3 ± 11.4 seeds.m^{-2} in 2013, decreasing to 5.7 ± 2.2 in 2017 (Figure 7).

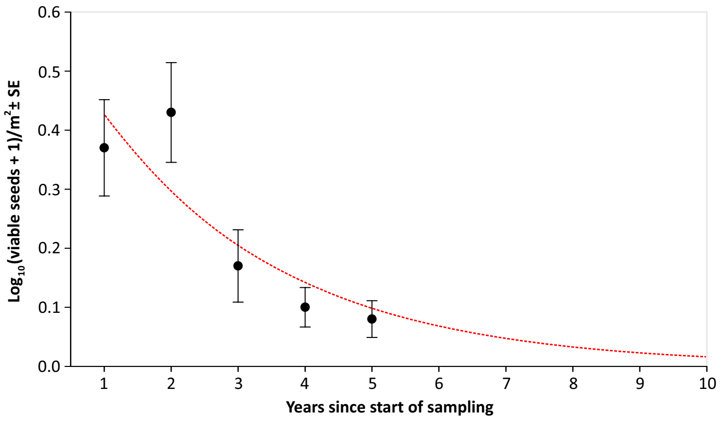

Figure 7. Density of viable seeds in the bitou bush seed bank at Kwinana between 2013 and 2017. Exponential decay trend (dotted line): Y = -0.72e$^{-0.41X}$, R^2 = 0.76, df = 4, $p < 0.05$.

Discussion

Source of the infestation

Bitou bush was possibly originally introduced into Australia in ship ballast, but was planted in the 1940s to 1960s for dune stabilisation in New South Wales (NSW) (Weiss et al. 2008). It is not known for certain how the Kwinana population arrived in Western Australia, but based on plant size (Scott 1996) and the presence of bitou plants in aerial photography (Scott et al. 2019) it is certain that they were present at Kwinana in 1995 (17 years before detection in 2012). The Kwinana area has well established transport linkages to NSW, through the use of the port area by Broken Hill Propriety Co. Ltd until April 1995 (BHP Steel International Group – Rod and Bar Products Division 1995). Another of BHP steel's sites at the time was Port Kembla, NSW, an area with an abundant bitou bush infestation (Love 1986). We hypothesise that that transport of goods between Port Kembla and Kwinana enabled the introduction of bitou bush seed. The other alternative is for plants to have arrived from southern Africa by ship and somehow been transported to shore, but at this stage we have not identified a putative vector.

It might be expected that the foci of the invasion would result in larger plants clustered together. But this is not the case, except for two of the largest plants near the old jetty. It is possible that a single point of introduction occurred and that some of the initial plants that germinated were planted as part of gardens that are now abandoned (bitou bush was used as a dune stabilizing plant). Molecular studies will be the best way to elucidate the origins of the Kwinana population, and may provide useful biosecurity guidance depending on the putative source population(s).

The most likely dispersal vectors for bitou bush seed in the population were foxes, mice, doves and parrots (Suppl. material 1: Table S2). No birds were observed on the bushes during the entire period of surveys, but bees were seen visiting flowers. Elsewhere in Australia and southern Africa, seed dispersal is mainly by birds and some mammals ingesting fruits and carrying seeds to new locations (Gosper 1999; Meek 1998; Weiss et al. 2008).

Current versus past presence of bitou bush

While it is possible that some plants could have disappeared from the population as part of site works, the documented history of little change to the landscape, in particular in the main infestation area, indicates that we have located every plant to have ever been present, both in space and time (excluding seeds and small seedlings of course). This conclusion is supported by the aerial photography which did not show additional plants in the past. It is also reasonable to suggest that all plants that have ever germinated and survived past the seedling stage are included in this study. The level of temporal and spatial population data collected here opens the possibility of modelling the invasion process once we have a better knowledge of the breeding system and number of introductions, as indicated by a study of the genetic diversity.

Bitou bush missed and subsequently found in surveys

The reduction in the number of new plants found in the most recent survey in 2018 is encouraging, but the discovery of large plants clearly missed in earlier surveys is of concern to all involved in the control program. The 2018 survey discovery of two bitou bush plants completely contained within the canopy of a large *Acacia cyclops* emphasised the role of detectability in eradication programs (Cacho et al. 2006; Regan et al. 2006). These similarly sized bitou bush plants probably germinated in 2013–2014, had grown underneath the acacia canopy and were not visible before about August 2017 (the area was surveyed each year; date of detectability within the acacia canopy based on aerial photographs). These discoveries show that despite experience and strategically planned and comprehensive surveying, plants can be difficult to find amongst other vegetation with similar leaf and flower colour (e.g. *Acacia saligna* and *Nicotiana glauca*). It also seems possible that a highly favourable climate over summer (e.g. the summer of 2016–2017) facilitated the growth within a year of previously undetectable small plants into larger flowering individuals.

Seed bank studies

The exponential decay of viability in the sampled seeds over 5 years shows a similar pattern to most seeds after a disturbance (e.g. soil cultivation); a peak in germination followed by steadily declining viability over time (Long et al. 2014). There was no significant decrease in seed viability in 2015–2017. However, extrapolation of the trend line indicates that the seed bank would not be completely depleted until at least 2020 or perhaps longer (Figure 7), depending on a number of caveats and assumptions.

The same pattern of seed distribution seen at Kwinana (i.e. large numbers of seed under the canopy, none 1 m away from the canopy) was observed over 12 sites in the native habitat of *C. monilifera* in South Africa (Scott 1996). The most striking difference between the seed bank at Kwinana population and the seed bank in South Africa (Scott 1996) is the absence of specialised seed predators at Kwinana. One of these (*Mesoclanis* sp.; Diptera) is now used as a biological control agent in eastern Australia (Edwards et al. 2009). In contrast, the South African populations of bitou bush had a low level of rodent predation, whereas this occurs in the Kwinana population at a frequency of about 1 to 3 seeds per m^2.

Seed production by isolated plants

No seed fragments were found in the soil in samples taken underneath the six plants on the outskirts of the infestation (yellow dots; except 791 and 792; Figure 2), despite these plants having flowered prior to the survey period. However, in 2018 seeds were found on two plants found growing together, hidden inside an Acacia plant (791 and

792 in Figure 2). No seed or seed fragments were found in the soil underneath these plants, indicating that seeds had not been produced in previous years. When found in April 2018, these two plants were in flower (which is why they were detected) and seed production had commenced. Mature seed still attached to the plant and covered with the dried remains of the fruit had 100% germination (sample of 10 seeds). All seeds on these plants were collected, but that does not exclude the possibility of seed dispersal by vectors such as birds. One other plant in 2018 had a few seeds, but a nearby plant for potential cross-pollination was not found.

Taken together, these observations provide circumstantial support for bitou bush being an obligate outcrossing species, perhaps with a very low level of selfing. Gross et al. (2017) performed cross pollination experiments on six populations from eastern Australia and found that in all populations bitou bush was an obligate outcrossing species. This implies that invasion would be more difficult than if the species was self-fertile (Baker's law; Baker 1955), although there are many examples of colonists with obligate outcrossing and the ideal of a "law" is questioned (Pannell et al. 2015). The key to this issue in the case of bitou bush is dispersal vectors. If seed and fruit size are a sufficiently small "package" at consumption so that more than one seed is deposited in the vector's droppings, and/or if there are perch sites that are frequently visited, then the requirement for outcrossing is unlikely to be a constraining factor because the chances are high that more than one plant will be growing in the same location.

Seed germination affected by allelopathy

While there has been a steady decline in the seed bank, a spike in seedlings in 2015 (Figure 3), indicates a possible wearing-off of allelopathic effects that prevent germination under the parent bush. Ens et al. (2009) found bitou bush roots and leaves to be phytotoxic and allelopathic when applied to five native species, and the root extracts are likely to have a long residence time in soil, particularly soils with little humic matter such as in the sand dunes. Structured germination trials would be required to confirm that allelopathy plays a role in germination inhibition in bitou bush.

Seed bank longevity

There was an average of 39.3 viable seeds per m^2 under bitou bush canopies in at the beginning of the management program in 2013. The number of plants that were large enough to be reproductive came to 151, representing a total canopy area of 1670 m^2 at the Kwinana site. Multiplying the seed density by the canopy area gives a total of 65,629 seeds in 2012. Using an exponential decline function (Figure 7) and extrapolating forward, the first year with zero seeds is 2020. There are a lot of assumptions in this calculation, including no further input into the seed bank, and an absence of bird dispersed seed. Indeed, while most seed production ceased in 2012, seeds were produced in 2018 on three plants, and there is a remote possibility of seeds entering the seed bank that year. Even so, the density of seeds is now so low that sampling via soil cores to assess seed bank viability has become unfeasible. Extrapolating from earlier core data (Figure 7) indicates that there is potentially a long tail of viable seeds (at low density), for at least another three to four years.

Seed viability over time for bitou bush is known to be variable. In this study, seed collected in 2012 and stored in the laboratory did not germinate in 2018 (unpublished results, bearing in mind the difficulty of extrapolating results generated in laboratory conditions to field situations). In eastern Australia, bitou bush seeds have remained viable in the soil for up to seven years (Kristine French, pers. comm.). However, surveys, such as currently practiced, will detect germination from the seed bank either as seedlings in the year of germination or as plants in the following year when they are more obvious. Taken together, these results indicate that seed longevity data from lab trials or natural populations (native or introduced) elsewhere may not apply to the Mediterranean environment of the Kwinana population. More complex modelling of seed bank decline trajectories and their uncertainties may produce a more robust understanding of required monitoring timelines for bitou bush at Kwinana. With the information currently available, a more realistic estimate is that the Western Australian population could persist via a viable seed bank until at least 2024. As such, monitoring and active management until this time is essential for achieving eradication.

Has delimitation been achieved?

Knowledge of the spatial and temporal extent of a weed's incursion is critical to any eradication effort (Panetta and Lawes 2005). Very few weeds have been eradicated from Australia, with most cases failing because the spread of infestations to areas well outside the target area. While it is not possible to say with absolute certainty that delimitation of bitou bush has been achieved in Kwinana, as there are many avenues where soil potentially containing seed could have moved offsite (Scott and Batchelor 2014), it is very encouraging that no bitou bush have been found in Western Australia outside of the known infestation area (Figure 1) in the seven years since first detection. The conclusion is reinforced by the non-detection of bitou bush in widespread monitoring in Western Australia for the other subspecies in Australia (boneseed *Chrysanthemoides monilifera* subsp. *monilifera*; see Figure 2 in Scott and Batchelor (2014)).

Feasibility of eradication

Bitou bush is subject to localised eradication to implement broader containment at the southern and northern extremes of its range in eastern Australia. These management programs have achieved mixed results. At the southern end in Victoria, bitou bush has been eradicated from two locations, Kew and Frankston (Adair and Butler 2010), although the current view on this is that local eradication may not have been achieved (Royal Botanic Gardens Victoria 2017). Two further populations, Daveys Bay and Mallacoota, are undergoing eradication, with Mallacoota well advanced (in 2010). An extra challenge for the Mallacoota population is the hybridization with the sibling species, *C. m.* subsp. *monilifera* (boneseed; Adair and Butler 2010). The Victorian eradication is complemented by a "containment line" in southern New South Wales (Cherry et al. 2008).

Queensland has had a long running eradication and containment program against bitou bush in the northern extent of its range, starting in 1982. After 10 years of control effort the 700 ha infestation was reduced to a few small infestations. By 2007 few scattered plants remained which are removed in annual surveys. Effectively this is a containment program to limit spread to the north (Wilson et al. 2017), which will need to run forever given the large population of bitou bush just to the south in New South Wales (Cherry et al. 2008). Recently Behrendorff et al. (2019) described progress since 1982 towards eradication of bitou bush on K'gari-Fraser Island in south east Queensland, where the plant is at the limits of its northern expansion. While the seed bank was not measured, counts of seedlings recorded each year at one fixed location indicated a potential seed survival of six years. However, a delimitation survey was not described and bitou bush plants were still being found after 20 years of survey.

Aside from Behrendorff et al. (2019) there is little published quantitative data from these eradication programs (a general problem of eradication projects noted also in New Zealand; Howell 2012), which makes it difficult to adopt lessons learnt or to take on improvements to techniques, or even to estimate how long eradication will take (and consequently the cost). The lack of such data to inform areas for improving management was raised as a priority by Wilson et al. (2017). In 2018 bitou bush is still managed as an eradication target in Queensland (Queensland Department of Agriculture and Fisheries 2018) and in Victoria (Royal Botanic Gardens Victoria 2017).

Indicators that eradication could be achieved in Western Australia, at least locally in Kwinana, are the decline to zero observed seedlings in 2018 and substantial areas where plants have not been seen for three years (Figure 1 d,e,f). Likewise, the decline in plants detected per hectare and per day of sampling effort also indicate that eradication could be achievable within a few years. The overall survey area was about 250 ha, well within the maximum feasible survey area (1000 ha) for which Panetta and Timmins (2004) and Rejmanek and Pitcairn (2002) indicate eradication could be achieved. Supporting this are the soil seed bank studies indicating that the number of viable seeds is approaching zero. Also supporting the feasibility of eradication is bitou's reproductive system, where it appears that cross pollination is necessary (Gross et al. 2017). This means that isolated plants can become large enough for detection, often by the presence of flowers, without the risk of contributing to the seed bank.

The main counter indication against feasibility of eradication is the continued detection each year of a few large plants, especially the detection in 2018 of two reproductive individuals (791 and 792, Figure 2), which potentially resets the clock of seed persistence. Detectability is a critical element of an eradication project (Panetta and Timmins 2004). Monitoring over the next couple of years will establish if detectability is the main risk. It is evident that small plants (or seedlings) are not always detected during the surveys, but are detected when emerging through the surrounding vegetation. Plants also come into flower within a year (Weiss et al. 2008). Early flowering has been noted in other eradication targets (Panetta 2015), resulting in the need to change survey frequency. Annual surveys in March or April still remain the best way of detecting these plants before seed set. Finally, a contra-indication for eradication continues to be the inability to detect small plants in aerial surveys. In addition, the eradication effort is made complicated by the multitude of landholders, unmanaged land where the vegetation is dense, and the inability to use fire as a management tool because of the proximity of petro chemical plants. Fire forms an important part of bitou bush management in eastern Australia (Lindenmayer et al. 2015).

Eradication program feasibility insight

The area of occupancy for bitou bush in Western Australia was small, and delimitation was defined early and has not changed despite the possibility of seed being moved by soil (Scott and Batchelor 2014). The ongoing delimitation of this population continues to be a critical element defining the eradication attempt and making possible the planning for surveys (Panetta and Lawes 2005). Initially we thought the target species would be easily detected among other vegetation, but that proved to be overly optimistic. A range of survey techniques, including aerial photography, was required to achieve necessary detection probabilities. The soil seed bank confounds the detection process because it is out of sight, but must be measured to establish that decline is occurring. Ongoing population recruitment via seedlings does not give this information, as germination is dependent on a wide range of factors that vary between years (Long et al. 2014). Fortunately, the infestation was not far from the main city, which reduced the costs of the management program. It remains to be seen if sufficient resources are available for the full eradication program, especially in the final stages when additional resources and care is required because the target abundance is very low (Wilson et al. 2017).

Buddenhagen and Tye (2015) list five barriers to achieving eradication based on a review of work carried out on the Galapagos Islands. In the following the barriers are assessed against the situation with bitou bush. Barrier 1 (insufficient effort) does not appear to be the case (Figure 4). There is a considerable literature on control of bitou bush so Barrier 2 (no control technique) is not likely. Barrier 3 (permission) is particularly relevant given the range of tenure at Kwinana (Scott and Batchelor 2014) and access to highly secure areas is an ongoing annual challenge, complicated by change in land ownership and changes in personnel. Barrier 4 is size and time. Both continue to be relevant, especially now that the plant (and seeds in the soil) have become rare at the study area and the danger of a false sense of confidence threatens ongoing investment to pursue the eradication program to an appropriate end point. Barrier 5 (available resources) will continue to be an issue as it is for any management strategy. Lastly, Barrier 6 (commitment and understanding) is possibly the most important and least quantifiable, and one where the strength of the collaboration between land managers and those leading the eradication program is critical.

How will we know that eradication has been achieved?

Of course we can never be certain that eradication has been achieved. Surveillance will need to continue once there are no plants at the infestation site and it will take some years before eradication can be confidently declared (even if uncertain). Various time periods have been proposed such as three years, based on a review of all eradication programs in New Zealand (Howell 2012) to five years (Rejmanek and Pitcairn 2002). Another reasonable suggestion is to monitor for the same duration as the seed bank longevity (in this case six to seven years; Dodd et al. 2015). The New Zealand experience is that eradication is more difficult than initially anticipated, so it would be wise not to under-estimate the resources and duration required (Howell 2012). Economic criteria have been proposed to define where monitoring stops, when the expected costs outweigh the expected benefits (Regan et al. 2006; Moore et al. 2011), but can this be meaningfully applied to weeds where the main impact is on the environment? Once we have zero plants present in annual surveys then we can realistically start to estimate the duration to eradication, for example, using methods such as developed in Rout et al. (2009).

Reflecting on one of the few successful weed eradications, *Bassia scoparia* in Western Australia, the plant was detected for eight years after the program began. Surveys concluded at each site if the plant was not detected for three years (Dodd and Randall 2002). However, *B. scoparia* was known to have short lived seeds, in the order of 1–3 years, with the soil seed bank largely exhausted after 12 months (Dodd and Randall 2002). Bitou bush in Western Australia can hope for a similar level of success, if the commitment to management of this population is maintained.

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References

Adair RJ, Butler KL (2010) Hybridisation within *Chrysanthemoides monilifera* and implications for the eradication of bitou bush from Victoria, Australia. Proceedings of the 17th Australasian Weeds Conference, 26–30 September, 2010, Christchurch, 378–381.

Baker HG (1955) Self-compatability and establishment after "long distance" dispersal. Evolution 9: 347–348. https://doi.org/10.2307/2405656

Behrendorff L, Harris SM, Muirhead IF (2019) Towards eradication: The history and management of bitou bush on K'gari-Fraser Island, Australia. Ecological Management & Restoration Jan 2019: 1–9. https://doi.org/10.1111/emr.12349

BHP Steel International Group – Rod and Bar Products Division (1995) A place in history: BHP Steel, Rod and Bar Products Division, Kwinana Market Mill April 1956 – April 1995. BHP, Kwinana.

Blackburn TM, Pysek P, Bacher S, Carlton JT, Duncan RP, Jarosik V, Wilson JRU, Richardson DM (2011) A proposed unified framework for biological invasions. Trends in Ecology & Evolution 26: 333–339. https://doi.org/10.1016/j.tree.2011.03.023

Buddenhagen CE, Tye A (2015) Lessons from successful plant eradications in Galapagos: commitment is crucial. Biological Invasions 17: 2893–2912. https://doi.org/10.1007/s10530-015-0919-y

Cacho OJ, Spring D, Pheloung P, Hester S (2006) Evaluating the feasibility of eradicating an invasion. Biological Invasions 8: 903–917. https://doi.org/10.1007/s10530-005-4733-9

Cherry H, Willsher L, Whyte B (2008) Bitou bush and boneseed eradication and containment in Australia. Plant Protection Quarterly 23: 38–40.

Dodd AJ, Ainsworth N, Burgman MA, McCarthy MA (2015) Plant extirpation at the site scale: implications for eradication programmes. Diversity and Distributions 21: 151–162. https://doi.org/10.1111/ddi.12262

Dodd J, Randall RP (2002) Eradication of Kochia (*Bassia scoparia* (L.) A.J. Scott, Chenopodiaceae) in Western Australia. In: Spafford-Jacob J, Dodd J, Moore JH (Eds) Proceedings of the 13th Australian Weeds Conference, Perth, Western Australia, 8–13 September, 2002, 301–303.

Edwards PB, Adair RJ, Holtkamp RH, Wanjura WJ, Bruzzese AS, Forrester RI (2009) Impact of the biological control agent *Mesoclanis polana* (Tephritidae) on bitou bush (*Chrysanthemoides monilifera* subsp. *rotundata*) in eastern Australia. Bulletin of Entomological Research 99: 51–63. https://doi.org/10.1017/S000748530800610X

Ens EJ, French K, Bremner JB (2009) Evidence for allelopathy as a mechanism of community composition change by an invasive exotic shrub, *Chrysanthemoides monilifera* spp. *rotundata*. Plant and Soil 316: 125–137. https://doi.org/10.1007/s11104-008-9765-3

Gosper CR (1999) Plant food resources of birds in coastal dune communities in New South Wales. Corella 23: 53–62.

Gross CL, Whitehead JD, de Souza CS, Mackay D (2017) Unsuccessful introduced biocontrol agents can act as pollinators of invasive weeds: Bitou Bush (*Chrysanthemoides monilifera* ssp. *rotundata*) as an example. Ecology and Evolution 7: 8643–8656. https://doi.org/10.1002/ece3.3441

Howell CJ (2012) Progress toward environmental weed eradication in New Zealand. Invasive Plant Science and Management 5: 249–258. https://doi.org/10.1614/IPSM-D-11-00001.1

Lindenmayer DB, Wood J, MacGregor C, Buckley YM, Dexter N, Fortescue M, Hobbs RJ, Catford JA (2015) A long-term experimental case study of the ecological effectiveness and cost effectiveness of invasive plant management in achieving conservation goals: Bitou bush control in Booderee National Park in eastern Australia. PLoS ONE 10(6): e0128482. https://doi.org/10.1371/journal.pone.0128482

Long RL, Gorecki MJ, Renton M, Scott JK, Colville L, Goggin DE, Commander LE, Westcott DA, Cherry H, Finch-Savage WE (2014) Seed persistence: The journey from dispersal to germination. Biological Reviews 90: 31–59. https://doi.org/10.1111/brv.12095

Love A (1986) Distribution of bitou bush along the N.S.W. coast. In: Love A, Dyason R (Eds) Bitou bush and boneseed: Proceedings of a Conference on *Chrysanthemoides monilifera*. NSW National Parks and Wildlife Service and NSW Department of Agriculture, Sydney, 8–9 August 1984, Port Macquarie, NSW, 53–64.

Meek PD (1998) 'Weed seeds and whoopsie daisies': viability of bitou bush *Chrysanthemoides monilifera* seeds in fox (*Vulpes vulpes*) scats. Plant Protection Quarterly 13: 21–24.

Moore JL, Runge MC, Webber BL, Wilson JRU (2011) Contain or eradicate? Optimizing the management goal for Australian acacia invasions in the face of uncertainty. Diversity and Distributions 17: 1047–1059. https://doi.org/10.1111/j.1472-4642.2011.00809.x

Myers JH, Savoie A, van Randen E (1998) Eradication and pest management. Annual Review of Entomology 43: 471–491. https://doi.org/10.1146/annurev.ento.43.1.471

Panetta FD (2004) Seed banks: the bane of the weed eradicator. In: Sindel BM, Johnson SB (Eds) Proceedings of the 14th Australian Weeds Conference, Wagga Wagga, New South Wales, 6–9 September 2004, 523–526.

Panetta FD (2007) Evaluation of weed eradications programs: containment and extirpation. Diversity and Distributions 13: 33–41.

Panetta FD (2015) Weed eradication feasibility: lessons of the 21st century. Weed Research 55: 226–238. https://doi.org/10.1111/wre.12136

Panetta FD, Lawes R (2005) Evaluation of weed eradication programs: the delimitation of extent. Diversity and Distributions 11: 435–442. https://doi.org/10.1111/j.1366-9516.2005.00179.x

Panetta FD, Timmins SM (2004) Evaluating the feasibility of terrestrial weed eradications. Plant Protection Quarterly 19: 5–11.

Pannell JR, Auld JR, Brandvain Y, Burd M, Busch JW, Cheptou PO, Conner JK, Goldberg EE, Grant AG, Grossenbacher DL, Hovick SM, Igic B, Kalisz S, Petanidou T, Randle AM, Rubio de Casas R, Pauw A, Vamosi JC, Winn AA (2015) The scope of Baker's law. New Phytologist 208: 656–667. https://doi.org/10.1111/nph.13539

Pejchar L, Mooney HA (2009) Invasive species, ecosystem services and human well-being. Trends in Ecology & Evolution 24: 497–504. https://doi.org/10.1016/j.tree.2009.03.016

Queensland Department of Agriculture and Fisheries (2018) Bitou bush *Chrysanthemoides monilifera* sub-species *rotundata*. https://www.daf.qld.gov.au/__data/assets/pdf_file/0018/61344/IPA-Bitou-Bush-PP10.pdf [accessed 29 April 2018]

Regan TJ, McCarthy MA, Baxter PWJ, Panetta FD, Possingham HP (2006) Optimal eradication: when to stop looking for an invasive plant. Ecology Letters 9: 759–766. https://doi.org/10.1111/j.1461-0248.2006.00920.x

Rejmanek M, Pitcairn MJ (2002) When is eradication of exotic pest plants a realistic goal? In: Vietch CR, Clout MN (Eds) Turning the tide: The eradication of island invasives. IUCN SSC Invasive species Specialist Group IUCN, Gland, 249–253.

Rout TM, Salomon Y, McCarthy MA (2009) Using sighting records to declare eradication of an invasive species. Journal of Applied Ecology 46: 110–117. https://doi.org/10.1111/j.1365-2664.2008.01586.x

Royal Botanic Gardens Victoria (2017) *Chrysanthemoides monilifera* subsp. *rotundata* (DC.) Norl. Bitou Bush. https://vicflora.rbg.vic.gov.au/flora/taxon/256a3a83-1950-4ed0-b1e3-f4637adc1d74 [accessed 30 April 2018]

Scott JK (1996) Population ecology of *Chrysanthemoides monilifera* in South Africa: Implications for its control in Australia. Journal of Applied Ecology 33: 1496–1508. https://doi.org/10.2307/2404788

Scott JK, Batchelor KL (2014) Management of *Chrysanthemoides monilifera* subsp. *rotundata* in Western Australia. Invasive Plant Science and Management 7: 190–196. https://doi.org/10.1614/IPSM-D-13-00052.1

Scott JK, Batchelor KL, Jucker T, Webber BL (2019) Aerial photography and dendrochronology as tools for recreating invasion histories: do they work for bitou bush, *Chrysanthemoides monilifera* subsp. *rotundata?* Biological Invasions 21(9): 2983–2996. https://doi.org/10.1007/s10530-019-02026-5

Shackleton RT, Shackleton CM, Kull CA (2019) The role of invasive alien species in shaping local livelihoods and human well-being: A review. Journal of Environmental Management 229: 145–157. https://doi.org/10.1016/j.jenvman.2018.05.007

Weiss PW, Adair RJ, Edwards PB (1998) *Chrysanthemoides monilifera* (L.) T. Norl. In: Panetta FD, Groves RH, Shepherd RCH (Eds) The biology of Australian weeds Volume 2. R.G. and F.J. Richardson, Melbourne, 49–61.

Weiss PW, Adair RJ, Edwards PB, Winkler MA, Downey PO (2008) *Chrysanthemoides monilifera* subsp. *monilifera* (L.) T.Norl. and subsp. *rotundata* (DC.) T.Norl. Plant Protection Quarterly 23: 3–14.

Wilson JR, Panetta FD, Lindgren C (2017) Detecting and responding to alien plant incursions. Cambridge University Press, Cambridge. https://doi.org/10.1017/CBO9781316155318

Supplementary material 1

Supplementary tables

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Data type: species data

Explanation note: **Table S1.** Number of individuals and average diameter of plant canopy (note this excludes seedlings) at Kwinana. Survey effort reflects the number of people days (i.e. 6 days could equal 2 people × 3 days, or a combination thereof). **Table S2.** Potential frugivores, granivores and/or seed dispersal agents amongst vertebrate animals observed on the Kwinana Industrial Estate in bitou bush infested areas. Daytime observations only.

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