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Turning ghosts into dragons: improving camera monitoring outcomes for a cryptic low-density Komodo dragon population in eastern Indonesia

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Abstract

Context. Detection probability is a key attribute influencing population-level wildlife estimates necessary for conservation inference. Increasingly, camera traps are used to monitor threatened reptile populations and communities. Komodo dragon (*Varanus komodoensis*) populations have been previously monitored using camera traps; however, considerations for improving detection probability estimates for very low-density populations have not been well investigated.

Aims. Here we compare the effects of baited versus non-baited camera monitoring protocols to influence Komodo dragon detection and occupancy estimates alongside monitoring survey design and cost considerations for ongoing population monitoring within the Wae Wuul Nature Reserve on Flores Island, Indonesia.

Methods. Twenty-six camera monitoring stations (CMS) were deployed throughout the study area with a minimum of 400 m among CMS to achieve independent sampling units. Each CMS was randomly assigned as a baited or non-baited camera monitoring station and deployed for 6 or 30 daily sampling events.

Key results. Baited camera monitoring produced higher site occupancy estimates with reduced variance. Komodo dragon detection probability estimates were 0.15 ± 0.092 – 0.22 (95% CI), 0.01 ± 0.001 – 0.03 , and 0.03 ± 0.01 – 0.04 for baited (6 daily survey sampling events), unbaited (6 daily survey sampling events) and long-unbaited (30 daily survey sampling events) sampling durations respectively. Additionally, the provision of baited lures at cameras had additional benefits for Komodo detection, survey design and sampling effort costs.

Conclusions. Our study indicated that baited cameras provide the most effective monitoring method to survey low-density Komodo dragon populations in protected areas on Flores.

Implications. We believe our monitoring approach now lends itself to evaluating population responses to ecological and anthropogenic factors, hence informing conservation efforts in this nature reserve.

Keywords. population monitoring, effective sampling, protected areas, apex predator, reptiles, *Varanus komodoensis*.

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Introduction

Large terrestrial predators are most often at risk from human actions and increasingly require effective conservation actions to ensure population persistence (Gittleman and Harvey 1982; Prowse *et al.* 2015; Penjor *et al.* 2019). The key requirement to establish effective conservation actions for apex predators is to accurately monitor population trends and status (Karanth *et al.* 2011). However, because apex predators are often rare or averse to capture or detect, non-invasive monitoring methods are routinely used to evaluate the effects of threatening processes or

conservation actions on their populations (Karanth *et al.* 2004; O'Connell *et al.* 2010). Similarly, the increasing use of hierarchical models such as site occupancy and n-mixture models, which account for imperfect detection, are now among the most common techniques used to provide population-level inference for apex predators (MacKenzie *et al.* 2002, 2006; Royle 2004; Kéry *et al.* 2005). Indeed, these methods are often well suited for threatened predator population studies (du Preez *et al.* 2014; Tan *et al.* 2017; Penjor *et al.* 2019; Searle *et al.* 2020), because threatened predators often persist at low densities where

individual-based recapture or resighting probabilities can be too low to allow for the alternate population estimates using mark–recapture type models (Williams *et al.* 2002; Kéry and Schmidt 2008; Couturier *et al.* 2013; du Preez *et al.* 2014; Tan *et al.* 2017; Searle *et al.* 2020).

Non-invasive monitoring techniques such as camera trapping are now increasingly used for monitoring terrestrial reptiles, a taxon with over 11 000 primarily predatory species (Ariefiandy *et al.* 2013; Jessop *et al.* 2013; Welbourne *et al.* 2015; Adams *et al.* 2017; Moore *et al.* 2020). Nevertheless, the use of cameras, as measured by the capacity to achieve adequate detection for robust population-level estimates, remains variable within and among reptile species because of the effects of body size, species habits and environmental factors (Ariefiandy *et al.* 2013; Welbourne *et al.* 2015; Richardson *et al.* 2017; Einoder *et al.* 2018). In the case of large reptiles, lower population densities, greater daily movement capacity, the effects of seasonal climatic variation, and smaller skin surface to ambient air temperature differences can all influence camera-based population monitoring effectiveness (Ariefiandy *et al.* 2013; Jessop *et al.* 2013; Welbourne 2013; Richardson *et al.* 2017; Hu *et al.* 2019). Furthermore, human activities can often disproportionately threaten large-bodied reptiles, causing their populations to be at much lower densities than normal and thus more difficult to monitor (Todd *et al.* 2010; Tingley *et al.* 2019). Thus, addressing these factors by modifying camera sampling designs to increase detection probability is a key consideration to monitor threatened reptile populations effectively. Under such circumstances, there may be compelling reasons to improve camera-based detection using baits or lures (i.e. attractants) to increase detection probability (O'Connell *et al.* 2010; Long *et al.* 2012; Read *et al.* 2015).

Multiple studies have reported that the use of baits or lures as attractants can vastly improve predators' detection sensitivity (du Preez *et al.* 2014; Austin *et al.* 2017; Comer *et al.* 2018). This result is especially important in predator populations where individuals can be cryptic or persist as low-density populations. Hence, attractants or baits may be essential to increase detection to prevent poor quality estimates of population-level parameters (Thompson 2013). For this reason, baited camera traps deployed during appropriate weather conditions can be advocated to optimise large-reptile detection probability (Jessop *et al.* 2013). Although, it is important to note that these benefits may need to consider how baits can affect a species' movement behaviour and create potential biases in any arising population-level estimates (Stewart *et al.* 2019).

The Komodo dragon (*Varanus komodoensis*) is the largest lizard and has an important ecological role as an apex predator (Jessop *et al.* 2006, 2019, 2020). The current distribution of Komodo dragons is restricted to five islands located in Komodo National Park and several fragmented populations on Flores Island (Jessop *et al.* 2007, 2018; Purwandana *et al.* 2014a; Ariefiandy *et al.* 2015; Jones *et al.* 2020). Populations on Flores Island have decreased because of anthropogenic activities and are now increasingly reliant on a small number of reserve areas to ensure their persistence (Ariefiandy *et al.* 2015, 2020; Jones *et al.* 2020).

Komodo dragon populations on Flores persist at much lower population densities (<1 dragon km^{-2}) than those observed in Komodo National Park (~ 10 dragons km^{-2} ; Laver *et al.* 2012;

Purwandana *et al.* 2014a; Ariefiandy *et al.* 2015, 2020). Multiple field methods have been used to estimate population trends of Komodo dragons (Ariefiandy *et al.* 2013, 2014). However, these can vary considerably in their monitoring effectiveness (Jessop *et al.* 2007; Ariefiandy *et al.* 2013, 2014; Purwandana *et al.* 2014a, 2015). On Flores, low densities and trap-wary behaviour of Komodo dragons favour wildlife cameras over direct trapping methods as a more effective population monitoring methodology (Ariefiandy *et al.* 2015). However, optimising camera monitoring design is still necessary to allow conservation managers to improve the data used to evaluate these most vulnerable populations (Jones *et al.* 2020). Here, we compare the effect of baited and unbaited camera sampling on the estimates of Komodo dragon detection probability and site occupancy, alongside other measures of monitoring efficacy and project running costs within the Wae Wuul Nature Reserve of Flores. Finally, we discuss the implications of our results for managing Komodo dragons within this protected area and, more broadly, for other populations distributed on the island of Flores.

Materials and methods

Study area

The Wae Wuul Nature Reserve comprises a protected area of 14.84 km^2 located on the western coast of Flores in eastern Indonesia (Fig. 1a, b). The reserve was established in 1985, aiming to increase protection of Komodo dragons beyond Komodo National Park. The climate is highly seasonal, dominated by a long dry season from March to November and a short wet season. Annual rainfall is less than 2000 mm (Monk *et al.* 1997). The study area comprises a hilly coastal landscape covered in multiple distinct vegetation communities. The two most common vegetation communities are savanna grassland (common species include *Eulalia leschenaultiana* and *Setaria adhaerens*) and savanna woodland (common species include *Borassus flabellifer* and *Zizyphus horsfeldii*) that cover $\sim 80\%$ of the study area (Auffenberg 1981). In valley floors holding permanent or ephemeral watercourses, drier vegetation communities are replaced by open deciduous monsoon forest ($\sim 20\%$ of the study area; dominant species include *Tamarindus indica*, *Schleichera oleosa* and *Cassia javanica*) or bamboo forest. These land cover types are representative of those found across the lowland coastal areas of major islands in this region of eastern Indonesia, including the adjacent Komodo National Park (Auffenberg 1981).

Study design

Twenty-six camera monitoring stations (CMS) were deployed within the Wae Wuul Nature Reserve. These CMS were placed within all key vegetation communities, including deciduous monsoon forest and savanna woodland. A minimum of 400 m separated all sites to improve data independence obtained from cameras (Ariefiandy *et al.* 2013, 2014). This 400-m distance between monitoring sites was based on the radius of the mean home-range area for Komodo dragons (Jessop *et al.* 2018; Purwandana *et al.* 2021). At the commencement of the study, each site was randomly assigned as a baited ($n = 13$) or non-baited ($n = 13$) camera monitoring station to ensure equal replicates within each camera method treatment. After the initial sampling period at each station, the assigned camera method

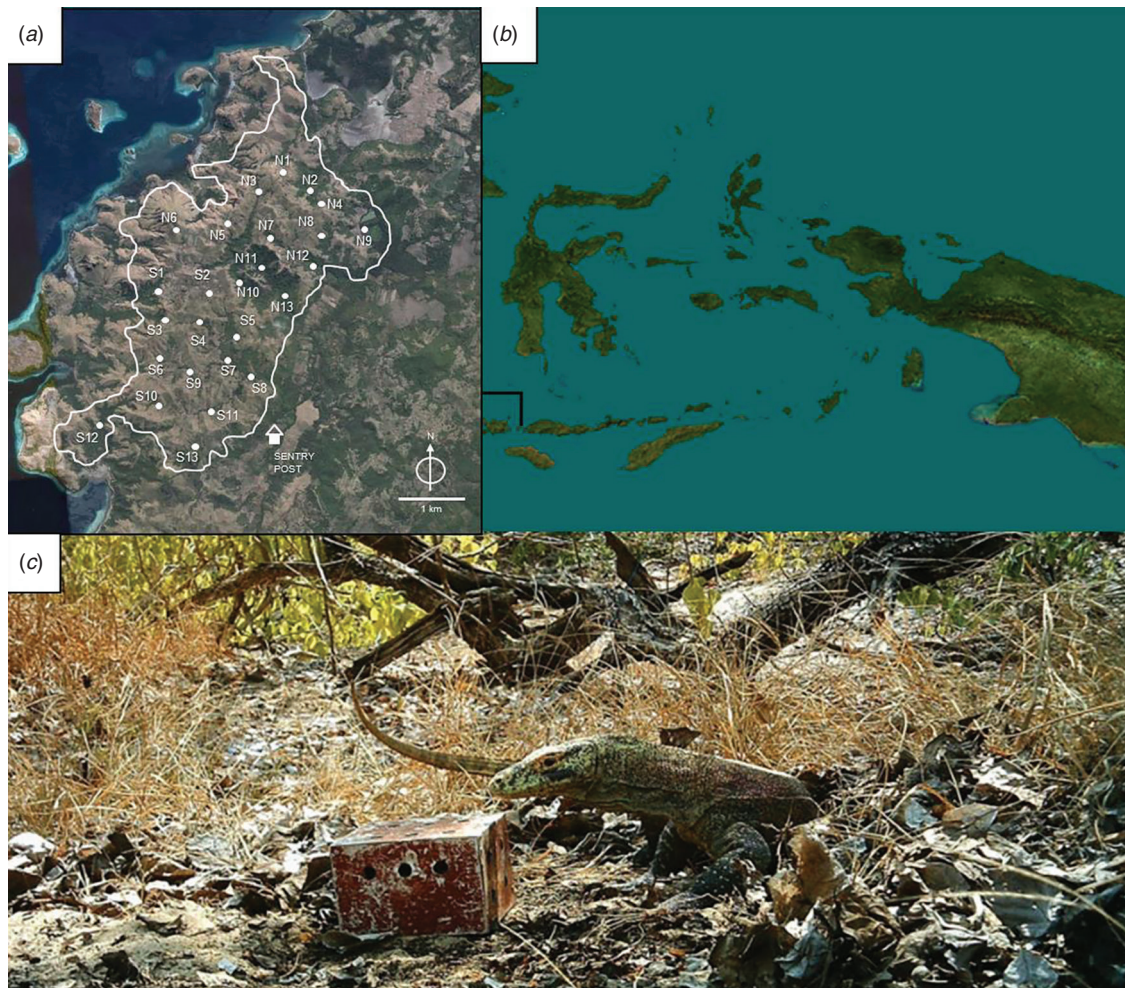


Fig. 1. The study evaluated the effect of bait attractant on Komodo dragon detection probability and site occupancy estimates by using camera monitoring stations deployed across (a) the Wae Wuul Nature Reserve located (b) on the western coast of Flores in eastern Indonesia. (c) An image of a Komodo dragon inspecting the meat attractant contained within a metal bait box.

was reversed to the alternate method to compare estimates of Komodo dragon detection probability and occupancy obtained for each method at each site. The study was conducted during June and July 2017 in the mid-dry season when environmental temperatures permit Komodo dragons to exhibit normal daily diurnal activity patterns and, hence, pending abundance, the potential for good detection probability (Harlow *et al.* 2010a, 2010b; Jessop *et al.* 2013).

Camera monitoring design

At each CMS, a single outward facing Bushnell camera (Model Trophy Cam HD 119678) was attached to a tree (40 cm above the ground) as described elsewhere (Ariefiandy *et al.* 2013, 2014). Cameras were programmed to take three photos and a 1-min video each time an animal triggered the device. At installation, all cameras were tested to confirm normal functioning. For CMS allocated to the bait treatment, we used two scent lures that comprised a small aluminium box (25 × 15 × 15 cm; L × W × H) and a suspended plastic bag, each containing goat meat that was placed 4 and 2 m in front or above the camera

respectively. Baited and unbaited CMS were deployed for 6 and 30 days of monitoring respectively. The uneven durations between treatments reflected our belief that baited cameras would require considerably less sampling effort to produce higher detection probabilities than those obtained from unbaited cameras. As Komodo dragons have been observed to investigate baits at traps for several minutes before entering traps or moving elsewhere, we also used a 30-min camera delay to prevent repeated photography of the same individual lizard (Ariefiandy *et al.* 2014). In addition, a 3-day non-monitoring period was used immediately after the transition from baited to unbaited sites. This waiting period was implemented to remove a potential carry-over bait effect that could have attracted Komodo dragons and inflated detection probability at sites then monitored with unbaited cameras. This research abided by the journal's guidelines on ethical standards.

Estimating detection and site occupancy estimates

We modelled the detectability and occupancy of Komodo dragons by using a single-season occupancy model, using the

Table 1. Model selection results testing effects of baited and non-baited cameras for influencing detection probability (p) and site occupancy (Ψ) of Komodo dragons within the Wae Wuul Nature Reserve in West Flores

K , the number of estimated parameters; logLik, loglikelihood; AIC, Akaike information criterion; Δ AIC, the difference in value between AIC of this model and the most parsimonious model; and AIC weights (w_i), a measure of relative model support

Model	K	logLik	AIC	Δ AIC	w_i
$\Psi(\cdot) p(\text{bait vs no-bait})$	3	365.43	371.43	0.00	0.71
$\Psi(\text{bait vs no-bait}) p(\text{bait vs no-bait})$	4	365.21	373.21	1.78	0.29
$\Psi(\cdot) p(\text{bait vs no-bait})^*$ daily survey variation	62	302.17	426.17	54.74	0.00
$\Psi(\cdot) p(\cdot)$	2	424.44	428.44	57.01	0.00

software Presence (Hines 2006). Site occupancy models use patterns of detection and non-detection over multiple surveys (sampling occasions) of a sampling unit (CMS) to estimate detection probabilities (p) and, thus, produce unbiased estimates of occupancy (ψ) (MacKenzie *et al.* 2002). We modelled the effect of baits on both the detection probability (p) and ψ relative to those cameras without baits (i.e. p, ψ). We partitioned the unbaited CMS detection probability data into two datasets, given the sampling duration differences between baited and unbaited CMS. One dataset comprised the first six, and the other the full 30 daily sampling events. Models were ranked using AIC, and we considered the effect of bait provision at CMS to be influential if the model AIC was >2 units below that estimated for the null model (Burnham and Anderson 2004).

Detection probability curves, probability of site absences and survey design costs

To assess the expected reduction in sampling effort provided by using baits at CMS, we produced detectability curves for CMS with and without baits. Detectability curves represent the cumulative probability (i.e. rate of increase) that Komodo dragons will be detected after a given number of sampling occasions in a site where the species is present (Wintle *et al.* 2005). Cumulative detection probability curves were estimated as $pk = 1 - (1 - p)^k$, where p is the species' per-survey detection probability within a given treatment and k is the given number of sampling occasions (MacKenzie and Royle 2005).

Next, we estimated the minimum number of sequential sampling occasions, with no detection required to be 95% certain (i.e. $\alpha = 0.05$) that Komodo dragons were absent from a surveyed site by using baited and unbaited cameras (Wintle *et al.* 2012; Ferreras *et al.* 2018). The probability (with $\alpha = 0.05$) of not detecting Komodo dragons after N sampling occasions at a given site is estimated by the formula

$$N > \frac{\log\left(\frac{\alpha}{1-\alpha}\right) - \log\left(\frac{\psi}{1-\psi}\right)}{\log(1-p)}$$

Here, values of p and ψ are specific to baited and unbaited CMS site occupancy estimates derived from the 6-day sampling period.

Finally, we compared the costs of sampling for baited and unbaited camera trapping methodology to achieve a similar monitoring outcome (i.e. $\alpha = 0.05$) by calculating protocol-specific costs of each technique, beyond common costs associated with camera purchases, as such we estimated

$$C(m) = \sum (C_d + C_r + C_b \times S_d + C_{bb} \times S_d + C_{cb} \times S_d)$$

where $C(m)$ = method specific survey cost, C_d = cost of camera deployment, C_r = cost of camera retrieval, C_b = cost of bait (US\$0.26 per camera per survey day (S_d)), C_{bb} = cost of bait boxes (US\$10.00 per camera), C_{cb} = cost of camera batteries (US\$0.20 per camera per survey day).

Results

The most parsimonious occupancy model ($\Psi(\cdot), p(\text{bait vs no-bait})$, model weight = 0.71) indicated that the effect of baits placed at CMS vastly improved Komodo dragon detection probability compared with the null model (Δ AIC = 57.01; model weight = 0.00; Table 1). Detection probability estimates for baited cameras (6 daily sampling events) were 15 and 5.5 times higher than those estimated for unbaited (6 daily sampling events) and long-unbaited (30 daily sampling events) camera sampling durations (Fig. 2a). Similarly, baited cameras produced 2.3 and 1.3 higher Komodo dragon site occupancy estimates at the equivalent and long-unbaited camera sampling durations (Fig. 2b). A goodness-of-fit test on the most parameter-rich model demonstrated that our data were not over-dispersed (i.e. $\hat{c} > 1$).

Effects of bait attractants on monitoring considerations

Baited cameras improved sampling efficacy and reduced monitoring costs compared with sampling using unbaited cameras. First, it was evident that based on cumulative detection probabilities, baited cameras, if deployed sufficiently long enough, could achieve perfect detection at sites with Komodo dragons, with much less survey effort than with unbaited cameras (Fig. 3). Compared with unbaited cameras, baited cameras reduced the sampling effort duration from 184 to 21 days to be certain (with α of <0.05) that Komodo dragons were absent from a site. Finally, because of the much-improved detection probability achieved with baited cameras, it reduced the overall study costs from US\$580.20 to US\$547.60, to obtain similar camera-based detection levels within the study area.

Discussion

The choice of an appropriate sampling method for monitoring threatened predator populations depends on interactions among the program objectives, scale and resources and a species's detection probability (Kéry and Schmidt 2008). We demonstrated that using baited-camera compared with unbaited-camera monitoring greatly improved estimates of Komodo dragon detection probability and site occupancy in the Wae

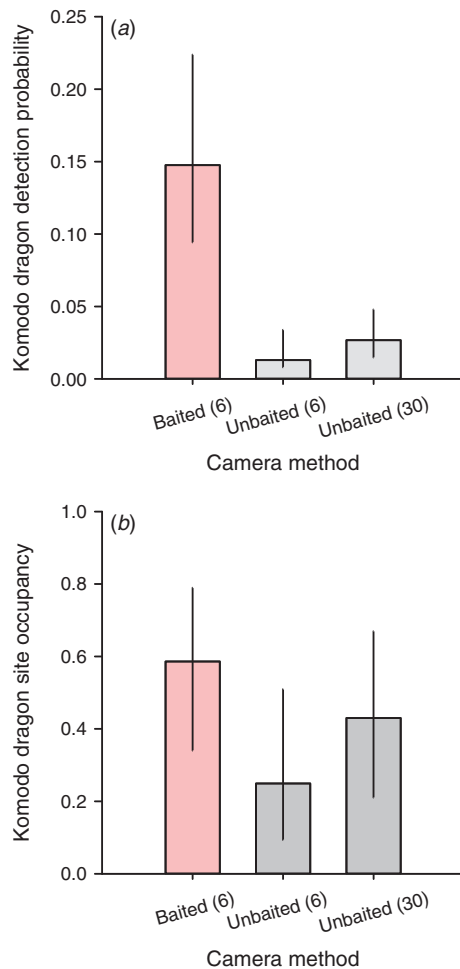


Fig. 2. Komodo dragon (a) detection probability and (b) site occupancy estimated from baited or unbaited cameras deployed for 6- or 30-day sampling events respectively. The bars report the mean estimate, and the upper and lower 95% confidence intervals are indicated by the error bars.

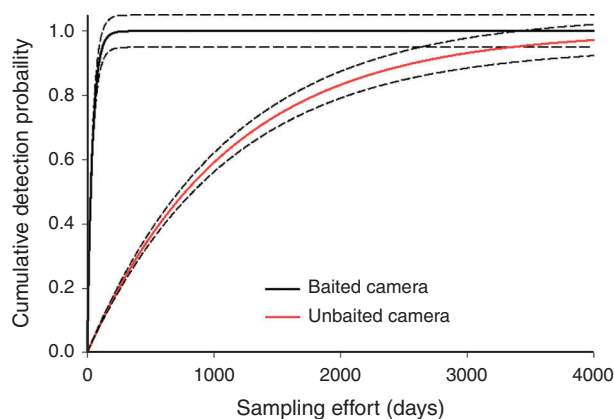


Fig. 3. The cumulative detectability curves for Komodo dragons estimated from baited and unbaited camera occupancy models. These curves represent the probability that Komodo dragons will be detected at least once with each treatment after sequential 1-week sampling period at each camera trap where Komodo dragons are present. The lines report the mean estimate and the dashed upper and lower lines are the 95% standard error of the mean.

Wuul Nature Reserve on Flores Island. Indeed, several clear advantages were evident from using baited cameras, including a reduced sampling effort and, ultimately, a more cost-effective monitoring design.

Obtaining a high detection probability is a key requirement to improve site-occupancy estimates for large predators that persist at low density (MacKenzie *et al.* 2006). Such sampling designs should aim to achieve a detection probability exceeding 0.15, so as to allow for better occupancy estimates for predators (O'Connell *et al.* 2010; Otto and Roloff 2011). With camera-based monitoring, there are several ways to increase species detection probability, including increasing the number of cameras deployed for longer survey periods or by also placing cameras in areas that increase detection opportunities of the focal species (e.g. along game trails; O'Connell *et al.* 2010; Geyle *et al.* 2020; Wysong *et al.* 2020). However, the use of attractants such as baits or lures is another common means to improve camera-based detection probability, but their use should be assessed to ensure improved efficacy (Read *et al.* 2015).

It was evident that bait attractants at camera monitoring stations greatly improved Komodo dragon detection probability by 3.5–5 times over similar or extended durations of unbaited camera monitoring. This finding is consistent with those of other studies that indicate similar benefits of using baits or lures at camera monitoring stations (du Preez *et al.* 2014; Austin *et al.* 2017; Tarugara *et al.* 2019). Importantly, these gains in detection probability alongside higher and more robust estimates of site occupancy offset the increased daily sampling costs owing to the purchase of goats as the bait source (Thorn *et al.* 2011).

Another key benefit of baited cameras was the considerable reduction (i.e. 5-fold) in the survey effort needed to achieve adequate Komodo dragon detection within the study area. Reducing survey effort without compromising detection probabilities has many obvious advantages (MacKenzie and Royle 2005). Most importantly, saved survey effort can be allocated into additional sites, survey visits or additional study areas in different ways (Sewell *et al.* 2012). From our perspective, the biggest advantage is that reduced survey effort can be invested into additional camera monitoring activities for more broadly assessing the conservation status of Komodo dragon populations. For example, we have recently used baited camera monitoring surveys beyond this study area to evaluate the distribution of the Komodo dragon across Flores (~400 monitoring stations across 1200 km of coastline; Ariefiandy *et al.* 2021). This feat would not have been possible without using baited cameras to achieve high Komodo dragon detection relative to their survey effort requirements.

It is argued that the use of attractants to increase a species detection must be considerate of any effects on monitoring estimates and arising inference (du Preez *et al.* 2014). For example, if increased estimates of detection at baited cameras arise because of bait effects on animal space-use or daily movements, it could bias parameter estimates. In the case of baited cameras, baits could increase residency times or attract animals beyond their normal home-range area to inflate estimates of detection probability and site occupancy (Stewart *et al.* 2019). This problem could be especially acute if individual animals, particularly those in low-density populations, are detected at multiple camera stations beyond their home range.

Consequently, ensuring spatial independence for camera data is a crucial aspect of monitoring design (Meek *et al.* 2014; O'Connell *et al.* 2010; Geyle *et al.* 2020). We know that the independence of data among camera monitoring sites is largely met for Komodo dragon, because our prior mark–recapture-based studies using traps with similar inter-site distances resulted in a <10% within-study recapture rate of individuals (Ariefiandy *et al.* 2013, 2014).

This study also indicated that estimates of Komodo dragon site occupancy recorded within the Wae Wuul Nature Reserve are significantly lower than those generally recorded for populations in the adjacent Komodo National Park (Purwandana *et al.* 2014b; Ariefiandy *et al.* 2015). Adult Komodo dragons, as apex predators, mainly prey on ungulates, particularly Rusa deer (*Rusa timorensis*), wild pig (*Sus scrofa*), and, in some locations, water buffalo (*Bubalus bubalis*; Auffenberg 1981; Bull *et al.* 2010; Purwandana *et al.* 2016). Thus, we attribute this lower occupancy estimate to be in part associated with the commensurate reduction of large ungulate prey availability on Flores (Ariefiandy *et al.* 2011, 2015, 2016; Jessop *et al.* 2020). Reduced prey is a presumed consequence of historical and increasingly contemporary human-mediated processes (e.g. fire, poaching, invasive predators) affecting Komodo dragon habitats on Flores (Ariefiandy *et al.* 2020).

Here we advocate that protected-area enhancement actions and community conservation approaches are needed to address the current threats to Komodo dragons on Flores. For example, unlike Komodo National Park, the Wae Wuul Nature reserve is comparatively under-resourced in staff and logistical resources. Thus, aside from ongoing monitoring of Komodo dragon populations, it is necessary to ensure that integrative conservation actions are used to ensure prey and predator persistence in this reserve (Ariefiandy *et al.* 2015, 2020). Thus, this reserve could benefit from additional infrastructure (e.g. ranger posts) and increased patrolling and surveillance measures that would benefit both Komodo dragons and their ungulate prey (Hilborn *et al.* 2006; Ariefiandy *et al.* 2015, 2020). However, as human activities increasingly modify the habitats that directly border this reserve, community-based conservation actions must also be implemented in neighbouring communities (Ariefiandy *et al.* 2015, 2020). For example, implementing conservation awareness meetings in local communities to inform and discuss the value of protecting natural values within this reserve are deemed essential (Kamil *et al.* 2019). Furthermore, working with communities to reduce rates of incursions of village dogs or livestock and stopping villagers from setting fire to habitats within or adjacent to the reserve could be important steps to promote the conservation of Komodo dragons in this key protected area on Flores (Ariefiandy *et al.* 2020).

In conclusion, our study demonstrated that optimising camera survey methods using baits compared with unbaited cameras can provide a better method for estimating Komodo dragon occupancy. This result was particularly important in this study because we aimed to effectively monitor a very low-density population in a key protected area on Flores. We believe our baited camera monitoring approach now lends itself to understanding population responses to ecological and anthropogenic factors, hence informing conservation efforts in this nature reserve (Ariefiandy *et al.* 2015).

Conflicts of interest

The authors declare that they have no conflicts of interest.

Author contributions

Study design and fieldwork: DP, AA, MA, SAN, MS and TSJ; data analysis: TSJ; writing: TSJ.

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