Hunting pressure a key contributor to the impending extinction of Bornean wild cattle

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Abstract

Widespread and unregulated hunting of ungulates in Southeast Asia is resulting in population declines and localised extinctions. Increased access to previously remote tropical forest following logging and changes in land-use facilitates hunting of elusive wild cattle in Borneo, which preferentially select secluded habitat. We collated the first population parameters for the endangered Bornean banteng and developed population models to simulate the effect of different hunting offtake rates upon survival and the recovery of the population using reintroduced captive-bred individuals. Our findings suggest that the banteng population in Sabah is geographically divided into four management units based on connectivity; the northeast, Sipitang (west), central and southeast, which all require active management to prevent further population decline and local extinction. With only 1% offtake, population growth ceased in the northeast and Sipitang. In the southeast and central units, growth ceased at 2% and 4%, respectively. Extinction was estimated at 21-39 years when offtake was 5%, occurring first in Sipitang and last in the Central unit. Supplementing the population with captive-bred individuals suggested that inbreeding was likely to limit population growth if using 20 founder individuals or less. Translocating two individuals for a 10-year period, starting after 20 years of captive breeding resulted in a faster population recovery over 100 years and a lower extinction probability. Our results suggest that shielding the population against further losses from hunting will be key to their survival in the wild, providing active management in the form of captive breeding is developed in the interim.

Keywords: Hunting, population modelling, VORTEX, tropical forest, Borneo.

Running page head: Impending extinction of banteng in Borneo

1.0 Introduction

Unsustainable hunting of wild animals for human consumption is an acute problem in Southeast Asia, where the impacts are exacerbated by globally unmatched rates of deforestation (Dobson et al., 2019). While there is increasing evidence that widespread and intensive hunting across the tropics has resulted in defaunation, the true extent of over-hunting and its specific impacts upon animal communities are poorly understood (Tilker et al., 2019). Ungulates often suffer widespread population declines, which are largely due to the increased pressures of agricultural development and expansion of concession land, habitat encroachment and unregulated hunting (Melletti & Burton, 2014; Steinmetz et al., 2010). In Borneo, there is strong evidence to suggest that hunting for ungulates has always been prevalent and was common-place in prehistoric indigenous communities (Aubert et al., 2018; Chazine, 2005), providing a major source of protein (Bennett et al., 2000). Today, however, hunting in Borneo is conducted by urban and rural communities for different purposes and using more sophisticated methods, and has likely reached unsustainable levels for the endangered Bornean banteng (*Bos javanicus lowi*). Differences in motivation for hunting are known to exist and the general methods employed are discussed at length by Harrison et al. (2016). The motivations for specifically targeting wild cattle appear to vary, ranging from subsistence hunting by locals and forest contractors for personal consumption, gifting of whole carcases for celebrations and festivals, sport hunting, and the acquirement of trophies for personal status or trade for medicinal properties (Harrison et al., 2016). Furthermore, Bornean banteng predate on crop plants in the grasslands of East Kalimantan (Indonesian Borneo) and are therefore perceived as a pest species and hunted (Hedges & Meijaard, 1999). Firearms and dogs have been the commonly-used methods since the early 19th century (Rutter, 1922), but the prospects of hunting success have dramatically improved given the provision of semi-automatic weapons and firearm sights, together with off-road vehicles, excavators used in timber extraction, boats and motorbikes. Bornean banteng are also caught in snares; enduring major injuries such as hoof dismemberment as a consequence of single-strand snares set at ground level along trails and abandoned logging roads in the forest (Gardner et al., 2019).

Over the past three decades, a declining trend in Bornean banteng has been described (Boonratana, 1997; Davies & Payne, 1982; Hedges & Meijaard, 1999; Olsen, 2003; Timmins et al., 2008). Localised extinctions have occurred in Sabah (Malaysian Borneo) (Melletti and Burton, 2014), in areas where previously remote forest was opened up by timber harvesting. First-hand reports from logging contractors in the west, central and northeast regions of Sabah describe sourcing of wild cattle bushmeat for logging camps during the 1980s and 1990s, thus reducing and in some cases, completely eradicating, any remaining individuals. In years following the extraction of timber, the access provided by abandoned logging roads has also almost certainly increased hunting pressure (Kleinschroth & Healey, 2017).

The Bornean banteng is a ‘Totally Protected’ species and listed under Schedule 1 of the Sabah Wildlife Enactment 1997 due to their small population size, which is currently estimated at 326 individuals (Sabah Wildlife Department., 2019). Poaching of Bornean bantengs in Sabah is therefore not permitted (Sabah Wildlife Department., 2019). Evidence of banteng poaching is challenging to obtain; it is seldom reported to the authorities, and the majority of incidences surface primarily via social media (Fig. 1) and are occasionally covered by local newspapers (Sarda, 2016; Sario, 2015). Major losses have been sustained over recent years, but the majority remain undocumented and with insufficient evidence or resources to bring about prosecutions. Losses are largely evidenced by photographs of hunters with their trophy and trophy heads on display in rural houses (Hedges & Meijaard, 1999). Recent known cases of poaching communicated through social media have documented the loss of four individuals in 2017, and three bulls in 2018, both within Sabah

(Goossens, pers. comm.). The number of reported cases is, however, thought to be unrepresentative of the actual number of sustained deaths, but regular offtake is likely to be unsustainable for this species that is already extremely low-density in Borneo owning to recent threats.

Studies of wild bantengs are relatively infrequent; the most widely documented subspecies are B. j. birmanicus in the Eastern Plains of Cambodia (Gray et al., 2011, 2012; Gray, 2012; Gray and Phan, 2011; Pin et al., 2020), B. j. javanicus in Java (IUCN-SSC Asian Wild Cattle Specialist Group, 2010; Pudyatmoko, 2017; Pudyatmoko and Djuwantoko, 2006), and a feral population of B. javanicus, presumed to be B. j. javanicus owing to their Balinese origins (see Bradshaw et al., 2006), in Northern Australia (Bradshaw et al., 2007; Choquent, 1993). In contrast, with limited baseline data available on the demography of Bornean bantengs and large gaps in our understanding concerning the impacts of hunting, planning optimal conservation procedures and measuring the effectiveness of mitigation strategies is extremely difficult. This study therefore aimed to collate the first set of biological parameters for the Bornean banteng and banteng per se (*B. javanicus)* using data from studbooks, published research and circumstantial data, and combined with anecdotal information on hunting prevalence. Population models were developed and parameterised using this new compilation of biological data to simulate the survival of Bornean banteng in Sabah given differing scenarios of offtake from hunting, and the recovery of the population by supplementing herds with reintroduced individuals bred in *in-situ* captivity.

2.0 Materials and Methods

Given the relative paucity of information on the intrinsic (biological) and extrinsic (environmental) factors influencing the population biology of the banteng species in general and the endemic Bornean banteng in Sabah in particular, a number of approaches were taken to maximise the reliability of the modelling that was undertaken. First, the IUCN Species Survival Commission (SSC) Asian Wild Cattle Specialist Group (AWCSG) collated all available information for all currently recognised subspecies of banteng (*B. j. lowi*, *B. j. javanicus*, and *B. j. birmanicus*) and for the species as a whole (*B. javanicus*) where the subspecies designation could not be attributed. Literature was identified through communications with members of the AWCSG, from the IUCN Red List Banteng species account, Chester Zoo, Copenhagen Zoo, San Diego Zoo, Houston Zoo, the Sabah Wildlife Department, the Sabah Forestry Department, and local printed newspapers. Data sources included the European and American studbook databases, direct observations recorded in the wild during field work, circumstantial information from stakeholders involved in species and forest management, newspaper articles and social media messages detailing hunting incidents of bantengs, and published literature available through scientific search engines Scopus, Web of Science and Google Scholar. This information was used to develop the baseline model for the species (Appendix: available upon request, key baseline parameters in Table 1). Second, a general protocol was adopted that would enable the model to be developed in a structured way, minimising unnecessary complexity: starting with: 1) developing a baseline model focusing on the intrinsic biological parameters that were obtained from literature and field research, and were agreed at a stakeholder consultation process; 2) using the intrinsic rate of growth from the above model to test whether the baseline model was parameterised realistically, followed by sensitivity testing of those parameters that were less certain to refine the model; 3) configuring a metapopulation model based on current estimates from forest areas known to contain banteng, including an appreciation of the connectivity among these areas or lack thereof; 4) modelling a variety of harvesting offtake rates at 1, 2, 3, 4 and 5% of the starting population size for all populations that were based on recent poaching events in Sabah, to assess how different hunting rates affect population viability; 5) evaluating management interventions including the cessation of hunting, the establishment of a captive population and its early use in population augmentation and the connection of two isolated subpopulations in Eastern Sabah.

Following the agreement of the baseline model at the consultation process, a population-specific model was initiated, initially comprising five subpopulations across Sabah, namely Sipitang, Southeast, Central-south, Central-north and Northeast regions (Fig. 3). Population estimates were provided by the AWCSG, and carrying capacity was estimated for each region based on a mean density estimate of 0.66 individuals per km^2 for intact forest and 0.33 per km^2 for disturbed forest, values extracted from IUCN data using density estimates for relatively undisturbed forests in mainland Asia, specifically Cambodia where it is estimated that an excess of 1,000 banteng existed across 15,000 km^2 (Gray et al., 2011), but only after 90% of the population were estimated to have been removed by uncontrolled hunting (Gardner et al., 2016). Preliminary density estimates in Sabah are much lower and range between 0.61-0.95 (0.32-1.66) individuals per 100 km^2 for Tabin Wildlife Reserve and 0.55-0.56 (0.21-2.09) for Malua Forest Reserve, based on natural marks using a spatially explicit capture-recapture framework (Gardner et al., 2019). Validation of the parameters used in the modelling was conducted through consultation with experts at the first banteng population viability working group held on the 27th November 2017 in Kota Kinabalu, Sabah (Malaysian Borneo) prior to modelling. Revisions to some model parameters were made during consultation with experts at the population viability working group, hereon referred to as PHVA group (Table 2).

A number of banteng poaching events have recently been recorded and it is estimated that 70% or more go unrecorded (PHVA group). Recent known hunting events of banteng in Sabah (12 individuals killed in the past 12 months, Goossens, pers. comm.) indicates that approximately 4% of the total Sabah population may be poached annually at the present time. A variety of harvest models were developed to examine offtake rates at 1, 2, 3, 4 and 5% of the starting population size for all populations. Additional models were proposed in order to examine the combined effect of sustained offtake with catastrophic poaching events (mass killings), but these were not pursued because they could not be reliably quantified.

Further modelling was carried out, first to model the establishment of a captive population as a resource for future translocations in the wild. This was founded with 20 individuals (including 15 individuals from the Northeast population if they need to be protected in a captive environment), and where one male and one female are harvested for translocation into the wild over a 10-year period, starting at year 10 and year 20.

All population modelling was carried out using Vortex v10 (Lacy & Pollak, 2017). Key literature used to guide this process included Lacy et al. (2015). Sensitivity testing based on the percentage of females breeding per year was carried out using parameters from Table 1 and 1,000 simulations.

3.0 Results

3.1 Development of a baseline model

Owing to shortages of biological data and the supplementation of population parameters from other banteng subspecies, our model parameters were not considered exclusive to *B. j. lowi* and lack extrinsic parameterisation. Sensitivity testing based on a higher number of females breeding per year (50%) with 1,000 simulations yielded an intrinsic growth rate of 4.3% ± 5.7 (SD). The results of this simulation can be found in Fig. 2.

3.2 Population model

The estimated population-specific parameters comprised population size, carrying capacity and its trend, and can be found in Table 3. For the five-population model, the carrying capacity (K) for Sipitang (current population size estimated, n=33) was initially estimated at 826 but revised to 301 owing to past and current management practices employed that have reduced suitable habitat; natural forest management (36.4%) and intensive tree plantations (ITP) that is clear-felled on rotation (63.6%). We excluded ITP from carrying capacity estimates owing to importance of natural forest cover, primarily tropical lowland dipterocarp, as key habitat for the banteng in Sabah, which was modelled by Lim et al. (2019). In other regions the population size and carrying capacity was n = 52 (later revised to 82) and K = 872 for the Southeast (Tabin, Kulamba); for Northeast (Paitan, Sugut) n = 35, K = 267; for Central-south (Maliau, Segama, Malua, Sapulut) n = 121, K = 3,642 and for Central-north (Deramakot, Segaliud-Lokan, Tangkulap) n = 85, K = 551. Following the initial estimation of K for Sapulut within the Central-south unit, further revision was also made given the history of extensive logging over the past 35 years; the proportions of forest that may, in reality, provide suitable habitat for banteng are smaller than the total area (4% conservation, 61% natural forest management in 2009). The carrying capacity is probably circa K=497, reducing the Central-south to K=3,374.

Given the estimated small population sizes present, inbreeding depression was modelled for all populations using the default vertebrate values in Vortex in the absence of other data (6.29 lethal equivalents and 50% of inbreeding depression due to recessive lethal alleles). The results of the first population-based simulation yielded growth rates varying between 0.011 (Sipitang) and 0.034 (Central-south) and only Sipitang and Northeast yielded any simulations trending to extinction (1% and 8%, respectively). The population-based model was then revised to a) increase the population size estimate for the Southeast population to 82 (including the value from Kulamba), and to b) amalgamate the central populations because animals can and have been observed dispersing (river crossing) between the two (PHVA group). At the same time the Central-south population estimate was revised upwards to 170 to account for individuals that may have existed in locations that were not surveyed, yielding a total central population estimate of 255 individuals.

3.3 Hunting

Figs. 4a and 4b shows the extinction impact of 3%, 4% and 5% hunting over a 40-year period in all subpopulations. A 3% offtake resulted in high extinction probabilities in all subpopulations except Central over a 40-year period and that 5% offtake had catastrophic consequences with no population having a probability of survival exceeding 40%. Median time to extinction for 5% poaching across subpopulations was just 24 years. Further poaching models showed that as little as 2% offtake per year, when combined with catastrophic hunting episodes, resulted in major population declines and extinction for all subpopulations except Central (not shown). However, when hunting was stopped after 50 years, the Central and Southeast populations could recover but only if further hunting occurred at 1% per annum or less (Table 4). Hunting was, as expected, a deterministic force for extinction in the model and we predicted that it needs to occur at very low levels (<1% per annum, if it goes on beyond 50 years). The current rate of harvest on the ground in Sabah, which may be as high as 4% per annum, is expected to result in the extinction of the Bornean banteng with 39% to 96% global probability (4 and 5% offtake, Table 4).

3.4 Additional modelling

Fig. 5 shows the consequences of this management strategy. Given that the population would be founded by just 20 individuals (three males and 17 females in three groups) it is important to include the possibility of inbreeding depression (red line, Fig. 5) and it is clear that ensuring that inbreeding is avoided during captive breeding is paramount if the population is to be productive (probability of extinction 0.58 versus 0.03). In addition, although both translocation strategies can provide individuals for translocation, the strategy that starts to supply individuals at Year 10 had a much higher probability of extinction at 0.2, compared with 0.08 when supplying individuals starting at Year 20. It was also noted that Kulamba and Tabin are currently unconnected but efforts are underway to reconnect the two areas; this may facilitate movement between both populations depending upon the corridor dimensions, security and availability of refuge provided by the vegetative cover. Modelling gene-flow between these populations confirmed that connection of these two isolated populations under a model including inbreeding depression would more than double the final population size over 100 years as opposed to keeping them isolated, but that there was very little difference if the reconnection happened at 5, 10 or 20 years from now.

4.0 Discussion

Our study provides the first extensive compilation of biological parameters from multiple sources for the banteng, and is also the first example of the PHVA approach applied to the Bornean banteng to model population projections in Sabah, and indeed for the banteng *per se* in any country. The collated parameters may serve as a foundation for modelling the population projections of other wild cattle species in mainland Southeast Asia and Java, and inform models investigating apex predators, such as the tiger (*Panthera tigris*) and dhole (*Cuon alpinus*). Given the provision of recent field data and information garnered from the consultation process with stakeholders and experts, we delineated four subpopulations of bantengs in Sabah – Sipitang (West), Southeast, Northeast, Central. Separate management for these four units is advocated, owing to their isolated nature and the absence of connectivity between forests, and the high level of difficulty for bantengs traversing the landscape that is dominated by vast monoculture plantations where they risk persecution (Bajomi & Takács-Sánta, 2011).

From our baseline model, our population growth rate was far lower than that reported by Hone et al. (2010) for feral B. javanicus in northern Australia, possibly owing to greater impacts from deterministic factors such as habitat disturbance, degradation and loss, and over-harvesting (Lacy et al., 2015). The results of our population-based models revealed that the vulnerability of extinction was greatest for the Sipitang and Northeast management units, due to their existing small population size and low annual growth rates that ultimately influenced the long-term declining trend in the population size. Our estimations of the impact of inbreeding are based on default values for vertebrates however, and the severity of the inbreeding depression varies widely among species (Lacy, 2019). B. j. lowi may deviate from these default values, thus it may be safer to assume our results are a best-case scenario for this declining subspecies. With the inclusion of hunting offtake rates, the cessation of population growth for these two management units occurred when offtake was low at only 1%. When systematically increased, population growth ceased for the Southeast unit at 2%, and for the Central unit at 4%. At 5%, the projected time to extinction was very short, with the complete loss of the species occurring in the next 21-39 years, with the first extinction occurring in Sipitang and the last in the Central unit. The frequency of evidence that has surfaced on social media in recent years, testifying the losses of bantengs from hunting in Sabah, indicates that 5% offtake is not an unrealistic reality on the ground. Localised extinctions have already occurred in Sabah during the last three decades (Melletti & Burton, 2014), and almost certainly within Kalimantan; hunting trophies confirmed the loss of at least 25 individuals in one area of Kalimantan during the 1990s (Hedges & Meijaard, 1999). Observations of the species in Kalimantan are now few, with only a handful of photographic captures on camera traps between 2012-2013 (P. Gardner, pers. comm.). Without further regulation and control, the loss of individuals from hunting will drive the demise of the species quicker than any other factor modelled. Despite the overall small population size of bantengs in Sabah, the inbreeding depression was found to be an extinction risk for only the two smallest and most geographically isolated units in the west (Sipitang) and northeast of Sabah. Our estimated time to extinction is comparable with that of the low-density sable antelope (*Hippotragus niger*) in Zimbabwe (Capon et al., 2013), which was predicted to decline at a rate of 16.7% per annum with extinction occurring after 18 years, but this was dependent on the observed baseline rates over a 5-year period (2000-2005). During this time, the population decline was driven by frequent losses of juveniles by lion predation and not, like our study, through poaching. Supplementing the population with 30 individuals slowed the decline marginally over the forecasted 100 years, however the most impactful risk mitigation measure was thought to be effective management of predator populations (Capon et al., 2013). A similar study that evaluated the effect of different management intervention scenarios on the declining Baird's tapir (*Tapirus bairdii*) in Honduras found that the current hunting levels would result in extinction in the next 100 years (McCann & Wheeler, 2012).

Reducing Baird's tapir offtake rates and frequency (to one adult per gender every three years) resulted in a population with positive growth over 100 years even if the initial population was reduced to ~20 individuals (McCann & Wheeler, 2012). This was an optimal interim strategy, which would allow time to establish recruitment mechanisms (McCann & Wheeler, 2012).

A captive breeding programme using 20 wild-caught founder individuals, primarily from the northeast unit (due to the small and isolated population in this location with the lowest carrying capacity), resulted in an inbreeding depression and a decline in the captive population after 40 years if no individuals were translocated to the wild. Translocation of individuals into the wild after 20 years of captive breeding had a lower extinction probability than the 10-year plan (Sabah Wildlife Department., 2019); this prolonged strategy would result in a quicker recovery of the population and would be a more effective use of biological and financial resources. With no drastic management intervention to stem the loss of individuals in the 20-year interim of captive breeding, extinctions would certainly be occurring within the management units and may negate the effort of translocations. More stringent and effective anti-poaching measures and enforcement of legislation are therefore fundamental to the bantengs' survival in the wild over the next two decades.

The severe habitat fragmentation and clearance for oil palm present severe movement difficulty for bantengs, especially in the northeast and, to a lesser-extent, in the west (Bajomi & Takács-Sánta, 2011). As such, dispersal of bantengs in these units to exchange gametes and maintain or increase genetic diversity is improbable. Coupled with intensive poaching using firearms within and around the management units and the injuries sustained by banteng from snares (Gardner et al., 2019), the long-term viability of these herds is of upmost concern. At present, translocation of individuals into these management units in order to bolster the genetic pool would not be an effective strategy given the unsecure nature of the reserves and the probability of eradication by poachers. No intervention, however, is not an option; given the advancing population decline, conservation of all individuals is required to retain genetic diversity within the population and the effective population size. Indiscriminate rescue-capture of some or all individuals from the Northeast unit for the establishment of a captive breeding programme may be a more productive strategy, thus ensuring conservation of individuals and their genetic diversity against further catastrophic environmental events including fires, which have decimated banteng forest habitat in the northeast (P. Gardner, pers. obs). Indiscriminate rescue-capture would be a blind process without prior knowledge on their genetic variation as obtaining high quality DNA has not been possible to-date, therefore correct management of breeding pairs and their genetic diversity would be integral to minimising inbreeding depression within a captive setting. Low genetic variability is currently an issue for captive-bred Malayan gaur (*Bos gaurus hubbacki*) in Peninsular Malaysia, as the programme commenced with few founder individuals and has resulted in multiple progeny with shared parents, and this now affects the survival rate of new-born calves (Md-Zain et al., 2019; Rosli et al., 2016). Avoiding total reliance on few founder individuals is paramount, as unforeseen biological complications like unviability could confound reproduction; *in-situ* captive breeding of the Sumatran rhinoceros (*Dicerorhinus sumatrensis*) in Sabah suffered a devastating catastrophe following the recent death of the last remaining wild-caught rhino due to uterine tumours (Gokkon, 2019). More than one *in-situ* breeding facility supplied with multiple wild-caught individuals originating from different management units would be preferable in Sabah. Not least to minimise mortality arising from disease transmission but to maintain wildlife security and avoid unintentional reintroduction, like that of the escaped captive-bred banteng in Thailand (Chaiyarat et al., 2017). Accidental release in Sabah is a real possibility considering the extensive damage caused by Bornean elephants (*Elephas maximus borneensis*) to a banteng forest enclosure in 2012-2013 (P. Gardner, pers.obs).

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Figures

Fig. 1: Bornean wild cattle are poached frequently in Sabah, but records of losses are difficult to obtain due to the secretive nature of the trade in banteng bushmeat and products. Evidence occasionally surfaces on social media where poachers often exchange, or in reports such as that by Olsen (2002) - lower left image.

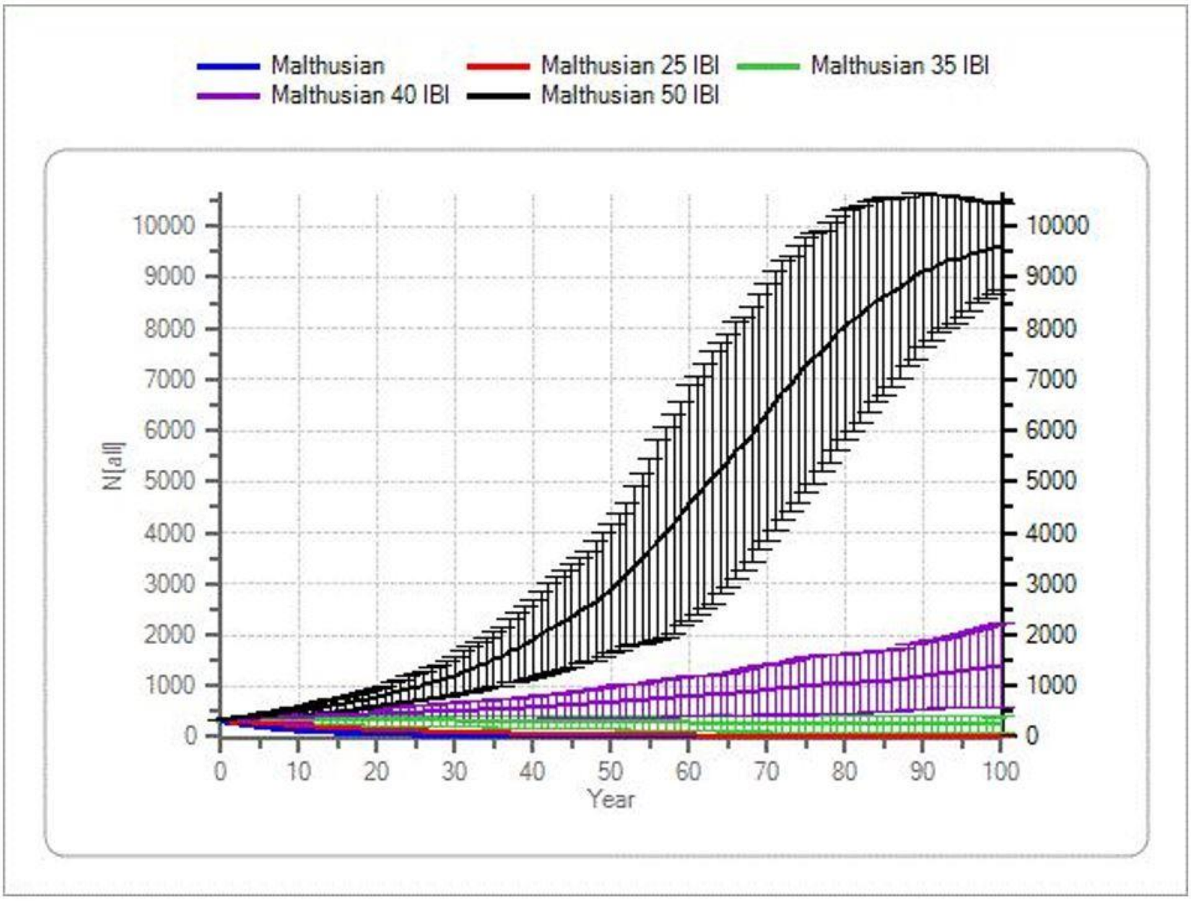

Fig. 2: Population size response (± SD) in the baseline model varying interbirth interval (percentage of females breeding per year, ranging from 15.8% [blue line] to 50% [black line]).

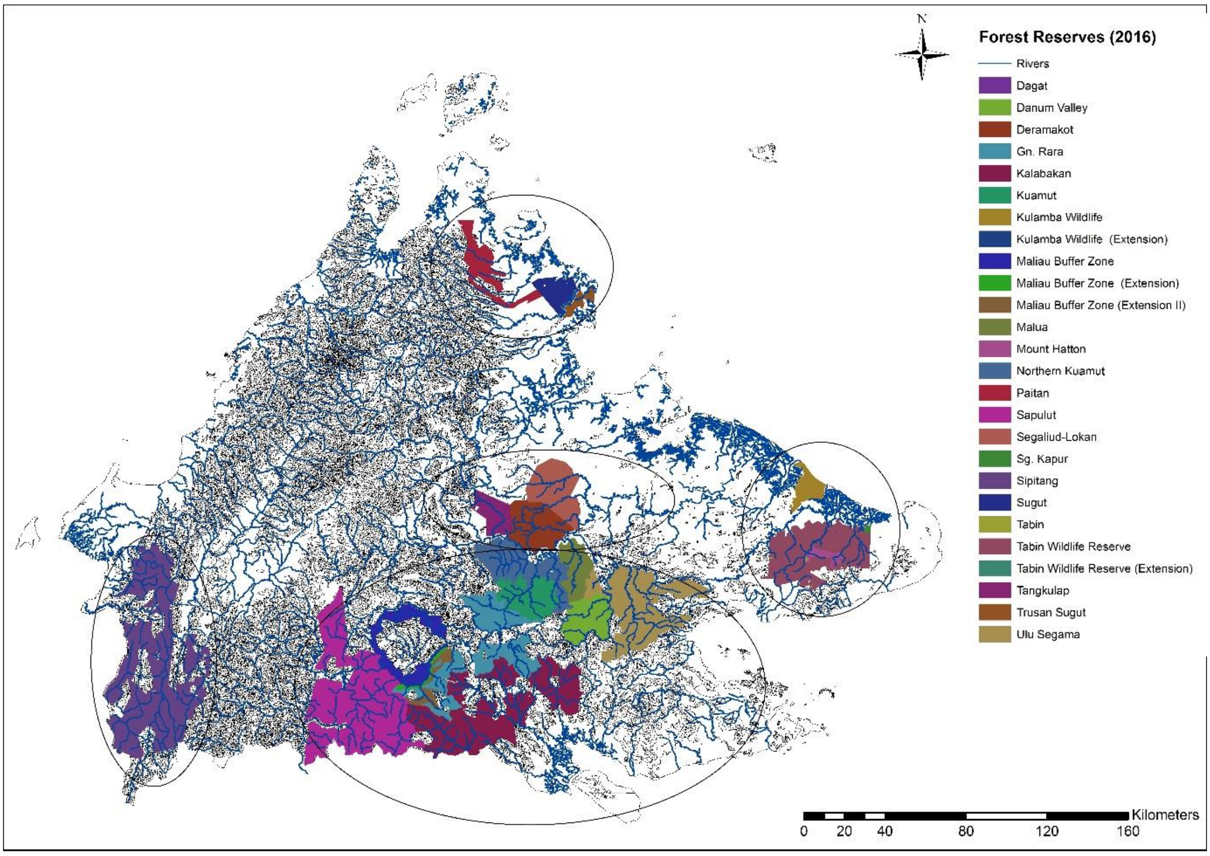

Fig. 3: The Bornean banteng foci population and five management units used for population viability modelling, located in the northern Malaysian state of Sabah, Borneo. The map was generated using ArcGIS® software version 10.1 by ESRI, with data from Natural Earth and the Sabah Forestry Department.

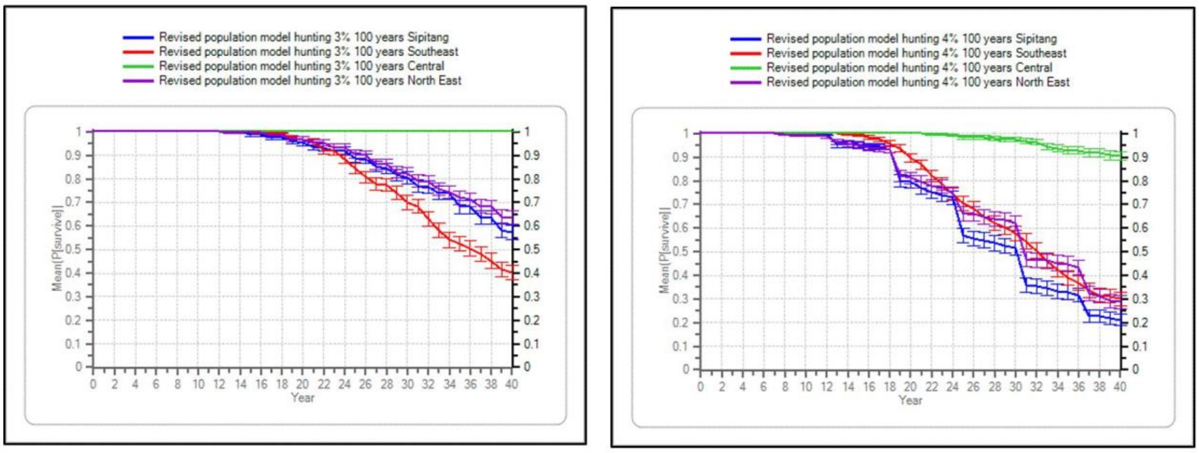

Fig. 4a: The probability of Bornean banteng survival (± SE) as a function of 3% and 4% annual offtake in each of the four subpopulations identified across the Malaysian state of Sabah.

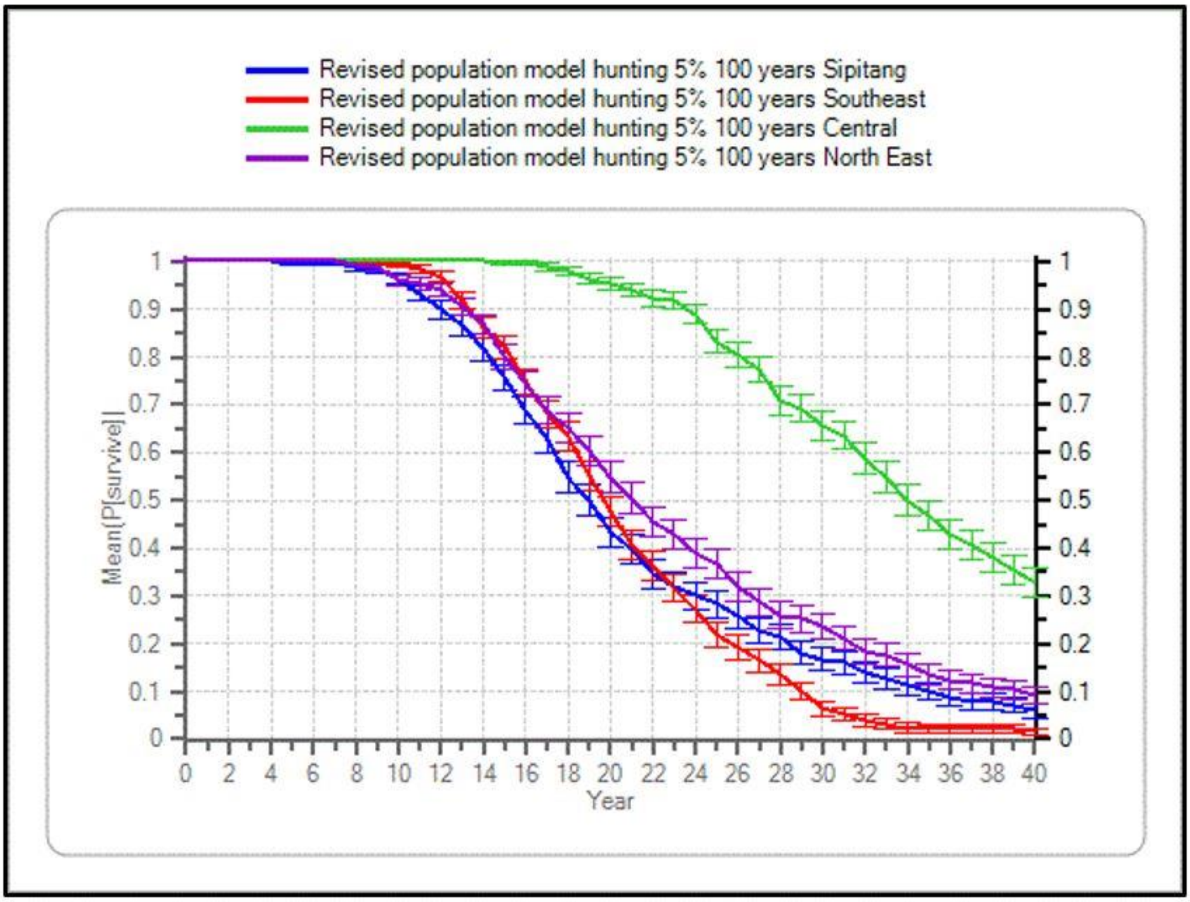

Fig. 4b: The mean probability of Bornean banteng survival (± SE) as a function of 5% annual offtake in each of the four subpopulations identified across the Malaysian state of Sabah.

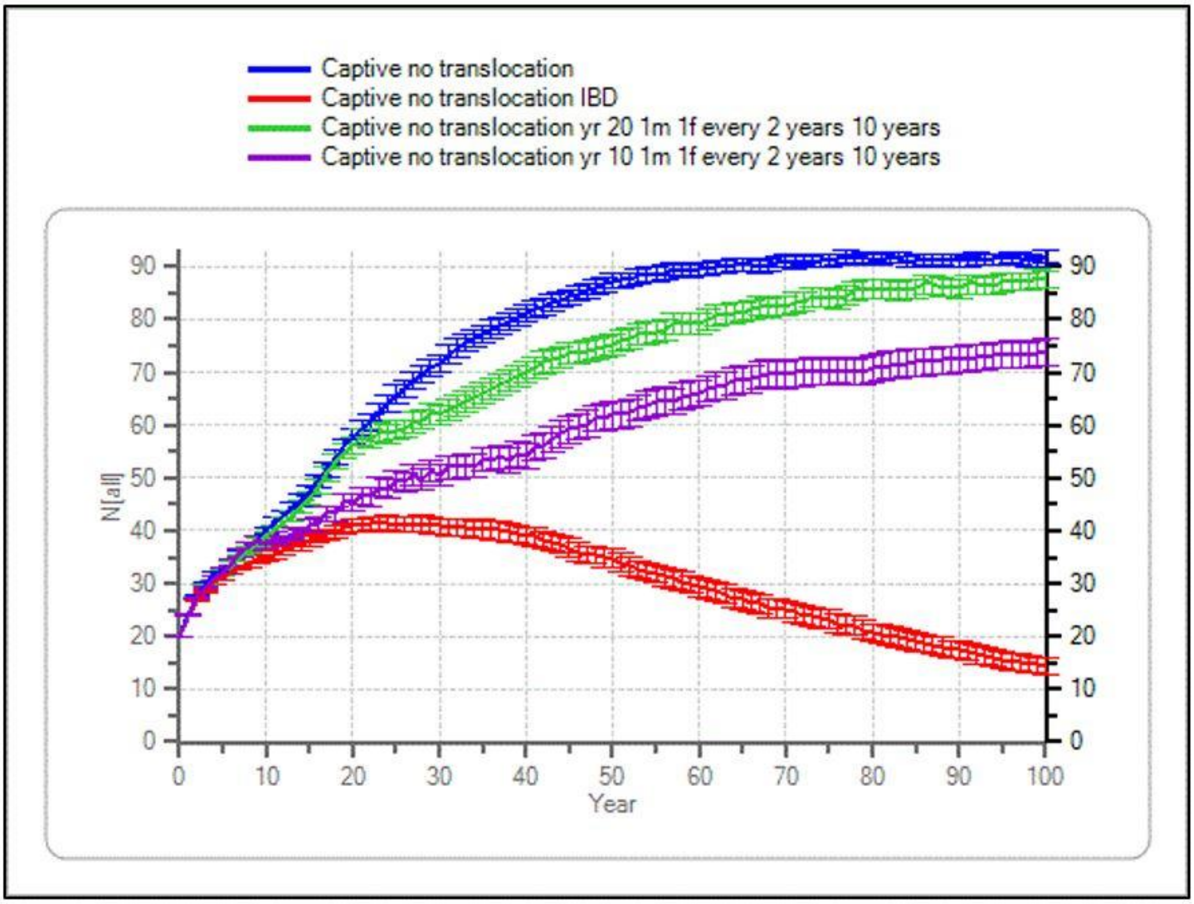

Fig. 5: Effects of exemplar captive management and translocation strategies on viability of captive Bornean banteng in Sabah.

Tables

Table 1: Population parameters for three banteng subspecies that were collated from various sources including studbooks, observations in the wild and the literature, and were used for modelling the survival of Bornean banteng in Sabah.

Parameter	Model value(s)	Justification
Number of years modelled	100	Approx. 10 generations
Correlation between reproduction and survival as a function of environmental variance	0.2	Unknown but expected to be relatively low due to K-selected nature of the species (i.e. low years for reproduction are not always low years for survival)
Reproductive system	Short term polygyny	Molecular analysis of feral banteng, Australia (Bradshaw et al. 2007)
Age of first offspring females	3	Gardner et al (AWCSG)
Age of first offspring males	3	Gardner et al (AWCSG)
Max age female reproduction	18	US studbook (AWCSG)
Max age male reproduction	19	US studbook (AWCSG)
Max lifespan	30	Unknown but thought to be 25+
Max number of calving events per year	1	AWCSG
Max number of offspring per calving event	2	AWCSG from *Bos javanicus javanicus* (twining frequency 0.012)
Sex ratio at birth	50:50:00	No other data
Density dependence	No	Intrinsic model
% adult females breeding per year (interbirth interval)	15.8%, 25%, 35%, 40%, 50%	Lower bound estimated from US studbook, values tested up to one offspring every two years (most commonly reported value for wild *Bos* species, including banteng)

Female mortality ± SD	20% ± 10 (Age 0-1); 2% ± 5 (Age 1-2); 4% ± 5 (Age 2-3); 11.9% ± 2 (Age 3+)	For subadults, AWCS data from *Bos javanicus javanicus*. For adults, estimated from the US studbook (both sexes) – equates to a max lifespan of ~28 years
Male mortality ± SD	26% ± 10 (Age 0-1); 8% ± 5 (Age 1-2); 4% ± 5 (Age 2-3); 11.9% ± 2 (Age 3+)	For subadults, AWCS data from *Bos javanicus javanicus*. For adults, estimated from the US studbook (both sexes) – equates to a max lifespan of ~28 years
Catastrophes	None	Intrinsic model
% males in breeding pool (can *potentially* breed)	100%	No additional data to start with
Initial population size	326	AWCSG – current best estimate for Sabah
Carrying capacity	10,000	Set as effectively infinite (so as not to limit population growth)

Table 2: Revised model parameters made during consultation with experts at a working group held in Kota Kinabalu, Sabah (Malaysian Borneo) on the 27th November 2017.

Parameter	**Model value(s)**	**Justification**
Inbreeding depression?	Yes	Relevant for smaller populations
Age of first offspring males	5	PHVA group assessment
Max age female reproduction	16	PHVA group assessment
Max age male reproduction	16	PHVA group assessment
Max lifespan	20	PHVA group assessment – lifespan shorter in the wild than in captivity
Max number of offspring per calving event	1	PHVA group assessment – can ignore very rare twinning events
% males in breeding pool (can *potentially* breed)	20%	PHVA group assessment – bachelor groups are common and harems comprise one adult male to five adult females.

Table 3: The population and carrying capacity estimates for Bornean banteng in Sabah used in population viability modelling.

Reserve	Forest class	Notes	Banteng	Area (km2) of suitable habitat	Good/ Intact %	k (km2 x 0.66 or 0.33)	N actual
Maliau Buffer Zone (Extension)	1	Protection Forest: Forest conserved for the protection of watershed and stability of soil, water conservation, and other environmental factors. Logging is not permitted in these areas.	Present	51.77	70-100	34	
Maliau Buffer Zone	1		Present	466.39		307	
Ulu Segama	1		Present	1271.16		838	
Maliau Buffer Zone (Extension II)	1		Present	138.02		91	40
Malua Forest Reserve	1		Present	339.31		223	36
Tangkulap Forest Reserve	1		Present	272.75		180	17
Trusan Sugut Forest Reserve	1		Present	86.49		57	
Sugut Wildlife Corridor	1		Present	2.97		n/a	
Gn. Rara Forest Reserve	2	Commercial Forest: Forests allocated for harvesting to supply timber and other produce contributing to the state's economy. Harvesting carried out using Sustainable Forest Management (SFM) Principles.	Present	1212.17	50-70	400	
Kalabakan Forest Reserve	2		Present	1819.72		600	
Kuamut Forest Reserve	2		Present	1163.71		384	30
Segaliud-Lokan Forest Reserve	2		Present	572.83		189	46
Deramakot Forest Reserve	2		Present	550.57		182	22
Sapulut Forest Reserve	2		Present	1506.11		497	15
Paitan Forest Reserve	2		Present	417.36		138	18
Sipitang Forest Reserve	2		Present	911.60		110	33
Sugut Forest Reserve	2		Present	232.71		77	15
Tabin Forest Reserve	6	Virgin Jungle: Forests conserved intact strictly for forestry research including biodiversity and genetic conservation. Logging strictly prohibited.	Present	4.08	70-100	n/a	
Dagat Forest Reserve	6		Present	1.62		n/a	
Mamahat Forest Reserve	6		Present	0.78		n/a	
Sg. Kapur Forest Reserve	6		Present	12.33		n/a	
Tabin Wildlife Reserve	7	Wildlife Reserves: Forest conserved primarily for the protection and conservation of wildlife. Logging prohibited.	Present	1117.92	70-100	738	52
Tabin Wildlife Reserve (Extension)	7		Present	2.65		n/a	
Kulamba Wildlife Sanctuary	7		Present	203.39		134	
Kulamba Wildlife Sanctuary (Extension)	7		Present	2.14		n/a	