Evaluation of three field monitoring-density estimation protocols and their relevance to Komodo dragon conservation.

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Abstract

Finding practical ways to robustly estimate abundance or density trends in threatened species is a key facet for effective conservation management. Further identifying less expensive monitoring methods that provide adequate data for robust population density estimates can facilitate increased investment into other conservation initiatives needed for species recovery. Here we evaluated and compared inference- and cost- effectiveness criteria for three field monitoring -density estimation protocols to improve conservation activities for the threatened Komodo dragon (*Varanus komodoensis*). We undertook line-transect counts, cage trapping and camera monitoring surveys for Komodo dragons at 11 sites within protected areas in Eastern Indonesia to collect data to estimate density using distance sampling methods or the Royle-Nichols abundance induced heterogeneity model. Distance sampling estimates were considered poor due to large confidence intervals, a high coefficient of variation and that false absences were obtained in 45% of sites where other monitoring methods detected lizards present. The Royle-Nichols model using presence/absence data obtained from cage trapping and camera monitoring produced highly correlated density estimates, obtained similar measures of precision and recorded no false absences in data collation. However because costs associated with camera monitoring were considerably less than cage trapping methods, albeit marginally more expensive than distance sampling, better inference from this method is advocated for ongoing population monitoring of Komodo dragons. Further the cost-savings achieved by adopting this field monitoring method could facilitate increased expenditure on alternative management strategies that could help address current declines in two Komodo dragon populations.

Key Words: density estimation, distance sampling, abundance occupancy models, apex predators, large reptiles, Komodo dragon.

INTRODUCTION

Robustly estimated trends in population abundance or density are key requirements to determine the conservation requirements of species (IUCN 2001; Yoccoz et al. 2001; Williams et al. 2002). This demographic information can determine the level of conservation prioritization (e.g. IUCN threat status rankings), or signal the magnitude and nature of conservation resources needed to initiate recovery efforts and ultimately be used to gauge the effectiveness of conservation actions on species recovery (Pollock et al. 2002). Despite population abundance being one of the most useful indicators to influence conservation management decisions it remains difficult to robustly estimate (i.e. precise and unbiased) (Yoccoz et al. 2001; Ke´ry et al. 2005). This problem arises because it is necessary to reconcile the central problem of imperfect detection, where a proportion of animals in the surveyed area go undetected, and hence leads to reduced abundance estimates (Williams et al. 2002). Intensive mark recapture sampling of individuals, each identifiable through unique tags, or applied, or natural, markings provides one major way to account for imperfect detection (MacKenzie et al. 2002; Williams et al. 2002).

However mark recapture surveys, especially in studies where direct capture of large animals is necessary, is inevitably expensive, time consuming and often leads to restricted areas of sampling at the expense of surveying large areas of a species distribution (Williams et al. 2002, Karanth et al. 2011). Perhaps unsurprisingly there has been considerable interest in coupling less intensive survey methods with the development of alternative model estimators that account for imperfect detection and still provide robust abundance or density estimates. Two examples include distance sampling and abundance type occupancy models (Buckland et al. 2001, Royle and Nichols 2003, Royle 2004, Thomas et al. 2010). Distance sampling is a widely used method that can estimate abundance/density using distance bounded count data drawn from linear transects or plots (Buckland et al. 2001, Thomas et al. 2010). However, there are also occupancy type models that explicitly estimate abundance (Λ) including the Royle-Nichols abundance induced heterogeneity model (Royle and Nichols 2003; Royle 2004). These abundance type occupancy models can use either count or presence-absence data obtained from various field detection methods and unlike mark recapture data do not require a unique capture history for each animal detected.

A second consideration to determine appropriate monitoring methods used to estimate population abundance or density estimates is to consider that money spent on monitoring can compete with other conservation activities that could better serve to ensure species persistence (McDonald-Madden et al. 2010; Possingham et al. 2012). Hence, trade-offs in investment between monitoring and conservation are often expected but remain rarely estimated. For example, if a chief management goal is to quantify abundance or density of a threatened species, and given multiple density estimators are available, then it would be logical to invest in the most-cost effective field sampling method that provides data sufficient to meet model assumption or convergence criteria need to estimate abundance or density. In doing so, saving on monitoring costs could be invested into alternative management actions that could improve conservation efforts (Possingham et al. 2012)

We aimed to evaluate and compare inference- and cost- effectiveness criteria of three different monitoring method-density estimation protocols that could potentially be used to conduct long term population monitoring of the threatened Komodo dragon (*Varanus komodoensis*). Currently, Komodo dragons inhabit five small islands in eastern Indonesia, with four island populations located within Komodo National Park (KNP) and several fragmented populations persisting on the larger island of Flores (Ciofi and De Boer 2004).

However, both Komodo dragon range size and some island populations have decreased significantly in recent decades raising conservation concerns for this species (Ciofi and De Boer 2004; Purwandana et al. 2014). Anthropogenic threats, including the poaching of Timor deer and habitat loss are suspected to be major causes of range reduction and population decline in Komodo dragons (Jessop et al. 2004, Jessop et al. 2006, Jessop et al. 2007; Purwandana et al. 2014). Whilst long-term population monitoring of Komodo dragons is advocated to enable management authorities to identify populations at risk, also finding the most cost-effective monitoring method could enable redirected investment into recovery options that could better managing recently identified declining populations (Jessop et al. 2007; Purwandana et al. 2014).

We first report results obtained from undertaking three field monitoring method- density estimation protocols that used (1) line transects to compile Komodo dragon sightings; and (2) cage traps and (3) passive infrared triggered wildlife cameras to collate presence/absence data for Komodo dragons at 11 sites in protected areas across Eastern Indonesia. We then analysed these data using distance methods or the Royle-Nichols abundance induced heterogeneity model to estimate Komodo dragon population density at each site. Next we report the cost-benefit ratio of each monitoring method-density estimation protocol by assessing criteria that considered the robustness of density estimates (i.e. benefits) relative to their financial expenditure (i.e. costs) (Parnell et al. 2013). In light of this information we then advocated which field monitoring method is considered best for the long-term population monitoring of Komodo dragons. Finally we considered how changes to current monitoring expenditure (i.e. use of cage trapping and mark recapture methods; Purwandana et al. 2014) could influence alternative conservation actions for this species.

MATERIALS AND METHODS

Study area

Our study was conducted at 11 sites on the five islands on which Komodo dragons still persist in Eastern Indonesia. Sites were situated across four islands in Komodo National Park (8°35’22”S, 119°36’52”E) and in the Wae Wuul Nature Reserve (8°35’50”S, 119°50’05”E) located on the west coast of Flores Island (Fig. 1). For the ten study sites within Komodo National Park: four sites were located on Komodo island, 1) Loh Liang (K1), 2) Loh Lawi (K2), 3) Loh Sebita (K3), 4) Loh Wau (K4); another four on Rinca island , 5) Loh Buaya (R1), 6) Loh Baru (R2), 7) Loh Tongker (R3), 8) Loh Dasami (R4); and a single site was located on each of the two small islands 9) Gili Motang (GM) and 10) Nusa Kode (NK). Site 11 was located in Wae Wuul Nature Reserve on the west coast of Flores. Hunting of ungulates is prohibited in these nature reserves and park rangers regularly patrol these 11 sites.

All islands share similar habitat characteristics and experience a tropical monsoonal climate (Monk et al., 1997). There are four main habitat types across (Auffenberg 1981, Monk et al. 1997). Tropical monsoon forest dominates above 500 – 700 m a.s.l. and deciduous monsoon forest (primarily tamarind *Tamarindus indica*) occurs in valley floors and along water courses. Savannah woodland and savannah grassland dominate drier areas. Komodo dragons preferentially use deciduous monsoon forest, as a consequence of their thermoregulatory requirements and the location of their prey (Harlow et al. 2010; Purwandana et al., 2014). Hence our field sites were situated in the preferred habitats of Komodo dragons.

Study Species

Komodo dragons are large (up to 80 kg) and long-lived monitor lizards (up to 60 years) that actively forage and kill prey (Auffenberg 1981; Jessop et al. 2006; Laver et al. 2012). Lizards are active year round consistent with their life in a tropical warm climate (Auffenberg 1981). Daily activities comprise predominantly diurnal foraging activities, where individuals based on telemetry studies are active across the day, and pending their size, can move up to several kilometres to seek prey (Auffenberg 1981; Imansyah et al., 2008). Individual lizards are largely solitary in habits, and multiple individual are only observed in close proximity when feeding on large prey (deer and buffalo; Bull et al., 2010) or during seasonal mating activities (Auffenberg 1981).

These facets of their biology including large body size, active forgaing habits and favourable year round climatic factors (that influence daily activity) should promote adequate detection to satisfy data requirements needed to facilitate density estimates from distance sampling and Royle-Nichols methods. Thus we considered that neither species specific (e.g. behavioural avoidance), nor environmental parameters (e.g. daily temperature) would unduly influence lizard activity and require modification of sampling protocols to address detection concerns (Courtier et al. 2013; Jessop et al. 2013).

Field Monitoring-Density Estimation Protocols

Distance sampling and density estimation

We conducted distance sampling surveys concurrently with cage trapping methods in 2010. Distance surveys were conducted in the early morning (06.30 – 09.30) and late afternoon (15.00 – 17.30) when lizards were most active, to increase the likelihood of sighting individuals (Imansyah et al. 2008). Transects were located systematically along grid lines in each of the 11 sites, with the distance between transects ≥500 m. We used hand held GPS (Garmin Summit, Kansas, USA) to locate that start point and hold a compass bearing between to the end point of each transect. As transect grids covered the extent of each study site they enabled sampling across multiple vegetation types where lizards occur.

We surveyed lizards along 111 transects of variable length (0.5 – 6.15 km) totalling 163.65 km of surveyed habitat. The same observers (AA and DP) conducted all surveys. Observers first walked 13.5 km of transect together to standardize methodology. Thereafter, the two observers surveyed alternate transects at each site. Surveys were conducted at a slow walking speed of 2 – 3 km h^{-1}. We only recorded lizard being detected if they were directly sighted, we did not consider detections from indirect signs (e.g. hearing movements through vegetation but the animal remained unsighted) as these would bias data and were not encountered during this study. As most direct sightings of lizards involved individuals moving at normal walking speed (~3 km/hr), we recorded the radial distance from the observer to the animal location at first sighting using a laser range finder (Bushnell Range Finder Elite 1500, Bushnell Corporation, Overland Park, Kansas, USA) and the bearing to the animal determined with an electronic compass (Garmin Summit, Kansas, USA). Perpendicular distances were calculated from the radial distances and sighting angles by trigonometry (Buckland et al. 2001).

We analysed data using the program DISTANCE 6.0 release 2 (Thomas et al. 2009; http://www.ruwpa.st-and.ac.uk/distance/) to estimate site-specific density estimates. Distance sampling relies on three assumptions to reduce bias in density estimates (Thomas et al. 2009). Assumptions include that animals sighted directly on the transect line are always detected, (i.e., g(0) = 1), that animals do not move deliberately to avoid or seek detection by observers and that distances to animal are measured accurately (Buckland et al. 2001). We had no prior belief that such assumptions would not be met during our distance sampling of Komodo dragons.

Buckland et al. (2001) recommended having at least 60 observations for robust estimation of density using distance sampling. Since there were far fewer than 60 observations of Komodo dragons at all sites, we used multiple-covariate distance sampling (MCDS) to estimate site specific density (Marques et al. 2007; Thomas et al. 2010). Here sites are treated as a factor covariate in a multiple-covariate detection function; this enables estimation of a global detection function that is then applied to estimate each site-specific observations to produce respective density estimates (Marques et al. 2007; Thomas et al. 2010). Mean site-specific cluster size (i.e. lizard group size) was estimated using the size-biased regression method (Buckland et al. 2001).

Our exploratory data analyses revealed that detection data for komodo dragons had long tails, so the 5% of detections with the greatest distances were discarded (Buckland et al. 2001; Thomas et al. 2010). Following Thomas et al. (2010), we evaluated the following detection functions, g(y), for lizards, where y is the perpendicular distance (m) of an observation from the transect. We compared the he half-normal and hazard-rate key functions and evaluated these with cosine and polynomial expansions. Further details on these keys and adjustments are given in Buckland et al. (2001). We used Akaike's Information Criterion corrected for small sample sizes (AIC_c) to assess the relative support for each model. Histograms, quantile-quantile plots and Cramér-von Mises tests were used to assess if data met the assumption of the distance sampling model. Following Buckland et al. (2001) and Thomas et al. (2010), site-specific estimates of lizard density are presented with 95% CI and the coefficient of variation (CV).

Presence-absence data collection field methods

a) Cage trapping

In 2010, we used a total of 230 trapping locations (i.e. a fixed point of trap placement) as sampling units that were distributed across 11 study sites on five islands. Within each study site, baited cage traps were placed at individual trapping locations (Lawi, n=32; Liang, n=32; Sebita = 21, Wau = 9, Baru = 22, Buaya, n=22; Tongker, n=13; Dasami = 24; Motang, n=16; Kode, n=12; Wae Wuul = 26) to capture Komodo dragons. Differences in trap number per site reflected site-specific variation in area and habitat type (traps are not placed in open vegetation such savannah woodland or savannah grassland). Traps comprised purpose built aluminum cage traps (300 cm L x 50 cm H x 50 cm W) fitted with a wire activated front door. The distance between trap locations was set at approximately 500 m in order to maintain independence among traps. Traps were positioned in forested areas to avoid any potential overheating of trapped individuals and that lizards too are much more common in these habitats relative to more open and hotter habitats (e.g. savannah grassland). Goat meat (≈ 0.5 kg) was used as bait to lure lizards into traps. Additionally a bag of goat meat was suspended 3-4 metres above each trap to act as a scent lure to further attract Komodo dragons to each trapping location. Traps are effective in capturing all lizards, except for hatchlings and small juvenile lizards (< 1 kg) that exhibit an arboreal life stage that precludes their capture using this method (Imansyah et al. 2008).

At each trapping location, monitoring activities occurred over three consecutive days, with each trap checked twice daily (8-11am and 2-5pm) for the capture of Komodo dragons resulting in six sampling events. The time interval between the morning and afternoon daily check for each trap was ~ 6 hrs. Cumulatively this sampling design provided 1374 trapping opportunities for Komodo dragon to be captured.

b) Camera monitoring

In 2013, we used a total of 230 camera detection stations (i.e. a fixed point of camera placement) as sampling units that were distributed across 11 study sites on five islands. Within each study site, baited camera detection stations were overlaid onto the cage trapping locations used in 2010 (Lawi, n=34; Liang, n=30; Sebita = 22, Wau = 8, Baru = 23, Buaya, n=22; Tongker, n=14; Dasami = 24; Motang, n=16; Kode, n=12; Wae Wuul = 26). These detection stations were overlaid onto the locations at which cage traps were used in 2010 for consistency of sampling.

Following methods outlined in Ariefiandy et al. (2013), Scout Guard cameras (model SG-560V) were attached to a tree (40cm above the ground). Cameras were programmed to take three photos each time the animal triggered the device. A 15 minute delay was included to prevent repeated photography of the same individual lizard. Goat meat (≈ 0.5 kg) was placed in aluminum boxes (25 cm L x 15 cm H x 15 cm W) and positioned three-four meters in front of each camera to lure into the field of view of each camera. In addition, similar to cage trapping additional bait was (≈ 5 kg) placed into plastic bag and suspended 2-3 meters above the bait box to further attract dragons to camera detection stations.

At each detection station, cameras were run continuously for three days. To be consistent with cage trapping we divided camera sampling into morning (8-12pm) and afternoon events (2-5pm). All Komodo dragon images captured within the six sampling events were used to denote the presence of lizards at a camera detection station. Cumulatively this sampling design provided 1374 detection opportunities for Komodo dragon to be photographed.

Density estimates using the Royle-Nichols abundance induced heterogeneity model

To estimate site specific Komodo dragon population density from cage trapping and camera station presence-absence data respectively we used the Royle-Nichols abundance induced heterogeneity model (henceforth the Royle-Nichols model) in PRESENCE 6.2 (Hines 2006). The Royle-Nichols model provides estimates of the parameters λ and r, representing average abundance per site and species detectability respectively (Royle and Nichols 2003). The parameter λ can be interpreted as an index of abundance. However, this assumes that detection of individuals is independent and site-specific abundance of individuals follows a Poisson distribution (which is the mixture distribution used in PRESENCE models), λ may also be interpreted as the expected number of individuals per sample unit (Royle and Nichols, 2003; MacKenzie et al., 2006). We thus divided λ by the sampling site area to estimate average Komodo dragon density across each site.

To ensure site specific estimates of λ were the most parsimonious model from our data we compared six models where we modelled combinations of λ as being either site variant or site invariant (λ_{site} and $\lambda_{.}$) and r as a function of as being either site variant, survey variant or site invariant ($r_{site,}$ r_{survey} and r.) (Table 3). We used AIC_c to assess the relative support for each model (Burnham and Anderson 2002).

Comparison and Cost-Benefit Analysis of Monitoring-Estimation Protocols

We conducted a Pearson product-moment correlation test to measure the strength of the linear relationship between pair wise combinations of the three density measures to assess their concordance. Next we considered multiple criteria to determine a simple benefit-cost ratio (BCR) analysis for assessing which of the three monitoring-estimation protocols provided the most-cost effective means to estimate trends in Komodo dragon density relative to the annual costs of monitoring costs (Pannell et al. 2013). First, to estimate the benefits of each protocol we tallied the number of sites for which each sampling method could provide a density estimate based on detecting Komodo dragon present within each site. Second we calculated the respective co-efficient of variation (COV) for the density estimates obtained from each method. To determine the net protocol benefits we obtained the quotient of these two benefit measures. To estimate costs we tallied all sources of expenditure that would be required to undertake annual monitoring in the context that each method would be used for long-term monitoring (Appendix 1). For monitoring costs we considered all equipment, logistical, administrative and labor costs. The benefit to cost ratio is then simply calculated as the quotient of benefits to costs and these ratios were then used to rank the three monitoring methods accordingly.

RESULTS

Distance sampling

The time required to conduct distance sampling along the 163.65 km of transects was 74 hours. A total of 34 dragons in 31 clusters (i.e. group size) were observed at the 11 sites, with a minimum of 0 and a maximum of 17 clusters observed at sites (Appendix 2). This equated to a low encounter rate of one dragon for every 4.8 km of transect surveyed. Mean (± SE) cluster size also varied among sites, ranging from 0 at six sites (R2,R4, GM,NK,WW) to 1.18 ± 0.10 at Loh Liang (K1) (Appendix 1).

The best detection function model for Komodo dragons was the hazard rate key with no adjustments (AIC_c = 3665.41; w_i = 0.89). The fitted detection function had a reasonable shoulder (Appendix 3a) and the q-q plot showed no substantial departures from expectation (Appendix 3b). The Cramér-von Mises tests were also non-significant ($P > 0.2$). Lizard densities estimated using distance sampling varied ranged from 0 lizards/km^2 at 5 sites (GM, NK, WW, R4, R2) to 17.3 lizards/km^2 at Loh Liang (K1) (Table 1).

Royle -Nichols Model

a) Cage –trapping based monitoring

Across the 230 trapping locations at 11 sites on 5 islands we captured 472 Komodo dragons from a total of 1386 sampling occasions. Ranking of six Royle-Nichols Abundance Induced Heterogeneity models used to estimate λ (average abundance per site) and r (innate species detectability) indicated overwhelming model support (w = 0.96) for the model $\lambda_{site}\, r_{survey}$ relative to the five other models considered (Appendix 4a). This top ranked model indicated that abundance was most influence by site and species detectability was most influenced by day of survey. The density estimates for sites within the two large islands (Komodo and

Rinca) ranged from 9.18 ± 0.85 to 38.47 ± 3.28 dragons/km^2 in Komodo National Park, and were relatively high compare to the two small islands of Gili Motang and Nusa Kode, 3.59 ± 1.21 and 6.20 ± 2.16 dragons/km^2 respectively. The Wae Wuul Nature Reserve on Flores had by far the lowest density estimate of 0.99 ± 0.22 dragons/km^2 (Table 2). The detection parameter r (range: 0.10 ± 0.02 - 0.20 ± 0.04) varied with day and exhibited a concave down pattern in daily survey detection (Appendix 5a).

b) Camera-trapping based monitoring

Baited camera stations were placed at 230 locations at 11 sites on 5 islands and resulted in 348 Komodo dragon photo detections from 1386 sampling events. Ranking of six Royle-Nichols Abundance Induced Heterogeneity models used to estimate λ (average abundance per site) and r (innate species detectability) again indicated overwhelming model support (w = 0.96) for the model $\lambda_{site}\, r_{survey}$ relative to the five other models considered (Appendix 4b). The site specific density estimates ranged from 0.54 ± 0.25 - 25.16 ± 8.48 Komodo dragons/km^2 (Table 2). The detection parameter r (range: 0.12 ± 0.02 - 0.22 ± 0.04) varied with survey and suggested higher detection obtained from morning camera monitoring compared to the afternoon (Appendix 5b).

Comparison of Methods

The relationships between density estimates derived from the Royle-Nichols model using cage trapping and camera monitoring data were highly significantly correlated (Pearson correlation: r = 0.82; P= 0.002). However density estimates from distance sampling were poorly correlated with the Royle-Nichols model estimates obtained using cage trapping (Pearson correlation: r = 0.20; P= 0.55) and camera monitoring data (Pearson correlation: r = 0.20; P= 0.55). Our distance sampling methods resulted in density estimates for 6 of the 11 sites. This meant that at 5 sites our transect sampling failed to detect Komodo dragons. In all cases we considered these monitoring results incidences of false absences, given that both cage trapping and camera monitoring detected Komodo dragons at these sites (Table 1). The coefficient of variation (COV) for site density estimates derived from distance sampling (148.62%) was much greater than that obtained from the Royle-Nichols model estimates that used data from cage trapping (76.68%) and camera monitoring (71.27 %) protocols.

With respect to annual monitoring costs, field methods varied two fold in expenditure (Table 2). With distance sampling (USD $19 K /yr) being the cheapest followed by camera (USD $26 K /yr) and cage trapping (USD $38 K /yr) (Table 2). Relative to their annual monitoring costs, the camera based density estimation protocol provided a 2-3 fold better benefits to cost ratio and was thus ranked superior to cage trap and distance sampling–density protocols, respectively (Table 2).

DISCUSSION

Obtaining accurate population estimates for large vertebrates, especially apex predators is often challenging and expensive but extremely important for implementing effective conservation plans (Karanth et al. 2011, Ray et al. 2005, Inskip and Zimmermann 2009; Yoccoz et al. 2001). Here we compared three field-sampling methods to estimate density for Komodo dragon at 11 sites across protected areas in Eastern Indonesia. There were two major limitations for the use of distance sampling in this study. First, the target species was not always observed during distance sampling based transect surveys despite lizards being detected using cage trapping or camera detection surveys within the same sites. This resulted in an absence of density estimates for 5 of the 11 sites. Second, far fewer than the minimum 60 required observations were obtained to permit for robust distance sampling estimation (Buckland et al. 2001) for Komodo dragons at any site. Although we used a global detection function and cluster sizes (i.e. multiple covariates distance sampling; Marques et al. 2007; Thomas et al. 2010) there were still too few observations to robustly estimate the density of Komodo dragons. Increasing the distance sampling effort (e.g. by walking more transects and/or by walking transects twice; Wingard et al. 2011) may enable detection of animals in those sites where false absences were observed and generally improve the overall encounter rate to produce better estimates. However, this could potentially require substantially more resources that would makes this field sampling method more expensive than alternative methods.

It was evident that the Royle-Nichols model estimates derived from cage trapping and camera detection based presence/absence data provided better density estimates than distance sampling. Whereby both methods permitted sufficient data for density estimates to be obtained at all sites and that their respective coefficient of variation estimates were lower than distance sampling. Though both field methods are more expensive to undertake than distance sampling their relative benefits to cost ratios clearly justified their use. Whilst both field methods permitted site-specific estimates of density, we do not know as yet how biased these estimates are. Typically occupancy models are very sensitive to estimation bias or poor model convergence when detection probability levels are low and lead to inflated estimates (Mackenzie et al. 2002; Mackenzie & Royle 2005; Mackenzie et al. 2006; Couturier et al. 2013). Future research is now needed to determine the degree of bias within the density estimates obtained from the Royle-Nichols model. This could be done by an approach similar to Couturier et al. (2013) and use simulation methods to determine what level of detection probability is required to reduce estimation bias to zero.

The second major motivation of this study was to consider which of the three different field monitoring-density estimation protocols was most cost effective. As since 2002, we have undertaken intensive cage trapping for mark-recapture studies of Komodo dragons at 10 sites on four islands in Komodo National Park, and more recently the Wae Wuul Nature Reserve on Flores (Ariefiandy et al. 2013a). For the most part, mark recapture study via cage trapping seems highly effective for documenting demographic trends in this species (Purwandana et al. 2014). However, our capacity for continuing ongoing long-term monitoring using mark recapture methods is finite given the economic and time constraints involved with this intensive method. Clearly the cheaper costs of sampling obtained from camera based methods advocate its use for future long-term monitoring of Komodo dragon population trends. Most, importantly a ~40% reduction in annual monitoring costs by replacing cage trapping with camera monitoring could permit reinvestment of expenditure from population monitoring into conservation efforts that could better serve this species (Purwandana et al. 2014). Examples of how reduced monitoring costs could better facilitate Komodo dragon conservation include funding activities (e.g. prey supplementation and assisted gene flow) to address causes of population decline on the two small islands (i.e. Gili Motang, Nusa Kode) recently identified in Komodo National Park (Purwandana et al. 2014).

More generally we see the value of applying camera monitoring frameworks to facilitate abundance or density estimation and improved conservation outputs for other threatened large terrestrial reptiles, especially other varanid lizard species. Currently many varanid species face broad scale or local population threats from direct killing for skin (used in leather products), meat and traditional medicine (Shine et al., 1998, Khatiwada and Ghimire 2009, Pernetta 2009). Further the exotic pet trade has further impacted other varanid species (Luxmore and Groombridge 1990, Jenkins and Broad 199). Introduction of invasive animals into Australia, including toxic prey and mammalian predators/competitors, are also having impacts on different varanid species (Griffiths and McKay 2007, Doody et al. 2009, Anson et al. 2013). Despite these varied and pervasive threats there remain relatively few quantitative attempts to robustly estimate demographic impacts to varanid lizard populations (Griffiths and McKay 2007, Doody et al. 2009, Anson et al. 2013a; Anson et al. 2013b), nor consider how cost trade-offs in monitoring activities could detract from alternative conservation activities needed to abate threats and recover populations (McDonald-Madden et al. 2010).

We perceive several advantages of applying camera based abundance based occupancy models to monitor Komodo dragon populations over existing mark recapture surveys using cage trapping. In conjunction with moving to a camera derived presence/absence surveys used in abundance based occupancy density estimates would considerably reduce time and labour costs and hence financial costs currently spent on trap-based Komodo dragon monitoring. Consequently, came based field monitoring protocols coupled with abundance type occupancy models could provide a potentially useful approach to achieve cost- and inference- effective monitoring necessary to inform on these species conservation requirements.

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Table 1. Site-specific Komodo dragon density estimates obtained from distance sampling and Royle-Nichols Abundance Induced Heterogeneity model using presence/absence data obtained from cage trapping and camera monitoring methods. Table reports mean survey specific detection estimates with standard error of the mean and lower and upper 95% confidence limits.

		Distance Sampling		Royle-Nichols Model			
Site	Site area (km^2)	Transect Length (km)	Site density (km^2) (95% CI)	**Cage traps/site (*N*)**	Site density ± SEM (km^2) (95% CI)	**Camera stations/site (*N*)**	Site density ± SEM (km^2) (95% CI)
Loh Liang (K1)	6.94	26.5	17.13 (8.82-36.01)	32	16.93 ± 1.54 (10.99-26.08)	32	11.07 ± 2.25 (7.39-16.6)
Loh Lawi (K2)	10.03	30	0.89 (0.1-5.8)	32	9.18 ± 0.85 (5.87-14.38)	32	6.51 ± 1.32 (4.37-9.69)
Loh Sebita (K3)	5.81	21	1.27 (0.22-7.4)	21	9.58 ± 0.97 (5.82-15.77)	21	11.13 ± 2.46 (7.23-17.04)
Loh Wau (K4)	0.83	9	11.87 (3.78-37.27)	9	38.47 ± 5.75 (20.64-71.68)	9	25.16 ± 8.48 (13.01-48.77)
Loh Buaya (R1)	5.5	12	11.13 (2.93-42.32)	22	27.53 ± 1.48 (17.91-42.32)	22	10 ± 2.24 (6.48-15.6)
Loh Baru (R2)	5.48	6	0 (0-0)	22	11.91 ± 1.37 (7.33-19.36)	22	3.78 ± 1.18 (2.01-7.01)
Loh Tongker (R3)	2.64	7.7	3.47 (0.53-22.87)	14	13.28 ± 1.67 (7.48-23.57)	14	11.19 ± 3.23 (6.36-19.73)
Loh Dasami (R4)	3.54	6	0 (0-0)	24	23.61 ± 2.2 (14.89-37.44)	24	10.78 ± 2.64 (6.64-17.22)
Gili Motang (GM)	3.9	9	0 (0-0)	16	3.59 ± 1.21 (1.7-7.58)	16	1.52 ± 0.78 (0.53-4.1)
Nusa Kode (NK)	1.07	7.6	0 (0-0)	12	6.2 ± 2.16 (2.27-16.91)	12	10.99 ± 4.26 (5.27-23.55)
Wae Wuul (WW)	14.84	28.85	0 (0-0)	26	0.99 ± 0.22 (0.5-1.99)	26	0.54 ± 0.25 (0.23-1.31)

Table 2. Benefit to cost ratio (BCR) analysis used to compare and rank three field monitoring-density estimation protocols advocated for ongoing population assessment of Komodo dragons at 11 monitoring site in protected areas of Eastern Indonesia.

Protocol	Benefit A) sites where density estimated	Benefit B) CV of density estimates(%)	Σ Benefits (A/B)	Monitoring costs (USD K/yr)	BCR	Rank (BCR)
1. Distance	6/11	149	0.04	19	0.002	3
2. Trap	11/11	77	0.14	38	0.004	2
3. Camera	11/11	71	0.15	26	0.006	1

CV = coefficient of variation

Figure Captions

Figure 1. The 11 study sites located within Komodo National Park and Wae Wuul Nature Reserve. The Komodo National Park sites comprise four Komodo Island sites (K1, Loh Liang; K2, Loh Lawi; K3, Loh Sebita; K4, Loh Wau) and four Rinca Island sites(R1, Loh Buaya; R2, Loh Baru; R3, Loh Tongker; R4, Loh Dasami). Additional Komodo National Park sites were located on each of the two small islands of Nusa Kode (NK) and Gili Motang (GM). The single Wae Wuul Nature Reserve site (WW) was located on the island of Flores immediately east of Komodo National Park. Polygons denote the boundaries of Komodo National Park and Wae Wuul Nature Reserve, and the inset depicts field site location within Indonesia.

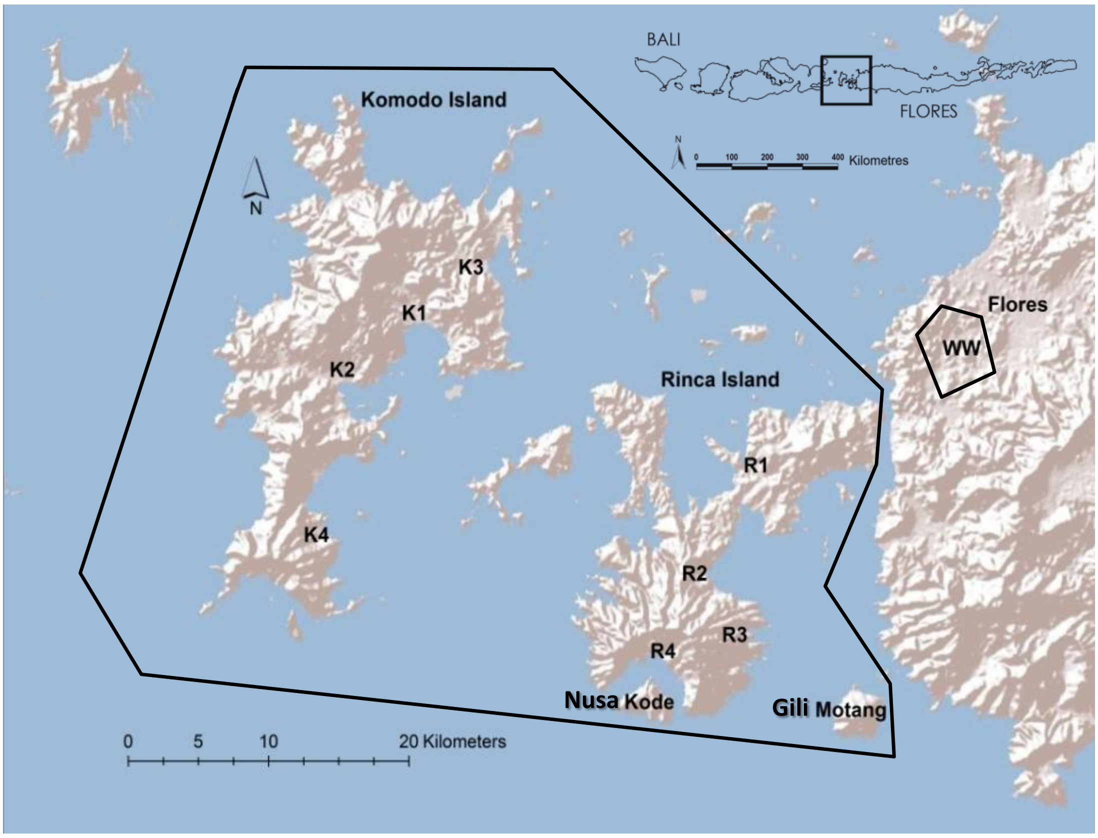

BALI
FLORES
N
0 100 200 300 400 Kilometres
Komodo Island
N
K3
K1
K2
K4
Flores
WW
Rinca Island
R1
R2
R3
R4
Nusa Kode
Gili Motang
0 5 10 20 Kilometers

Figure 1

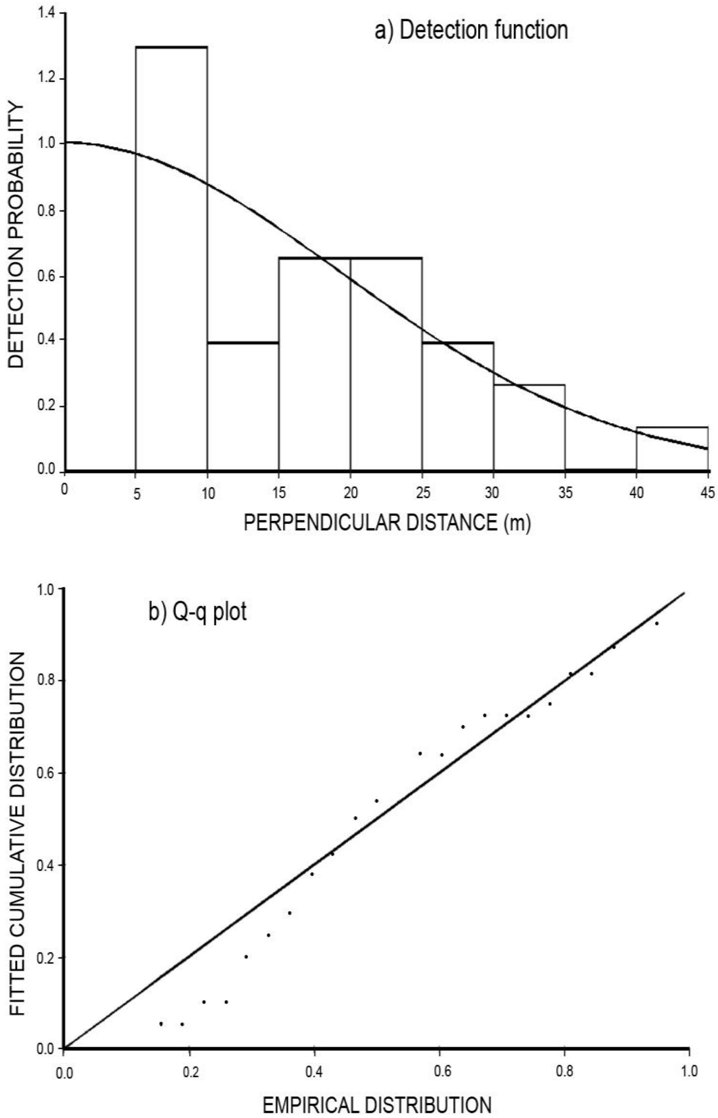
Appendix 1. Annual field monitoring costs for three field survey method used to obtain data for estimating Komodo dragon density at 11 sites in protected areas in Eastern Indonesia.

Method	Description
1. Distance Sampling along transects	
Monitoring objective	Conduct distance sampling over 165 km of transe
Monitoring Cost Component	
ancillary monitoring equipment costs	range finder/ gps/
time for monitoring (days), transportation and set up of monitoring equipment	(165 km / 10 km/days) / 1 team of people
2 permanent research officers	
additional casual field staff	person days at $10/day
running costs	boat hire/flights/accommodation/food at $85/day
Administrative infrastructure (annual)	office/storage
Running costs	
2. Cage traps	
Monitoring objective	Cage trap at 230 locations across 11 sites for 3 da
Monitoring Cost Component	
ancillary monitoring equipment costs	custom 3m Traps, carriers, GPS, maintenance
time for monitoring (days), transportation and set up of monitoring equipment	(230 [sites] x 3.2 [days]) / 8 [traps]
2 permanent research officers	
additional casual field staff	person days at $10/day
running costs	boat hire/flights/accommodation/food at $120/day
Administrative infrastructure (annual)	office/storage
Running costs	
3. Camera detection stations	
Monitoring objective	Survey 230 camera stations across 11 sites for 3 d
Monitoring Cost Component	
set up and running monitoring equipment costs	passive infrared Cameras, batteries, GPS , mainte
time for monitoring (days), transportation and set up of monitoring equipment	(230 [sites] x 3.6 [days]) / 30 [cameras]
2 permanent research officers	
additional casual field staff	person days at $10/day
running costs	boat hire/flights/accommodation/food at $95/day
Administrative infrastructure	
Running costs	

Appendix 2. Transect length, number of clusters and mean (± SE) cluster sizes observed during distance sampling of Komodo dragons at 11 sites in Komodo National Park and Wae Wuul Nature Reserve, Indonesia.

	Komodo dragons	
Site	No. clusters	Cluster size
Loh Liang (K1)	17	1.18 ± 0.10
Loh Lawi (K2)	1	1
Loh Sebita (K3)	1	1
Loh Wau (K4)	4	1.00 ± 0.0
Loh Buaya (R1)	7	1.00 ± 0.0
Loh Baru (R2)	0	0
Loh Tongker (R3)	1	1.00 ± 0.0
Loh Dasami (R4)	0	0
Gili Motang (GM)	0	0
Nusa Kode (NK)	0	0
Wae Wuul (WW)	0	0
TOTAL	31	

Appendix 3. Detection function for Komodo dragon distance sampling data collected in Komodo National Park and Wae Wuul Nature Reserve, Indonesia, in 2009-2010. The detection function (solid line in left column figures) is hazard rate key, and the histograms are the frequencies of observations. Q-Q plot for Komodo dragon distance sampling data collected in Komodo National Park and Wae Wuul Nature Reserve, Indonesia, in 2009-2010.



Appendix 4. Ranking of Royle-Nichols Abundance Induced Heterogeneity model used to estimate λ (average abundance per site) and *r* (species detectability) using cage traps (a) and camera stations (b). Table describes Akaike Information Criterion (AIC), change in AIC (ΔAIC) relative to the most parsimonious model, model weight (*w*) and estimated number of parameters (K).

Model	**AIC**	**ΔAIC**	***W***	**K**
(a) Cage Traps				
λ_{site} P_{survey}	1548.77	0	0.97	18
λ. P_{site}	1555.48	6.71	0.03	13
λ_{site} P.	1565.01	16.24	0.00	13
λ_{site} P_{site}	1570.72	21.95	0.00	24
λ. P_{survey}	1624.97	76.2	0.00	7
λ. P.	1641.42	92.65	0.00	2
(b) Camera stations				
λ_{site} P_{survey}	1379.45	0.00	0.97	18
λ. P_{site}	1387.42	7.97	0.02	13
λ_{site} P.	1388.43	8.98	0.01	13
λ_{site} P_{site}	1394.62	15.17	0.00	24
λ. P_{survey}	1426.49	47.04	0.00	7
λ. P.	1434.85	55.40	0.00	2

Appendix 5. Estimates of Komodo dragon detection probability (*r*) derived from the top ranked Royle-Nichols model obtained from cage trapping and camera monitoring surveys. Table reports mean survey specific detection probabilities (*r*) with standard error of the mean and lower and upper 95% confidence limits.

Trapping Survey Order	
(a) Cage Trapping	Estimate
Day 1 morning survey	0.14 ± 0.03 (0.09-0.2)
Day 1 afternoon survey	0.16 ± 0.03 (0.11-0.23)
Day 2 morning survey	0.19 ± 0.04 (0.13-0.28)
Day 2 afternoon survey	0.20 ± 0.04 (0.14-0.2 9)
Day 3 morning survey	0.16 ± 0.03 (0.11-0.24)
Day 3 afternoon survey	0.10 ± 0.02 (0.07-0.15)
Average	0.16 ± 0.03 (0.11-0.23)

(a) Camera Trapping	Estimate
Day 1 morning survey	0.21 ± 0.03 (0.15-0.29)
Day 1 afternoon survey	0.17 ± 0.03 (0.12-0.24)
Day 2 morning survey	0.22 ± 0.04 (0.16-0.3)
Day 2 afternoon survey	0.14 ± 0.03 (0.1-0.2)
Day 3 morning survey	0.21 ± 0.04 (0.15-0.3)
Day 3 afternoon survey	0.12 ± 0.02 (0.08-0.17)
Average	0.18 ± 0.03 (0.13-0.25)