RESEARCH REPORT

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Impacts on nontarget avian species from aerial meat baiting for feral pigs

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Summary Bait containing sodium fluoroacetate (1080) is widely used for the routine control of feral pigs in Australia. In Queensland, meat baits are popular in western and northern pastoral areas where they are readily accepted by feral pigs and can be distributed aerially. Field studies have indicated some levels of interference and consumption of baits by nontarget species and, based on toxicity data and the 1080 content of baits, many nontarget species (particularly birds and varanids) are potentially at risk through primary poisoning. While occasional deaths of species have been recorded, it remains unclear whether the level of mortality is sufficient to threaten the viability or ecological function of species. A series of field trials at Culgoa National Park in south-western Queensland was conducted to determine the effect of broadscale aerial baiting (1.7 baits per km^2) on the density of nontarget avian species that may consume baits. Counts of susceptible bird species were conducted prior to and following aerial baiting, and on three nearby unbaited properties, in May and November 2011, and May 2012. A sample of baits was monitored with remote cameras in the November 2011 and May 2012 trials. Over the three baiting campaigns, there was no evidence of a population-level decline among the seven avian nontarget species that were monitored. Thirty per cent and 15% of baits monitored by remote cameras in the November 2011 and May 2012 trials were sampled by birds, varanids or other reptiles. These results support the continued use of 1080 meat baits for feral pig management in western Queensland and similar environs.

Key words: *feral pig, invasive species, natural resource management, poison.*

Introduction

Feral pigs damage the environment and are responsible for economic losses in agricultural enterprises across much of Australia (see Bengsen *et al.* 2014), which ultimately triggers control by land managers. Control techniques utilised include trapping, aerial and ground shooting, commercial harvesting and recreational hunting (Mitchell 2008; Gentle & Pople 2013), but baiting with sodium fluoroacetate (1080) typically remains the most widely used and efficient technique to manage pig populations (Mitchell 2011; Bengsen *et al.* 2014). In Queensland, broadscale management of feral pigs, particularly in northern and western pastoral areas, remains heavily reliant upon aerial shooting or distribution of 1080 meat baits. Meat baits comprise ~75% of the bait material distributed per year for feral pig control in Queensland, with large amounts (>50 tonnes) of meat baits applied annually (see Fig. 1). Meat baits are assumed to represent naturally occurring food items (e.g. carcasses) and therefore be readily accepted by pigs. Feral pigs have a relatively high tolerance of 1080 (LD$_{50}$ in meat bait ~2.45 mg/kg body weight Gentle *et al.* 2008) and a large adult body size (25–175 kg; Van Dyck & Strahan 2008), so large doses of 1080 are required to

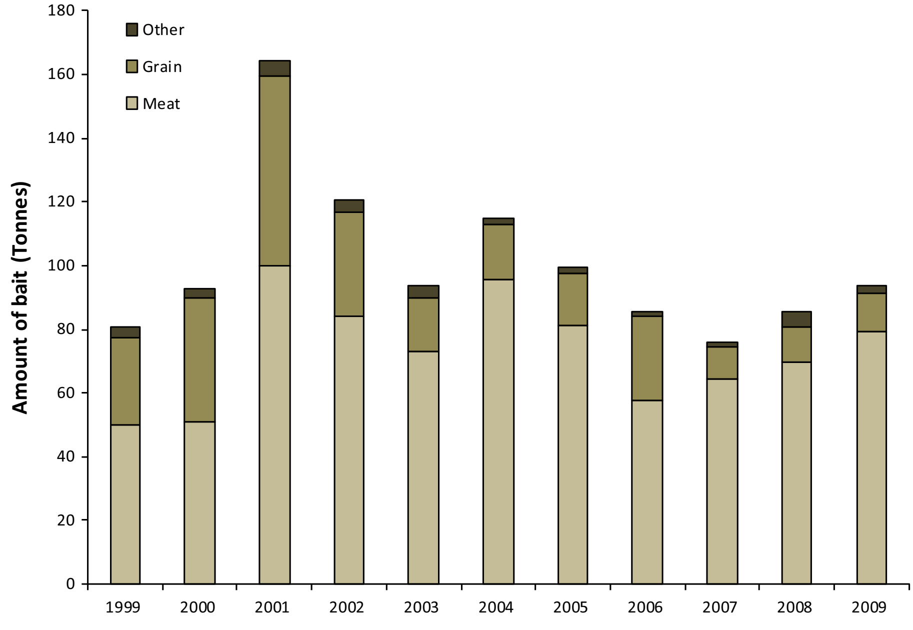

Figure 1. The amount of meat, grain and other (including fruit, vegetable and PIGOUT®) bait mixed with 1080 to control feral pigs in Queensland, 1999–2009. Source: Queensland Department of Agriculture, Fisheries and Forestry.

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ensure it is lethal. Meat baits used to control feral pigs consist of ~500 g of boneless red meat containing 72 mg of 1080. While such baits may be effective for reducing pig populations (Mitchell 1998), the high 1080 content poses a potential poisoning risk to nontarget consumers.

Based on published species' sensitivity to 1080 and the 1080 content in feral pig meat baits, at least 20 native Australian bird species are at risk through primary poisoning (McIlroy 1984). Birds that are likely to consume pig meat baits and would be susceptible to their 1080 content include the Australian Raven (*Corvus coronoides*), Australian Magpie (*Cracticus tibicen*), Pied Currawong (*Strepera graculina*), Black Kite (*Milvus migrans)* and Wedge-tailed Eagle (*Aquila audux*) (McIlroy 1983). Some species of varanid, particularly the Gould's Goanna (*Varanus gouldii*) are also at risk. Table 1 shows 1080 toxicity for some potential bait consumers.

A review of 1080 use in Australia highlighted the nontarget risk from pig meat baits (APVMA 2008). The surface distribution of baits, often by aircraft over extensive areas, makes it likely that many carnivorous species (particularly raptors) will encounter baits. These theoretical concerns are supported by field observations of uptake of meat baits by nontarget species (Hone & Pedersen 1980; McIlroy 1983; Fleming *et al.* 2000) and, in some cases, toxicology analyses have confirmed 1080 poisoning as the cause of death (R. Parker, Biosecurity Queensland, 2013 pers. comm.) Occasionally, bird carcasses (primarily raptors) have been found following baiting operations (e.g. Hone & Pedersen 1980; McIlroy 1983), but typically, few deaths have been recorded. Differences in methodology, such as bait size (~140 g, Fleming *et al.* 2000; 190 g, Hone & Pedersen 1980; ~500 g, current study), the use of nontoxic bait and distribution strategies (e.g. ground *vs* aerial, bait density), may all affect whether nontarget species find and consume bait, making comparison and interpretation of such findings difficult. Nevertheless, such information identifies species at risk.

In a review of nontarget impacts of predator baiting, Glen *et al.* (2007) described the evidence of impact on nontarget species as ranging from weak (such as sensitivity of a species to the toxin) to strong (such as data indicating that nontarget species can consume meat baits, or observation of some nontarget species deaths). Other factors, such as the relative palatability of the bait to each species, different species' foraging habits, availability of alternative foods, amount of bait (and toxin) consumed by each individual and the proportion of individuals in a population consuming toxic bait, can affect the extent of mortality in a population (McIlroy 1984). Given the difficulty in considering all these factors, one can only reach a definitive conclusion of negative impacts if observed mortality of nontarget species is supported by sustained reductions in population density (Glen *et al.* 2007).

Table 1. Toxicity of 1080 for a sample of potential bait consumers. Data from McIlroy (1983)

Species	Adult body mass (g)	LD_{50} (mg kg^{-1} body weight)	Amount of 1080 (mg) for LD_{50}
Feral Pig (*Sus scrofa*)	55 000	1.04	57.20
Wedge-tailed Eagle (*Aquila audux*)	3100	9.5	29.45
Little Raven *(Corvus mellori*)	560	3.1	1.74
Australian Raven (*Corvus coronoides*)	585	~5.1	~2.98
Australian Magpie (*Cracticus tibicen*)	320	9.91	3.17
Little Crow (*Corvus bennetti*)	400	13.4	5.36
Black Kite (*Milvus migrans*)	560	18.5	10.36
Laughing Kookaburra (*Dacelo novaguineae*)	300	>6.0	>1.80
Gould's Goanna (*Varanus gouldii*)	840	43.6	36.62
Australian Magpielark (*Grallina cyanoleuca*)	90	~6.75	0.61

Data not available for some species of interest.

The potential for nontarget deaths, and the need for field trials to determine the impact of pig-poisoning campaigns on nontarget species, has long been recognised (see McIlroy 1983), but remains unaddressed – despite considerable study of other predator baiting practices (see Glen *et al.* 2007 for review). No trial has monitored 'population-level' impacts on nontarget species from meat baits used for feral pig control. This needs to be examined to ensure that feral pig baiting operations are acceptably target specific, including avoiding any broader ecological consequences (i.e. 'knock-on effects') from nontarget mortality. This study investigates the impact of pig baiting on the density of likely bait-consuming species of avifauna, particularly corvids and raptors, and briefly discusses the management implications for meat baiting practices used for feral pig control.

Methods

The most practical means to determine a population-level impact is to monitor populations of theoretically susceptible bird species for evidence of decline prior to and following baiting campaigns. Bird abundance was monitored before and after 1080 baiting campaigns on treatment (baited) and control (unbaited) sites in south-western Queensland in autumn and spring 2011 and again in autumn 2012.

Study sites

Study sites were located in the semi-arid rangelands of south-western Queensland. Sites consisted of Culgoa Floodplain National Park (28°55′20″S, 146°59′17″E) and three nearby properties (Kulki 28°42′38″S, 147°15′50″E; North Kulki 28°37′54″S, 147°14′03″E; and Tambingey 28°38′17″S, 147° 08′10″E), south of Bollon, south-western Queensland (Fig. 2). Culgoa Floodplain National Park was aerially baited for feral pigs as part of a biodiversity conservation programme and served as the treatment site; Kulki, North Kulki and Tambingey served as control (i.e. nontreatment) sites. Culgoa Floodplain National Park (hereafter Culgoa; 619 km^2) consists of relatively diverse landscapes comprising a floodplain of the

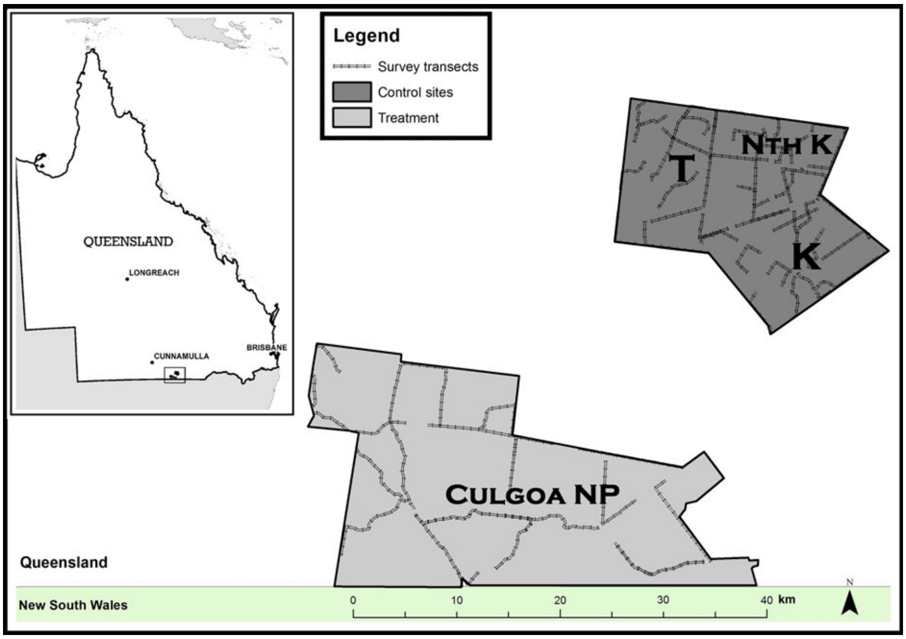

Figure 2. Location of study area in Queensland (inset) and study sites used to monitor bird populations in study area (main). Site treated with 1080 meat baits: Culgoa Floodplain National Park (Culgoa NP). Nontreatment sites: Kulki (K), North Kulki (Nth K) and Tambingey (T). The location of transects used to survey bird populations is also shown.

Culgoa River (with associated Coolabah-dominated flats) with Brigalow and Gidgee, and Mulga and associated communities on the elevated stony ridge country (botanical nomenclature follows Milson 1997). Kulki (122 km^2), North Kulki (115 km^2) and Tambingey (116 km^2) are nearby (~30 km; Fig. 2) independently managed properties predominantly used for cattle and goat production, with small areas under cultivation on Kulki.

Baiting

At Culgoa (the treatment site), feral pig baiting is conducted biannually in autumn and spring (usually April/May and October/November, respectively). Baits were prepared according to the standard for feral pig control in Queensland (DEEDI 2009). Fresh pieces of kangaroo meat with a mean weight of 538 g (SD = 93.5, n = 40) were injected with 2 mL of 36 mg/mL 1080 solution. Baits were distributed by aircraft on parallel, east-west transects systematically placed at 2 km intervals across the study sites. Baits were spaced at ~175–200 m, to provide a bait density of ~1.7 baits per km^2 within the study site boundaries.

To account for any effect of the meat bait alone on changes in bird abundance, nontoxic kangaroo meat was also distributed on the three control sites using identical methods to the Culgoa site to create a procedural control. Nontoxic and toxic baits were distributed on the control and treatment sites, respectively, before midday of the same day.

Bait monitoring

To identify the species consuming bait and potentially at risk through primary poisoning, a sample of forty 1080 baits (representing ~3.7% of total baits deployed) was monitored with cameras at Culgoa during the November 2011 and May 2012 trials. Baits were placed on the ground by hand, on the same day as the aerial bait deployment, in four transect lines of ten baits each. The transect lines were placed in habitats representative of those found within the National Park. Baits were spaced at ~200-m intervals (to simulate aerial bait spacing), at least 30 m from vehicle tracks. Remote digital cameras (Bolyguard®/Scoutguard® SG550) were fixed on nearby fence posts, logs, or trees and orientated to monitor any species interacting with baits. All baits incorporated VHF radio transmitters (~5 g, 150 MHz, Sirtrack, Havelock North, New Zealand) to help locate, and determine the fate (i.e. consumed, not consumed) of any baits removed. Baits and monitoring cameras were checked on multiple occasions over either a 34-day (May 2012) or a 72-day (November 2011) period. The interaction of photographed individuals with each bait was classified as: (i) approached (i.e. approached bait); (ii) investigated (i.e. bait moved); (iii) sampled (i.e. chewed, pecked, torn apart or partially consumed); or (iv) consumed. Categories were not mutually exclusive; higher-level interactions also included the lower-level interactions (e.g. consumed baits were also recorded as approached, investigated and sampled). Consumed baits were not replaced. See Millar (2012) for more details on bait monitoring.

Population monitoring

Densities of selected bird species were monitored using vehicle-based surveys along predefined transects using a selection of roads, tracks and linear features (e.g. fencelines, disused bore drains) representative of the habitats in each study site. Vehicle-based surveys are well suited for surveying bird taxa that are visually conspicuous (Bibby *et al.* 1992; DEWHA 2010; also see Twigg & Kay 1994), such as the species of interest in this study. In addition, the open habitats on the study sites and the need to survey large areas efficiently dictated the use of driven transects. Transects were seven to ten kilometres in length at Culgoa (19 transects), while those at Kulki, North Kulki and Tambingey (23, 22 and 13 transects, respectively) were typically 5 km in length. Transect length was reduced for each of the smaller control sites to maintain sufficient replication at the transect level. Each transect was driven at 10–20 km per hour. Where necessary, the vehicle was stopped to use binoculars to aid identification. Unidentified observations (<1% of those recorded) were excluded from analyses. A digital rangefinder was used to estimate the perpendicular distance from the transect to each bird or group of birds detected.

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Birds were counted on sites before and after each of the three aerial baiting campaigns in May and November 2011, and May 2012. Counts were undertaken within a 10-day period immediately prior to baiting and were initiated 2–3 weeks postbaiting. Bird species monitored were those either identified as potentially carrion- or bait-consuming or known to investigate meat baits and be susceptible to 1080 (Hone & Pedersen 1980; McIlroy 1983; M. Gentle unpublished data). These species included the Australian Magpie, Brown Falcon (*Falco berigora*), Australian Kestrel (*Falco cenchroides*), Wedge-tailed Eagle, Pied Butcherbird (*Cracticus nigrogularis*), Grey Butcherbird (*Cracticus torquatus*), Australian Raven (*Corvus coronoides*), Little Raven (*Corvus mellori*) and Torresian Crow (*Corvus orru*). The latter three corvid species were grouped for analyses due to difficulty in accurately distinguishing them in the field.

Density estimation

To calculate densities of each species, conventional line transect (CLT) analyses were performed in DISTANCE 6.0 (Thomas *et al.* 2010). Line transect methods were chosen as they are more precise and efficient for estimating density of the bird species of interest than point, cue and snapshot counts, and are particularly suited to sampling open habitats (Buckland 2006). Five detection functions were considered: a uniform key function with either a cosine or simple polynomial series expansion, a half-normal key function plus a Hermite polynomial series expansion, or a hazard-rate key function plus a cosine series expansion. Akaike's information criterion (AIC) was used to select the most parsimonious model and number of adjustment terms in the series expansion. A detection function was modelled on data pooled for each site, as sample sizes were generally inadequate (<50) for each sampling period. Detection functions were expected to vary between sites more than survey periods because of differences in vegetation structure. Density estimates were then derived for each survey period using the site-specific detection function. Variance formulae are given by Buckland *et al.* (1993).

Effect of baiting on bird abundance

For each bird species, density estimates prior to baiting were compared to those following baiting in both the treatment and control sites. The change in density for the treatment site (Culgoa) was then compared to the change in the pooled estimate from the control sites to quantify any baiting effect.

A split–split plot ANOVA was used to test for significant differences in mean bird abundance (birds per km^2) between treatments (baited and unbaited), time periods (pre- and post baiting) and baiting campaigns (May 2011, September 2011 and May 2012). The main plot stratum corresponded to the site, subplot the date and sub–sub-plot pre- and post baiting. The residuals were checked for any outliers and violations of the assumption of homogeneity of residual variances. No transformations of the data were necessary. Statistical testing was performed in GenStat (16th Edition), and the level of significance set at 5% for all testing.

Results

Bait monitoring

Species groups recorded interacting with baits during both trial periods are shown in Table 2. Remote camera images recorded that many baits were visited by multiple species, on multiple occasions. Feral pigs were the primary consumer in both trial periods, consuming 15% of monitored baits. Bird species recorded

Table 2. Number of visited baits grouped by interaction category and taxa in the November 2011 (79 days) and May 2012 (34 days) trial periods. Percentages of the total baits laid are shown in parentheses

Trial	Taxon	Approached	Moved	Sampled	Consumed
November 2011 (79 days)	Pig	18 (45)	10 (25)	9 (22.5)	7 (17.5)
	Cat	12 (30)	1 (2.5)	–	–
	Fox	2 (5)	2 (5)	1 (2.5)	1 (2.5)
	Bird	19 (47.5)	9 (22.5)	8 (20)	–
	Varanid	24 (60)	13 (32.5)	3 (7.5)	–
	Other reptile	19 (47.5)	7 (17.5)	1 (2.5)	–
	Small mammal	1 (2.5)	–	–	–
	Echidna	–	–	–	–
	Unknown	16 (40)	4 (10)	3 (7.5)	2 (5)
	Total	40 (100)	26 (65)	19 (47.5)	10 (25)
May 2012 (34 days)	Pig	11 (27.5)	7 (17.5)	6 (15)	5 (12.5)
	Cat	12 (30)	1 (2.5)	1 (2.5)	1 (2.5)
	Fox	7 (17.5)	3 (7.5)	–	–
	Bird	21 (52.5)	6 (15)	6 (15)	1 (2.5)
	Varanid	–	–	–	–
	Other reptile	–	–	–	–
	Small mammal	–	–	–	–
	Echidna	2 (5)	–	–	–
	Unknown	1 (2.5)	1 (2.5)	1 (2.5)	1 (2.5)
	Total	35 (87.5)	15 (37.5)	13 (32.5)	8 (20)
Both trials combined	Pig	29 (36.3)	17 (21.3)	15 (18.8)	12 (15)
	Cat	24 (30)	2 (2.5)	1 (1.3)	1 (1.3)
	Fox	9 (11.3)	5 (6.3)	1 (1.3)	–
	Bird	40 (50)	15 (18.8)	14 (17.5)	1 (1.3)
	Varanid	24 (30)	13 (16.3)	3 (3.8)	–
	Other reptile	19 (23.8)	7 (8.8)	1 (1.3)	–
	Small mammal	1 (1.3)	–	–	–
	Echidna	2 (2.5)	–	–	–
	Unknown	17 (21.3)	5 (6.3)	4 (5)	3 (3.8)
	Total	75 (93.8)	41 (51.3)	32 (40)	18 (22.5)

Categories are not mutually exclusive, for example consumed baits are also recorded in approached, investigated and sampled.

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approaching baits were the Brown Falcon, Australian Magpie, Australian Magpie Lark, Whistling Kite (*Haliastur sphenurus*), Pied Butcherbird, corvids and the White-winged Chough (*Corcorax melanorhamphos*). In the November 2011 trial, eight baits (22.5%) were sampled by birds, specifically corvids (seven) and the Australian Magpie (one). Of the baits sampled by corvids, one was ~20% consumed, with the seven other baits <20% consumed. No baits were entirely consumed by birds. Varanids approached 24 baits (60%) but only sampled three baits (7.5%). One bait was ~30% consumed, and the remaining two ≤10% consumed. One of the baits partially consumed (~30%) by a Gould's Goanna was also sampled by a Shingleback Lizard (*Tiliqua rugosa aspera*), which consumed the remainder of the bait (~70%). Two baits were consumed by unknown species that could not be identified due to camera failure.

In the May 2012 trial, a total of six baits (15%) were sampled by birds including corvids (three baits), White-winged Choughs (two baits), Brown Falcon (one bait) and Whistling Kite (one bait), but the proportion of bait consumed was not assessed. One bait was consumed by a bird (Whistling Kite), and one by an unknown species. No varanids or other reptiles interacted with monitored baits during this trial period.

Pooling data from both trials, 22.5% of baits were sampled by nontarget species, but typically, only a small proportion (<10%) of the bait material was sampled (where this was assessed). The only nontarget species consuming entire baits were one bird (Whistling Kite) and three unknown species. This represents a 1.3% consumption rate by nontarget species, with <4% unknown.

Density estimation

Densities of each bird taxon for each trial period are shown in Figure 3. Over all treatment and control sites, the Australian Magpie (2.2–10.8 birds per km^2) and corvids (0.6–8.2 birds per km^2) were the most abundant bird taxa monitored; densities of each raptor taxon were generally much lower (<2.7 bird per km^2). Collectively, there was a fluctuating, but considerable density of potentially bait-consuming birds in both control (5.8–26.6 bird per km^2) and treatment sites (6.2–12.2 bird per km^2) over the course of the study.

Effect of baiting on bird abundance

The mean change in bird density on control and treatment sites for each baiting period for each taxon monitored is shown in Table 3. A treatment effect is shown as the difference in density changes between treatment and control sites. Comparison of the mean difference of the three trials for each taxon indicates that most species (i.e. five from the seven species/groups monitored) showed a mean increase in density following treatment. There was an overall decline in mean densities of only the Australian magpie and grey butcherbird, consistent with a negative effect of baiting.

ANOVA indicated no significant differences in bird abundance between the treatment and control sites or within sites prior to and following baiting (Table 4). However, the interaction between baiting periods and treatment (treatment versus control sites) approached significance ($P = 0.055$) for corvids indicating an inconsistent response to baiting. The baited area had higher corvid abundance in May 2011, but lower abundance in November 2011 and May 2012 compared to the average of the control sites.

The density of Australian Magpies and Pied Butcherbirds trended to decline between subsequent trial periods across all sites, but again this was not significant ($P = 0.10$ and $P = 0.07$, respectively).

Discussion

We found no evidence that feral pig control with aerially deployed meat baits resulted in any significant changes in the short-term abundance of potential bait-consuming birds at Culgoa National Park. Monitoring individual baits confirmed that many meat baits were approached by nontarget species, particularly corvids, Australian Magpies, raptors and varanids, but few were consumed (only 1.3% of the monitored baits laid were entirely consumed by nontarget species). However, baits were often sampled (partially consumed) by birds (17.5%), varanids and other reptiles (~5%) and unknown species (5%). It was difficult to estimate the amount of bait consumed with confidence because baits will also decline in size and weight from rapid air-drying, and losses from insect consumption and physical breakdown. Nevertheless, the visual estimates of the amount consumed from the November 2011 trial suggest that the proportion consumed is typically low (<10%). Consumption of even 10% of bait (at a nominal 7.2 mg 1080) may well exceed an approximate lethal dose (LD_{50}) for a corvid or Australian Magpie (see Table 1). Extrapolation from LD_{50} may underestimate the risk to nontarget species as some susceptible individuals may be killed by consuming smaller doses, but is useful to highlight species at risk for study. Variations in toxicity and exposure to individual baits with uneven 1080 content demonstrate the difficulty in predicting susceptibility to primary poisoning through bait consumption alone. This supports the need to use more appropriate measures (*viz.* monitoring population-level change) to determine impacts on nontarget populations (Glen *et al.* 2007).

The relatively low rates of bait disturbance by birds (~19%) in our study is similar to that reported by Cowled *et al.* (2006) (~7.5%), but in strong contrast to that reported by Fleming *et al.* (2000), where 58% of meat baits laid were taken by birds. Methodological differences between studies are likely responsible; bait in our study (500 g, 72 mg 1080) was larger than that used by Fleming *et al.* (2000) (~150 g, nontoxic biomarker) and Cowled *et al.* (2006) (250 g unpoisoned meat). Small baits would be more likely to be successfully handled and removed by nontarget species whose body weight is often <1000 g (see Table 1). Bait size is known to be an important factor in nontarget poisonings in possum control operations, with small bait fragments more easily consumed by birds (Eason *et al.* 2011). There are often differences in the uptake and palatability of toxic and nontoxic bait, possibly because of detectability of the toxin (Sinclair & Bird 1984; Gentle

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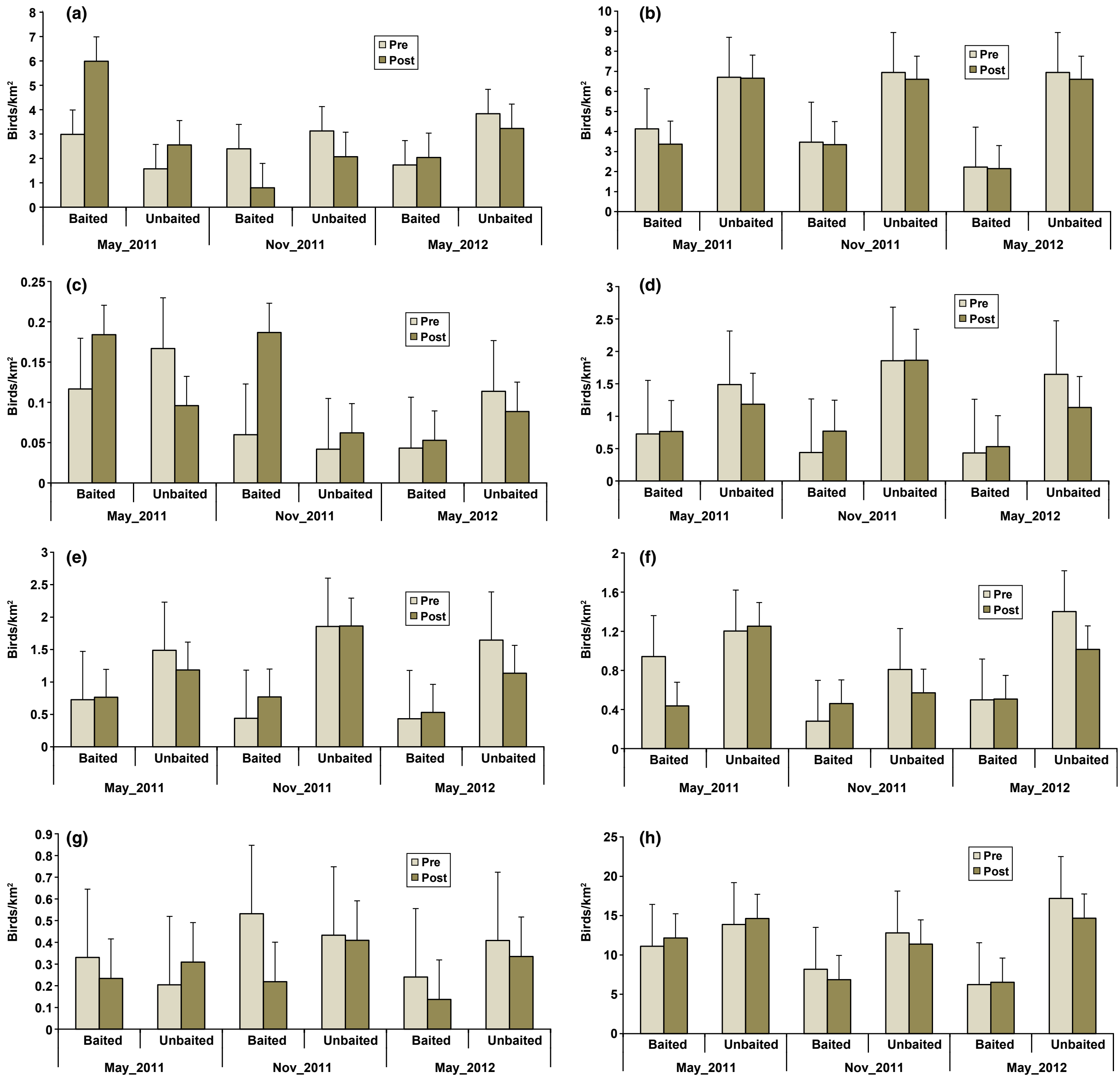

Figure 3. (a–g) Mean densities (+ estimated standard errors) of each bird taxon monitored prior to and following three baiting periods at Culgoa (baited) and Kulki, Tambingey and North Kulki (unbaited) sites. (a) Corvids, (b) Australian Magpie, (c) Wedge-tailed Eagle, (d) Falconidae, (e) Australian Kestrel, (f) Pied Butcherbird, (g) Grey Butcherbird and (h) all bird taxa.

2005). While bait composition and size may have contributed to differences in uptake, it also is likely that site or seasonal differences are contributing factors.

Regardless of the recorded interference of baits by nontargets, the results indicate no consistent, significant declines in bird abundance on the baited site (Culgoa) relative to the unbaited control sites. Several species increased in abundance following baiting, but this response was inconsistent and unrelated to a treatment (baiting) effect, given similar trends on the unbaited control sites. Such fluctuations probably reflect avian species movement in response to local resource availability, which occur regularly but are difficult to predict (Chan 2001). The lack of a negative treatment effect is consistent with the birds monitored here being common and widespread in the semi-arid areas where pigs are routinely controlled with meat baits (www.birdata.com.au). However, we cannot discount effects to other

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Table 3. Change in density of taxa (birds km^{-2}) between pre- and postbaiting surveys on Culgoa (the treatment site) and Kulki, North Kulki and Tambingey (pooled, control sites). Approach follows Westbrooke *et al.* (2003)

Taxon	Trial	Net change by treatment	Culgoa	Kulki, North Kulki, Tambingey
Corvids	May 2011	+2.02	+3.00	+0.98
	Nov 2011	−0.54	−1.60	−1.05
	May 2012	+0.91	+0.31	−0.61
	Mean difference	+0.80	+0.57	−0.23
Australian Magpie	May 2011	−0.72	−0.77	−0.05
	Nov 2011	−0.27	−0.12	+0.16
	May 2012	+0.27	−0.07	−0.34
	Mean difference	−0.24	−0.32	−0.08
Wedge-tailed Eagle	May 2011	+0.14	+0.07	−0.07
	Nov 2011	+0.11	+0.13	+0.02
	May 2012	+0.03	+0.01	−0.03
	Mean difference	+0.09	+0.07	−0.03
Falconidae[1]	May 2011	−0.34	+0.04	+0.38
	Nov 2011	+0.32	+0.33	+0.01
	May 2012	+1.32	+0.10	−1.22
	Mean difference	+0.43	+0.16	−0.28
Australian Kestrel	May 2011	+0.08	+0.01	−0.06
	Nov 2011	+0.14	+0.15	+0.01
	May 2012	+0.21	+0.10	−0.11
	Mean difference	+0.15	+0.09	−0.06
Pied Butcherbird	May 2011	−0.55	−0.50	+0.05
	Nov 2011	+0.31	+0.18	−0.13
	May 2012	+0.39	+0.01	−0.39
	Mean difference	+0.05	−0.11	−0.16
Grey Butcherbird	May 2011	−0.20	−0.10	+0.10
	Nov 2011	−0.50	−0.31	+0.19
	May 2012	−0.03	−0.10	−0.07
	Mean difference	−0.24	−0.17	+0.07
All birds (pooled)	May 2011	−1.51	−0.73	+0.77
	Nov 2011	+0.07	−1.34	−1.41
	May 2012	+2.80	+0.30	−2.51
	Mean difference	+0.46	−0.59	−1.05

[1]Falconidae consists of two species recorded, the Brown Falcon and the Black Falcon.

species, nor discount any historical or long-term changes in species composition or abundance from meat baiting. Nevertheless, Culgoa is well known for its avian diversity (>180 species), and there are no anecdotal evidence or data to suggest any species loss or decline that has coincided with baiting practices (Andy Coward, Culgoa Floodplain National Park pers. comm. 2014). Varanids are one group of nontarget species that may be susceptible, and are known bait consumers (Woodford *et al.* 2012), but were not surveyed during this study. Future monitoring of varanids through a baiting campaign is warranted to determine any population-level impacts. Nevertheless, we conclude that the lack of effect on nontarget bird populations supports the continued use of 1080 meat baits to control feral pigs in western Queensland and similar environs.

Management implications

While the results of this study indicate little risk to nontarget bird populations, it remains important to minimise nontarget exposure to meat baits to keep the risk low and to maximise uptake by pigs. There are options for reducing the nontarget uptake of meat baits, including distributing baits in the late afternoon/evening, dyeing baits green, covering or burying baits or using feeding deterrents, only placing toxic bait where feral pigs are feeding, or using pig-specific feeders (McIlroy 1983; McIlroy *et al.* 1993; Hone 2002; Elsworth *et al.* 2004; Bengsen *et al.* 2011; Mitchell 2011). However, many of these techniques would preclude the aerial application of

Table 4. Results of split–split plot ANOVA of taxa densities (birds km^{-2}). For each taxon, significance probabilities are given for each effect: site (treatment versus control), date (three baiting periods) and baiting (before and after survey). Residual mean squares are shown in parentheses

	Source of variation	d.f.	Corvids	Australian Magpie	Wedge-tailed Eagle	Falconidae	Australian Kestrel	Pied Butcherbird	Grey Butcherbird	All birds (pooled)
Site	Treatment	1	0.976	0.231	0.642	0.387	0.406	0.168	0.809	0.418
	Residual	2	(21.359)	(11.032)	(0.002)	(3.142)	(2.511)	(0.271)	(0.278)	(136.832)
Date	Date	2	0.419	0.098	0.394	0.470	0.362	0.069	0.682	0.179
	Date x Treatment	2	0.055	0.195	0.533	0.686	0.579	0.606	0.855	0.170
	Residual	4	(1.094)	(2.523)	(0.008)	(0.401)	(0.28267)	(0.099)	(0.103)	(6.733)
Pre–Post	Pre–post	1	0.963	0.840	0.917	0.132	0.870	0.379	0.626	0.484
	Date x Pre–post	2	0.210	0.970	0.217	0.216	0.930	0.895	0.713	0.572
	Treatment x Pre–post	1	0.557	0.878	0.063	0.096	0.612	0.841	0.383	0.677
	Date x Treatment x Pre–post	2	0.728	0.965	0.598	0.835	0.979	0.579	0.846	0.880
	Residual	6	(1.845)	(2.590)	(0.002)	(0.052)	(0.083)	(0.193)	(0.039)	(6.603)

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baits, or restrict current strategies, ultimately reducing the cost-effectiveness relative to other ground-based control techniques. Varanid uptake of meat baits may be easily reduced through distributing baits during the cooler months when ectotherms, like the lace monitor, are less active (Jessop *et al.* 2013). The benefits of such strategies need to be balanced against possibly compromising the proposed outcomes of the baiting campaign (e.g. to reduce environmental impacts).

Aerial baiting offers significant logistical and economic advantages to ground baiting for broadscale control, but such a strategy can increase availability to nontarget species, particularly when pigs have a restricted or localised distribution (Mitchell 2011). Ground baiting can be highly effective (e.g. Twigg *et al.* 2005) especially with prior free-feeding and targeting preferred foraging habitats (Mitchell 2008; Bengsen *et al.* 2014). While there are a variety of alternative bait types available for ground baiting (Mitchell 2011), options for aerial deployment are limited. The one alternative 1080 bait currently available for broadscale pig control (Pigout™, Animal Control Technologies Australia) offers shelf stability and ease of handling and is reportedly target specific in most parts of Australia (Cowled *et al.* 2006) except areas of the Queensland wet tropics (Bengsen *et al.* 2011). Alternative toxins and delivery systems are currently being developed for feral pigs (Cowled *et al.* 2008), but will still provide potential risks to nontarget consumers given the large doses required to kill pigs. While it remains important to continue to improve baiting practices to minimise the likelihood of nontarget deaths, the results from this study are reassuring, with negligible nontarget avian impacts in semi-arid environs.

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