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Potential cane toad short to medium term control techniques – the biological feasibility and cost of exclusion as a mitigating control strategy

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1. Executive Summary

1.1 Invasive cane toads *Bufo marinus* are spreading rapidly across northern Australia, raising serious public and scientific concerns for the long-term persistence of many potentially vulnerable wildlife populations. A number of mitigating solutions to the cane toad problem have been proposed, including direct killing of toads, biological control, or the establishment of secure areas from which cane toads are excluded.

1.2 Here we address the exclusion strategy, seeking in particular to provide estimates of the cost of isolating long-term viable populations of 12 species of susceptible native fauna managed to remain free of cane toads, using advanced methods in population viability analysis modelling combined with our experience in wildlife management in the difficult environments of northern Australia.

1.3 Our results reveal a relatively wide disparity across different taxa in the minimum habitat areas required for long-term persistence, ranging from as little as 16 km^2 for mangrove monitors, to 220 km^2 for northern quolls, to vast areas of up to 50 000 km^2 for wide ranging species such as wedge-tailed eagles.

1.4 An area the size of the Cobourg Peninsula (2 207 km^2), which would be relatively cost-effective to isolate as a landscape-scale exclosure, appears to be large enough to support viable populations of most small mammal and reptile species such as quolls, goannas and predatory snakes (and presumably most amphibians and insects, which also have relatively small home range requirements or high average densities), though it would still fail to capture fully the areas for some of the largest free-ranging species, include most top avian predators such as wedge-tailed eagles. For Garig National Park, the cost of construction of a exclusion fence across the neck of the Cobourg Peninsula of 6 km length would be approximately \$3.6–5.7 million, with annual maintenance costs in the range of \$0.4-0.9 million.

1.5 In reality the situation is more complex, because captial and recurring costs for the construction and maintenance of fencing are governed not only by the size of the area to be enclosed, but also by the choice of location (e.g. it is more efficient to fence off the neck of a natural partial exclosure such as the Cobourg Peninsula than it is to create a perimeter around an inland site) and the nature of the materials used in construction (e.g. sourcing local or recycled building materials is more cost-effective). Overall costs could also be reduced by created secure area exclusions that encompass the ranges of multiple species for which such conservation action is deemed warranted

1.6 Although the results presented herein represent preliminary estimates for the logistics of the "secure area" strategy of cane toad impact mitigation, they do nevertheless provide wildlife managers with some of the key information required to rationally and efficiently allocate time, money and habitat areas to maximise conservation benefits in the face of cane toad encroachments.

2. Introduction

Since its introduction in Australia in 1935 the cane toad has spread throughout much of Queensland, northern New South Wales and the Northern Territory (van Dam et al. 2002). The cane toad is predicted to further increase its range, primarily throughout coastal and near-coastal regions of tropical Australia, to encompass an area of approximately 2 million km^2 (Sutherst et al. 1995).

The spread of the cane toad into the Northern Territory has been documented reasonably comprehensively since about 1980 (e.g. Freeland and Martin 1985), and some preliminary work done on impacts on frog communities (Freeland and Kerin 1988) but recognition of the imminence of the threat to Kakadu National Park did not stimulate significant public investments in research on impacts or control until the 1990s. A substantial investment was then made to explore options for biological control, but few additional impact studies were done, and those completed were too short and unfocused to produce more than ambiguous results (Catling et al. 1999).

A consequence of this neglect has been that Australia is unprepared to respond to local, regional, national and international concern at the threats presented to highly significant sites like the World Heritage listed Kakadu National Park. There are too few data to permit the most basic cost-benefit analysis of proposals for control, or to demonstrate that no attempt at control is a reasonable option

In common with most other pest control programs, it is probable that effective management of cane toad impacts will require a combination of approaches, including (if feasible and socially acceptable) biological control, conventional methods of destruction, and exclusion. This report addresses the role of exclusion, whether employed as a dominant strategy or in combination with other methods.

Here we provide estimates of minimum viable population sizes (MVPs) and the area required to support those MVPs for 12 candidate species chosen for their probable susceptibility to the invasion of the toad and/or public perceptions of their particular significance. Species of particular interest to Indigenous people are included (Altman et al. 2003). It also describes the physical means of applying exclusion structures and analysis's the potential capital and recurring costs.

3. Objectives

a. Estimate minimum viable population sizes and habitat areas (MVPs) for a range of fauna thought to be at greatest risk from cane toad invasion.

b. Assess MVPs at a range of thresholds for probability of population failure over selected time-frames.

c. Using available empirical data on the home range size and population densities of the target species, determine the area required to support MVPs.

d. Estimate costs of capital and recurring costs of enclosing MVPs in a number of plausible landscape settings and at different levels of risk of population failure.

e. Describe relationships between capital and recurring expenditures and the probability of securing viable populations within toad exclosures.

4. Minimum viable populations – an overview

Predicting the persistence of small populations has become a key issue in ecology and conservation biology. A large and growing number of species are threatened with extinction from human associated factors (habitat loss, over-exploitation, pollution and invasive species such as the cane toad) and stochastic factors (demographic & environmental fluctuations, inbreeding, loss of genetic variation, and natural catastrophes) (World Conservation Monitoring Centre 1992). Empirical studies have shown population size and habitat area to be strong predictors of extinction vulnerability (Terborgh and Winter 1980; Berger 1990). Given that the resources available to conservation programs are finite, and data on many endangered species are inadequate or unavailable, there is a critical need for general rules for predicting minimum reserve size and the minimum viable size of wildlife populations. With this key information, time, money and habitat areas can be rationally and efficiently allocated (Lacy 1992). Further, since political and administrative decisions are frequently made without the time or data for detailed, case-specific evaluations (Pressey et al. 1993), general, yet scientifically reliable, estimates of minimum viable population sizes and habitat areas (MVPs) are essential. An MVP can be broadly defined as the smallest size required for a population or species to have a predetermined probability of persistence for a given length of time, given real-world constraints (see reviews by Simberloff 1988; Nunney & Campbell 1993).

The MVP approach is already widely applied as a heuristic decision-making tool. For example, the concept is employed (consciously or sub-consciously) each time a land-use planning decision sets aside habitat to conserve a species. The area set aside translates to an estimated population size considered to be sufficient to persist for some unspecified period. Similarly, the World Conservation Union (IUCN) specifies population size cut-offs for all taxa in its rules for categorising endangerment under the Red List criteria (e.g. a "lower risk" population has a greater than 90% probability of survival over 100 years; IUCN 2000). The conservation of wild nature often requires that decisions be made immediately, and without the benefit of complete information. The options in decision-making are to use the best, albeit imperfect, scientific information and tools available given the data to hand (e.g. MVP estimates), or for humans to make subjective decisions that are notoriously inaccurate, or wait for the collection of copious and more precise data while species go extinct.

Various attempts have been made to estimate MVP. Being primarily theoretical, these approximations of MVP have been based on genetics (Franklin 1980; Lande and Barrowclough 1987), demography (Lande 1988; Menges 1992), environmental stochasticity (Shaffer 1981; Lande 1988), and all factors combined (Soule 1987). These estimates imply MVPs ranging from a few hundred to many thousands of individuals, with considerable variance likely to be found among taxa and species (Shaffer 1987). The few empirical estimates made have been based on observed extinction rates of mammals in National Parks in North America and of boreal mammals on mountain tops in the United States (Belovsky 1987). Thomas (1990) compiled data from island biogeography and other sources, suggesting a median number of 5,500, but concluded that MVPs would likely be an order of magnitude greater in species with high variation in population sizes, such as small mammals and insects.

5. Procedures used to estimate MVPs

It is not possible to do deliberate field experiments to estimate MVP on a range of taxa as is required for a study such as the present one. A feasible and flexible approach is to use population viability analysis (PVA) procedures to estimate MVP for a wide range of species (Shaffer 1981; Burgman et al. 1993). Conceptually, MVP and PVA are closely linked (Ewens et al. 1987). PVA is a means for predicting the probability of extinction by using life history information to build a model of a species and its environment, and projecting the population's fate using stochastic computer simulation (Gilpin and Soule 1986; Boyce 1992). PVA implicitly or explicitly models the synergism between stochastic factors (necessary to avoid underestimating MVP), and permits large numbers of MVPs to be estimated in a reasonable time-span. Further, the tools of PVA have been validated using retrospective tests of well studied systems and shown to be unbiased (Brook et al. 2000).

There is no generally agreed upon definition of over what time frame population persistence should be measured, nor what extinction probabilities should be used. We therefore evaluated a range of risk levels and time-frames, ranging from 1 to 50% probability of extinction over 20 to 1000 years. However, a standard definition of a <10% probability of extinction over 100 years was used unless otherwise specified. Since the practical imperative was to define a minimum habitat area for each species, we specifically estimated the carrying capacity (equilibrium population size, given density dependent effects) required to deliver a MVP that satisfies the relevant risk-time definition (see Beissinger and Westphal 1998). The MVPs here reported are expressed as in terms of total population size (males and females).

The underlying statistical models upon which the inferences presented in this report are based on PVA modelling and associated statistical analysis of a large compilation of well-studied, long-monitored species from across the globe with good information on population dynamics (Brook et al., manuscript in preparation). These 1198 species spanned a wide range of taxa, biomes and life histories. Population dynamics time-series data were obtained from various online, text and primary sources. A major reference source was the Global Population Dynamics Database (cpbnts1.bio.ic.ac.uk/gpdd/), which provides time-series data for nearly 5,000 populations spanning over 1400 species. Other sources were used where the data was either superior to that of the GPDD, or where time-series data were unavailable from the GPDD. However, due to ambiguities and inconsistencies within the GPDD, as well as inherent differences among the many data sources, a strict set of filtering criteria were subsequently derived, permitting the objective removal of time-series data deemed unworthy or inconsistent. This allowed for the establishment of a reduced and coherent database which was suitable for cross-species analysis.

Information-theoretic model selection procedures were used to assign relative weights (strength of evidence) to an *a priori* candidate set of five population dynamics time-series models fitted to the long-term monitoring data of each species and based upon variants of the generalized population dynamics model:

$$\log\left(\frac{N_{t+1}}{N_t}\right) = a\left(1-\frac{N_t}{K}\right)^{\theta} + \varepsilon$$

Depending on the values of a, K, and θ (as estimated from the population time-series data), the stochastic population model can represent a random walk, exponential growth or density dependent growth/limitation at varying levels of intensity (Dennis and Taper 1994). Model parameterisation and time-series data analysis was conducted using @RISK (Palisade Corporation 2000), a stochastic simulation "add-in" for Microsoft Excel®. The model variants used for this study were as follows:

- Random walk (RW) 1p (σ)
- Exponential (EX), 2p (a, σ)
- Logistic (LG), 3p (a, K, σ)
- Gompertz (GZ), 3p (a, ln[K], σ)
- θ-Logistic (TL), 4p (a, K, σ, θ)

MVPs were generated using each of the above models by conducting a series of runs with different initial population sizes and carrying capacities (e.g. 50, 100, 500, 1,000, and 5,000). When these did not encompass the threshold MVP probabilities of population survival for the required definition of MVP, higher or lower starting values were added until the desired thresholds were attained. Extinction risk was regressed against log *K* (to linearise the results), and the predicted MVP then interpolated. Subsequently, runs around this predicted value were performed to refine the MVP estimate, until the results were within 1% of the required probability. 100 simulation replicates of each run were used initially, to keep computer time to reasonable levels, and for the final assessments, 1,000 replicate simulations are used to provide greater precision. The final, model-averaged MVP estimates were calculated by scaling each individual model prediction by the model's AIC_c weight estimated during the maximum likelihood fitting procedure for each species (refer to Burnham and Anderson 2002, for methodological details).

The results of the model-averaged simulations were then used to derive a statistical approximation relating the estimated MVP for all of the 1198 species to the ecological correlates described in sections 7 and 8. To do this, multivariate generalized linear mixed models (GLMM) were fit using the *R* statistical package v1.8.1 (Ihaka and Gentleman 1996), specifying a normal error distribution with an identity link function, where log(model-averaged MVP) was the response variable. The error structure of GLMM corrects for non-independent of statistical units (species), in this case due to phylogenetic relatedness, and permits the 'random effects' variance explained at different levels of hierarchical clustering (Class/Order/Family) to be decomposed. The seven derived predictor variables were modelled as 'fixed effects'. This procedure was repeated for a range of risk levels and simulation duration to provide estimates across a range of different MVP definitions.

The final GLMM models so derived were used to determine the MVPs for the candidate taxa evaluated in this report, after arriving at adequate estimates of the six composite predictor variables described on p. 6-7. Minimum habitat areas (MHAs) were determined as the product of the estimated MVP for a given species and the habitat area required per individual (based on known or inferred home range size or average density, listed later in Table 2). For a worked example, consider the northern quoll, which has an average home range size for females of 2.3 ha (Schmitt et al. 1989). The estimated MVP for this species for a <10% risk of extinction over 100 years was 19 100 individuals. The minimum habitat area, assuming overlap of male and female home ranges, would be determined as follows:

> MHA = 19100 x 2.3 x 0.5 [sex ratio] = 21 965 hectares = 220 km^2, or a fenced area of roughly 14.8 km x 14.8 km.

There are three fundamental assumptions associated with the approach used. 1) No habitat loss (since the concern is with the minimum habitat area to be maintained over a given time frame). 2) Current threats and life history parameters do not change in the future (e.g. human impacts do not get worse). 3) Individual populations are discrete and isolated (not distributed in a source-sink or metapopulation configuration). However, these assumptions are stringent only as concerns the MVP estimate for that particular population.

6. Ecological correlates

Twenty-four morphological, life history, ecological and behavioral attributes that have been shown or postulated to correlate with extinction risk were collected for a suite of 1198 species (Brook et al. in preparation).

C1 Body weight: average adult weight (male and female) measured in grams.

C2 Body length: average adult length (male and female) measured in millimetres. Body length or size was defined as tip-of-beak to tip-of-tail for birds, tip-of-snout to vent for reptiles, tip-of-nose to tip-of-tail for mammals.

C3 Reproductive type: 1) sexual, 2) asexual, 3) hermaphroditic.

C4 Age at sexual maturity: average age at which individual's first mate (female), given in months. Note that all of the species considered in this report reproduce sexually.

C5 Lifespan: maximum age attained by individuals in the wild, measured in months.

C6 Generation length: average age of breeding adults at the time their young are born, in months.

C7 Social grouping: taken to be grouping of breeding adults. Categorised as 1) solitary (single parent), 2) monogamous pair (where young expelled once mature), 3) small family group (includes minor polygamy, polyandry), 4) gregarious (territorial mammals with harems, promiscuous species.), 5) colonial (large breeding colonies in birds, breeding ponds - frogs, non-guarding fish).

C8 Dispersal ability: Categorised as 1) immediate (<1 km), 2) local (up to 10 km), 3) landscape (up to 100 km), 4) regional (up to 1000 km), 5) continental/trans-oceanic (1000 - 10 000 km), 6) greater (10 000km +).

C9 Disturbance type: Direct loss (culling etc) or indirect loss (pollution, competition with weeds etc.) were scored by either 1 where experienced otherwise 0.

C10 Fragmentation (range decline): Scores for range decline were given according to the extent of loss, thus: 1) species occupies < 1% of former range or almost all habitat unsuitable, 2) species occupies 1-10% of former range or suitable habitat, 3) species occupies 10-50% of former range area, 4) species occupies 50-100% of former range or decline unknown/thought to be small.

C11 Geographic distribution: Score criteria were: 1) very narrow endemic - < 50 square km's or 20km (linear), 2) narrow endemic - < 500 square km's or 100 km (linear), 3) confined to single biome (see notes for definition), 4) regional, 5) continental and greater (transoceanic & migratory).

C12 Population size: categorical estimation of effective adult population at time of study. Categories were: 1) <50, 2) 50-500 (or unknown and thought to be small), 3) 500-5000, 4) 5000-50 000, 5) 50 000-500 000, 6) >500 000

C13 Fertility: number of eggs laid or young born per female, per annum.

C14 Population trend: trend at time of study, given as 1) increasing, 2) stable, 3) declining.

C15 Trophic level: 1) primary producer, 2) detritivore, 3) herbivore, 4) omnivore, 5) carnivore.

C16 Niche breadth: 1) specialist or 2) generalist.

C17 Relationship with Homo sapiens*:* scored as 1) positive - benefit from human disturbance or have been successfully introduced out of native range, 2) negative - no benefit gained by species.

C18 IUCN listing: Following IUCN, categories were extinct (EX), extinct in the wild (EW), critically endangered (CR), endangered (EN), vulnerable (VU), near threatened (NT), least concern (LC), data deficient (DD) or not evaluated (NE).

C19 Legal protection: categorised according to whether a species was either 1) protected in native region, or listed under CITES Appendices II or I or 2) unprotected.

C20-24 Biome: Temperate (includes coniferous forest, mixed hardwood-conifers, temperate deciduous forest, montane forest, Mediterranean shrubland or chaparral and eucalyptus woodland), Tropical (tropical savanna or thorn forest and tropical forest), Arid (Tundra, semi-desert, desert and grassland), Freshwater (lakes, rivers, wetlands and swamp regions) and Marine (shoreline, pelagic and benthic).

Home range: Average home range size.

Seven composite predictors were derived from the 24 attributes listed above:

P1, Biome (5 level factor): An additive score index was used to assign a species to one of the five biomes (temperate, tropical, arid, freshwater and marine). For example, a species whose geographic range extended across coniferous forest, mixed hardwood-conifers and temperate deciduous forest (temperate) as well as tropical savanna (tropical), would be assigned to the biome temperate since the score of occurrence here would have been greater than that within tropical.

P2, Conservation status (2 level factor): species were considered to be *threatened* (score = 1) if they scored 1) under legal protection, were listed under the IUCN red list as anything other than *least concern* (excluding *data deficient* and *not evaluated)*, or where the global population numbered less than 500 individuals. Otherwise deemed *lower risk* (score = 0).

P3, Geographic range (continuous predictor 0-1): Geographic distribution scores were assigned categorically (see above) and then converted to a continuous predictor 0-1 (by subtracting the total by minimum score possible and then dividing by the range). A high score indicated assumed narrow distribution.

P4, Human impact (continuous predictor 0-1): This considered the extent of range of habitat loss, as well as direct loss or indirect loss. Human impact = ((range decline (1-3) + direct loss (0-1) + indirect loss (0-1)-1)/4. Final score converted to a range from 0-1 by subtracting minimum possible value from end value and dividing by the range. Range decline was given greater weighting here.

P5, Body size: Measurements were converted using the natural logarithm.

P6, Ecological flexibility (continuous predictor 0-1): Dispersal ability, trophic level and the extent of ecological specialization are taken to be surrogates of 'ecological flexibility', assuming that those species thought to be more 'flexible' than others are better adapted to change. A high score indicated assumed less flexibility. Species at the top of the food web were assumed to be less flexible ecologically than those at the bottom. Ecological specialisation or niche breadth considered synthetically both feeding specialisation and habitat specialisation. The final algorithm here was thus (dispersal ability (1-6)/6) + (trophic level (1-4)/4) + (specialisation (1-2)/2) – (1/6 + 1/4 + 1/2) / (3-(1/6 + 1/4 + 1/2)).

P7, Demographics (continuous predictor 0-1): Considered as the reproductive life history of a species i.e. age at sexual maturity, fertility, reproductive strategy and longevity. Highly fecund, short-lived species are assumed to be more resilient than long-lived species with extended gestation periods, at least in response to short term and major change related to human impacts. Generation length was not used owing to inadequate data. These parameters were categorised and additive values allowed for a final score (high score indicated assumed high demographic risk). Algorithm used was (fertility + longevity + sexual maturity + reproductive strategy)-4/(11-4).

7. Rationale for selection of candidate species

The set of candidate species evaluated in this report were selected on the basis of the following considerations:

(a) Identified or suspected vulnerability to cane toad impacts (van Dam et al. 2002);

(b) To represent a broad taxonomic spectrum of vertebrates;

(c) To capture a variety of life history types;

(d) To encompass iconic north Australian species;

(e) Likelihood of sufficient demographic and environmental preference data to estimate minimum viable population size and minimum habitat areas using the indirect inference methods developed by one of this report's authors (Brook).

Twelve candidate species were chosen, being the dingo *Canis lupus dingo*, northern quoll *Dasyurus hallucatus* (mammals), black-necked stork or Jabiru *Epippiorhynchus asiaticus*, blue-winged kookaburra *Dacelo leachii*, wedge-tailed eagle *Aquila audax*, black bittern *Ixobrychus flavicollis*, Australian bustard *Ardeotis australis* (birds), black-headed python *Aspidites melanocephalus*, northern death adder *Acanthophis praelongus*, northern sand goanna *Varanus panoptes*, mangrove monitor *Varanus indicus*, and frill-necked lizard *Chlamydosaurus kingii* (reptiles). A detailed description of each species ecology and life history are given later in the report in the appendix.

8. Summary data for the 12 candidate species

TaxGrp	Mam	Mam	Bir	Bir	Bir	Bir
Class	Mammalia	Mammalia	Aves	Aves	Aves	Aves
Order	Carnivora	Dasyuromorphia	Ciconiiformes	Coraciiformes	Accipitriformes	Ciconiiformes
Family	Canidae	Dasyuridae	Ciconiidae	Alcedinidae	Accipitridae	Ardeidae
Genus	*Canis*	*Dasyurus*	*Ephippiorhynchus*	*Dacelo*	*Aquila*	*Ixobrychus*
Species	*Lupus*	*Hallucatus*	*asiaticus*	*Leachii*	*audax*	*flavicollis*
Common name	Dingo	Northern Quoll	Black necked stork	Blue winged Kookaburra	Wedge tailed eagle	Black bittern
Mean mass g	16000	660	6000	300	3626	360
Length mm	1230	500	1150	410	925	630
Wspan mm			2000	720	2100	800
Repdive type	1	1	1	1	1	1
Min age months	24	11	36	12	72	24
Longevity months	108	36	240	132	480	60
Repro've grouping	3	1	2	3	2	2
Dispersal ability	3	1	2	2	4	4
Habitat loss	2	1	1	1	1	1
Direct loss	3	2	2		2	3
Indirect	1	3				
Pollution						2
Severity	2	1	2	3	2	2
Range decline	4	3	4	4	4	4
Population size	4	4	4	5	5	4
Distribution	5	4	5	5	5	5
Density				4		
Fecundity	5	6	3	2.5	2	4
Pop trend	2	3	2	2	3	2
Trophic level	5	5	5	5	5	5
Specialization	2	2	2	2	2	2
Relation to HS	2	2	2	2	2	2
Status	LC	NT	LC	LC	LC	LC
Legal protection	1	1	1	1	1	1
Eucalypt	1	2		1	1	
Trop forest	3					
Savanna	2	1		3	2	
Semidesert	4	3			3	
Desert	5				4	
Lakes/ponds			2			3
River			3			1
Marsh			1	2		2
Shoreline			4			4

TaxGrp	Bir	Ram	Ram	Ram	Ram	Ram
Class	Aves	Reptilis	Reptilis	Reptilis	Reptilis	Reptilis
Order	Gruiformes	Squamata	Squamata	Squamata	Squamata	Squamata
Family	Otidae	Pythonidae	Elapidae	Varanidae	Varanidae	Agamidae
Genus	Ardeotis	Aspidites	Acanthophis	Varanus	Varanus	Chlamydosaurus
Species	australis	melanocephalus	praelongus	panoptes	indicus	kingii
Common name	Australian Bustard	Black headed python	Northern Death Adder	Northern Sand Goanna	Mangrove monitor	Frill necked lizard
Mean mass g	4950	2000	301	2200	1100	635
Length mm	950	1600	600	460	421	230
Repdive type	1	1	1	1	1	1
Min age months	48	36	15	48	12	18
Longevity months	120	240	108	180	180	72
Repro've grouping	1	1	1	1	1	1
Dispersal ability	4	2	2	2	1	2
Habitat loss	1		1			1
Direct loss	2	1		1	1	2
Indirect			2			
Pollution	3					
Severity	1	3	2	3	3	2

Range decline	4	4	4	4	4	4
Population size	5	4	5	6	6	6
Distribution	5	4	5	4	5	5
Density					500	100
Fecundity	1.5	12	23	11	3	11
Pop trend	3	2	3	2	2	2
Trophic level	4	5	5	5	5	5
Specialization	2	2	2	2	2	2
Relation to HS	2	2	2	2	2	2
Status	NT	LC	LC	LC	LC	LC
Legal protection	1	1	1	1	1	1
Eucalypt		1	3			2
Trop forest		4				
Savanna	1	2	2	2		1
Semidesert	2	3	1	3		3
Shoreline				1	1	

9. Estimates of MVPs and MHAs

Estimates of minimum viable population size are very scale dependent, and tend to increase approximately linearly with projection time (number of years a population must remain viable), but non-linearly as the definition of the acceptable risk level (probability of extinction) becomes more conservative. The MVP-Risk-Time surface is illustrated in Figure 1 for the northern sand goanna, *Varanus panoptes*, and very similar relationships were evident for the other 11 candidate species (Table 1). The general result is that larger population sizes will reduce the likelihood of extinction, especially if the absolute risk is quite low, and larger population sizes are also required to buffer against long-term population fluctuations, where chance events such as a succession of poor years may cause extinction even when "average" conditions are not expected to drive an overall population decline (see Shaffer 1981, for a sobering example of this, the Heath Hen).

The results of the MVP evaluations for a range of risk/time definitions for the 12 candidate species are presented in Table 1, ranked from lowest to highest MVP. The differences amongst the different species in their MVP tend to arise because of contrasting modes of reproduction and patterns of survival (e.g., a strategy of producing many offspring with low survival rates [reptiles] versus few, well nurtured offspring [mammals]), generation length, body size, environmental variability etc. (Gilpin and Soule 1986; Reed et al. 2003). MVPs vary by almost an order of magnitude across the 12 species, with the general result being that we should be thinking in terms of several thousands of individuals if our goal to maintain viable populations of these vertebrates.

Two of the definitions used in Table 1 have been applied in Table 2 to address the more practical management question of how much habitat area would be required to support these MVPs. The definition of a 20% risk of extinction over 20 years and a 10% risk over 100 years encompasses short- and long-term perspectives on viability, and are of practical conservation relevance because they represent the risk-time thresholds used to define a species on the borderline between IUCN's (2000) (IUCN) definition of *Endangered* and *Vulnerable* (20% in 20 yr), and *Vulnerable* and *Lower Risk* [not threatened] (10% in 10 yr).

Species with small home range requirements or high average densities tend to require relatively small areas to maintain viable populations. For example, a viable frill-neck lizard (*Chlamydosaurus kingii*) population would require and area of 15.5 km^2, equivalent to a square enclosure with a boundary fence of only 3.9 km length on each side (see Table 2),

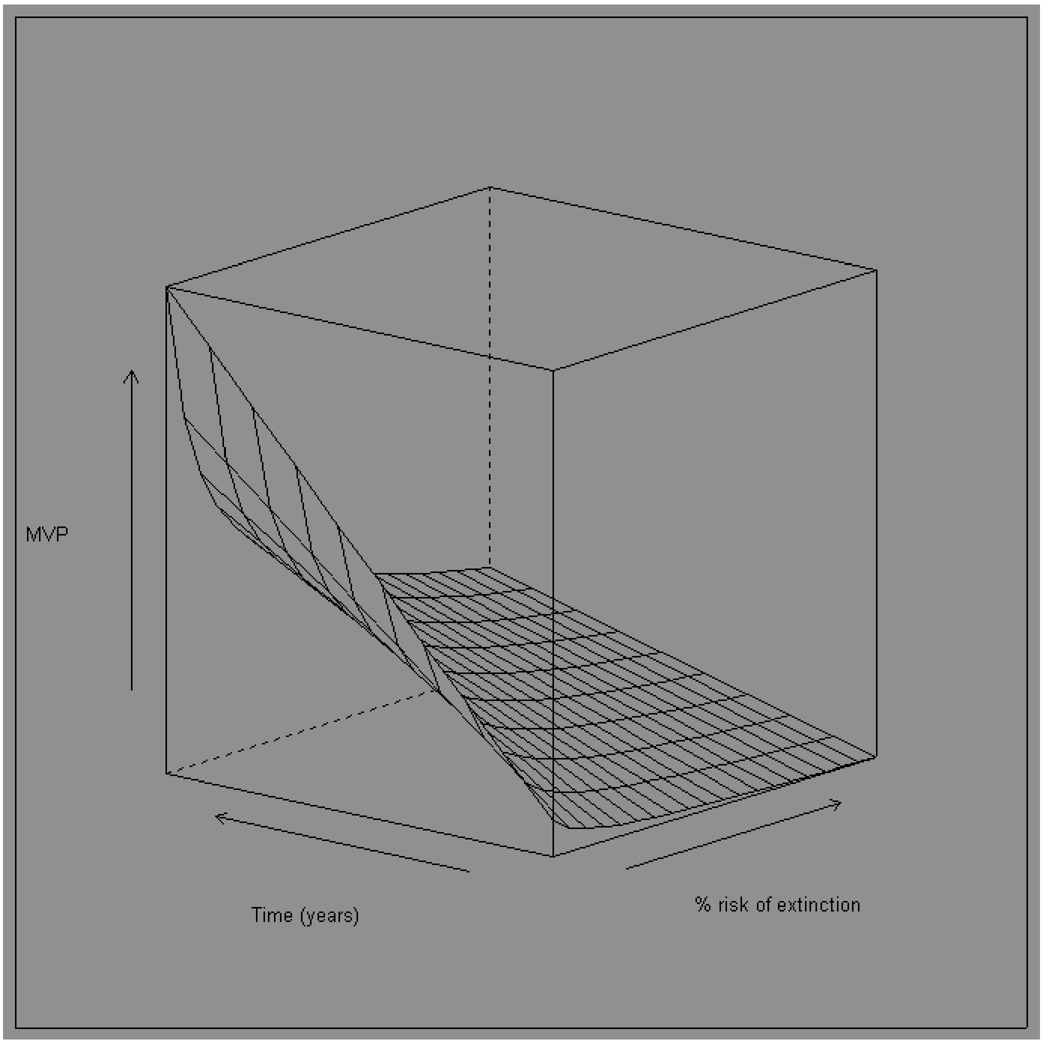

Figure 1. Relationship between MVP, risk level (expressed as the probability of extinction, ranging from 0 to 20 %) and projection time (ranging from 1 to 100 years) for the northern sand goanna, *Varanus panoptes*. MVP ranges here from 150 – 12 150 individuals.

Table 1. Estimates of minimum viable population size (MVP) for 12 candidate species which are suspected to be vulnerable to cane toad impacts, ranked from smallest to largest, using five different risk/time definitions (e.g. 10%/100 yr is the population size required for a less than 10% risk of extinction over a 100 year period).

Species	20%/20yr	10%/100yr	1%/50yr	1%/100yr	1%/1000yr
Wedge tailed eagle	500	2,200	4,700	13,500	193,800
Dingo	500	2,300	4,800	13,700	196,600
Black necked stork	700	3,200	6,900	19,600	281,300
Australian Bustard	800	3,600	7,700	21,800	313,800
Northern Sand Goanna	900	4,000	8,500	24,300	349,800
Black headed python	1,100	5,000	10,600	30,100	432,800
Black bittern	1,200	5,400	11,500	32,700	470,200
Mangrove monitor	1,200	5,400	11,500	32,700	470,100
Blue winged Kookaburra	1,400	6,200	13,200	37,500	539,300
Frill necked lizard	1,400	6,400	13,600	38,600	555,400
Northern Death Adder	1,700	7,500	15,900	45,500	653,200
Northern Quoll	4,300	19,100	40,600	115,600	1,661,800

based on an MVP of 6,400 individuals (Table 1) and a home range size of 0.7 ha.

Conversely, wide ranging and sparsely distributed species (e.g. top avian predators such as the wedge-tailed eagle) have huge area requirements that could not be feasibly enclosed by any boundary exclusion (Table 2). Moreover, from a practical standpoint, species with very high dispersal capabilities via flight are unlikely to confine their movements to enclosures. In this case, it is more a question of providing sufficient natural prey, such as small mammals and reptiles, in toad free areas.

To provide some perspective of scale to the habitat area requirements cited in Table 2, the entire extent of Kakadu National Park is 19,804 km^2, and the Cobourg Peninsula is 2 207 km^2 (Garig Gunak Barlu National Park), which could be conceivably isolated from cane toads by means of a relatively short boundary fence along its narrowest point of connection to Arnhem Land – see next section). Thus the relatively cost effective fencing of the Cobourg would likely support viable population of a host of small mammal and reptile species (and presumably most amphibians and insects, which have considerably smaller home range requirements again), but would fail to capture fully the areas for some of the largest free-ranging species. Worryingly, however, recent field observations suggest that toads may have already penetrated the south-eastern fringes of Garig (information related to Dr. Donald Franklin by one of the Park's senior rangers, John Williams, 30 September 2004), suggesting that it may be too late to isolate this particular region.

Table 2. Minimum habitat area (MHA in km^2) estimates for the 12 candidate species which are suspected to be vulnerable to cane toad impacts and using two different risk/time definitions. MHA was calculated using the MVP estimates given in Table 1 multiplied by the home range estimate (HR) for the species. Perim = length (km) of the total perimeter of a rectangular fenced enclosure required to encompass the MHA for each species.

		20% in 20 yr		10% in 100 yr	
Species	**HR km^2**	**MHA**	**Perim**	**MHA**	**Perim**
Wedge tailed eagle	35	8,750	374	38,500	785
Dingo	39	9,750	395	44,850	847
Black necked stork	10	3,500	237	16,000	506
Australian Bustard	25	10,000	400	45,000	849
Northern Sand Goanna*	0.143	43	26	191	55
Black headed python	0.32	176	53	800	113
Black bittern	5	3,000	219	13,500	465
Mangrove monitor*	0.009	3.6	7.6	16	16
Blue winged Kookaburra	0.4	280	67	1,240	141
Frill necked lizard*	0.007	3.3	7.2	15	16
Northern Death Adder	0.25	213	58	938	123
Northern Quoll	0.023	50	28	220	59

* 2:1 female to male sex ratio in a given territory is assumed

10. Exclusion as a conservation strategy

Exclusion of threatening processes from areas of habitat for species of special concern is a well-established and critical conservation tool. The general approach has been applied to a range of influences, including endemic or exotic zoonotic or other animal disease, exclusion of fire, management of legal or illegal harvest, or invasion of exotic animals and plants additional to the cane toad. Methods can be equally diverse, encompassing destruction of the hosts or vectors of disease agents, removal of habitat features critical for invasive organisms or fostering other conditions that make habitat less favourable, direct killing of invaders, or biological control through predators or parasites.

In the case of the cane toad, densities obtained may be so high and distribution sufficiently wide as to make direct killing problematic (PJ Whitehead, unpublished data). Biological control is also challenging and is at least a decade away from proof of a system that has a reasonable chance of being accepted by a sceptical public (A. Robinson, pers. comm.). Habitat modification to reduce suitability for toads might include such steps as maintaining or encouraging the development of dense ground cover in riparian fringes that interfere with the movement and other behaviours of toads (e.g. Freeland and Kerin 1991). Unfortunately, the dominant land use (grazing) is associated with disturbance of the ground layer. Regular removal of low vegetation by fire over much of the landscape, including riparian fringes (Russell-Smith et al. 2003) is also at odds with maintenance of densely vegetated habitats.

However, exclusion has served Australia well for millennia. The physical isolation of the great southern land favoured evolution of unique and mega-diverse assemblages of flora and fauna. Settlers learned some harsh lessons about the potential impacts of introducing animals like the rabbit, and subsequently bolstered our natural defences against invasion with rigorous quarantine systems that have mostly served the agricultural community well.

Within the nation, exclusion of “pests” has been practised on a physical scale rarely seen. The dingo fence, constructed in the 19th Century, stretches for more than 5000 km, and was designed to protect the eastern Australian grazing lands from the wild dogs of the interior, which were probably introduced to Australia by Aboriginal people, thousands of years before settlement. The extraordinary and often successful efforts made by the Western Australian Government to exclude agricultural pests like starlings *Turdus turdus* and house sparrows *Passer domesticus* by relying principally on the natural barriers of the Nullabor Plain and arid interior supplemented by ruthless destruction of intruders, are now the stuff of Australian legend.

Given this history, it is perhaps worth reflecting on the reasons for the much more passive stance in regard to the creeping invasion of the toad. First, the species does not damage agriculture or otherwise significantly threaten the mainstream economy. Impacts on economic interests are mostly confined to the Aboriginal customary economy, which depends heavily on wildlife threatened by toads for high quality food (Altman et al. 2003). Maintenance of the customary economy has never been treated as a serious issue by wildlife or pest management authorities. Second, the invasion of the Australian mainland has been gradual. During the long period of range expansion, no more than anecdotes were gathered about its effects on conservation values (see Freeland 1984; Burnett 1997). Animals affected by the presence of toads apparently declined abruptly in abundance but then mostly recovered to varying degrees. No species was shown to have become extinct due to toads during a period when extinctions of arid zone mammals were rife (Morton 1990), so there was no conservation imperative to halt the invasion, even were it considered possible to intervene effectively. Finally, although toads were seen as an inconvenience, such as a threat to the health of pets (Freeland 1984), the inconvenience caused to non-Indigenous Australians was too minor to warrant large public investments in control.

However, the fact that this report, among others, was commissioned is testament to increased public discomfort when it became obvious that Australia had done little or nothing to protect the values of the World Heritage Kakadu National Park from the effects of toads. Their intrusion threatened both the natural and cultural heritage values for which the park had been listed. The work stimulated by that recognition has provided the first rigorous quantitative demonstration of the impact of toads on Australian fauna (M. Oakwood, unpublished; D. Holland unpublished). That work has confirmed the severity of the initial increases on mortality of northern quolls and some goannas in the presence of toads, but suggested no plausible responses.

11. Options for Toad Exclusion

During 2003, the Sessional Committee of the Environment and Sustainable Development of the Northern Territory Legislative Assembly inquired into issues associated with the entry of cane toads into the Northern Territory. A number of submissions to the Committee dealt with the issue of exclusion. Proposals and related argument covered two very different scales.

First, there was material on steps that householders could take to keep their yards toad free. As these sorts of measures are unlikely to contribute significantly to the protection of viable populations of the native fauna known to be at greatest risk from toads, they are not considered further here.

Second, there were proposals for erecting a barrier to exclude toads from Cobourg Peninsula, the site of Garig Banuk Barlu National Park. This proposal was supported by the Garig Board, and was under active consideration by the Parks and Wildlife Service (PWS) of the Northern Territory. As a consequence some work was done by the PWS and the Department of Infrastructure, Planning and Environment to explore the feasibility of such a barrier. This report draws on that material to explore a wider range of options.

The treatment is based fundamentally on recognition that principal determinants of the nature and scale of the response to cane toads will be assessment of the costs, likely effectiveness and putative benefits of such barriers over the long term. Our goal is therefore to provide realistic comparisons of capital and recurring costs of barriers protecting areas large enough to contain populations of vulnerable fauna large enough to be viable over the long term.

Sites for cane toad exclusion

We consider that to illustrate the implications of our data in a heuristically useful way, it is important to provide context and relate the results to genuine proposals or options that provide compelling illustrations of particular aspects of the problem. We have therefore chosen to relate our estimates of MVP and areas of habitat needed to sustain those populations to:

(1) The construction of a cane toad barrier across the neck of the Cobourg Peninsula designed to exclude toads from Garig Banuk Barlu National Park, the first site listed under the Ramsar Convention on Wetlands of International Importance.

(2) Quarantining of islands, (excluding the Tiwi Islands, which have already been invaded by toads), including islands used to establish populations of northern quolls using wild stock taken from Kakadu National Park and closer to Darwin.

(3) The construction of barriers on the mainland to enclose areas of habitat favourable for one or more of the species considered here, including consideration of options based on one large area or a number of smaller sites.

Design of cane toad barriers

Estimates of costs of barriers capable of excluding toads are based on designs made by the Department of Infrastructure, Planning and Environment in Darwin (Lyle Campbell, personal communication). The proposed barriers are to be constructed of sheets of compressed fibre panel, 12 mm thick and 1.2 m wide, linked by metal angle and capped with metal flashing. The panels are to be placed in 30 cm deep trenches, refilled with rammed earth or concrete. The panels would thus stand 90 cm above the substrate, high enough to prevent adult toads jumping over them, with their surface being sufficiently smooth to prevent climbing. The panels are likely to be resistant to minor impacts, but will crack or shatter under impacts from larger falling branches or trees, vehicles or large feral animals like buffalo or horses.

Animals that dig deeply, like feral pigs, or burrow like a range of native species (goannas, small mammals) may undermine or tunnel under panels. Toads may use or enlarge such excavations. As a consequence, barriers will require regular and relatively close, fine-scale monitoring to maintain their integrity. Estimates have been made of the cost of such monitoring based on stated assumptions.

Cost estimates for all structures were based on use of new materials and full commercial costs for fabrication and erection. In order to expand the range of plausible options considered, we also provide estimates assuming that costs could be halved by use of second hand materials and some voluntary labour. For exclusion options such as small islands that may appear to be too small to maintain vulnerable fauna over the long term, we have also provided some preliminary estimates of the cost of maintaining separate populations of relevant fauna in captive breeding colonies, which might be used to supplement island populations as required. The range of variables considered is summarised in Table 3.

Table 3: Variables considered in estimates of the costs of cane toad exclusion.

Variable	Issue	Sources of variation in cost
Construction of barriers	To limit probability of incursion and the number of toads gaining access to sites warranting protection	Materials and construction methods Durability Human and vehicle access (gates) Drainage lines and hence additional constructions costs and increased risk of failure Maintenance of associated fire and treefall breaks Interest rate on capital requirements
Maintenance of structures	To minimise periods of vulnerability through failure of barriers	Regular clearing of firebreaks Frequency of inspection Range of sources of damage, including accident (vehicles), tree-fall, feral animals, erosion, flood, other washout
Surveillance	To detect incursions quickly To demonstrate that exclosure is effective in terms of species requiring protection	Frequency of inspection of habitats favourable to toads Total area and range of habitats subject to inspection Design of surveys (precision) required to detect change in abundance of vulnerable fauna
Response to incursions	To respond effectively to real incursions and to false alarms	Intensity and duration of response Spatial extent of response Response measures
Maintaining captive populations	To reduce risk in event of catastrophic failure (e.g. cyclone) and to support use of smaller than optimal sites (especially islands)	Size of captive populations Number of captive populations

12. Key assumptions underpinning cost estimates

In the absence of reliable information about cane toad impacts on our candidate native species, generating estimates of area required and resultant costs of exclusion require a number of key assumptions. The most important are:

(1) habitats capable of sustaining fauna vary markedly in quality and hence the densities they can support but, for most species, the prospects of locating large tracts of uniformly optimal habitat are low;

(2) our estimates of minimum habitat area (MHA) are best treated as requirements for habitat of "average" quality (usually containing areas of high quality habitat separated by a matrix of lesser quality and sometimes marginal habitat) and so may be considerably larger than required in optimal habitat;

(3) sustained or intermittent increase in mortality of any level above the "background" embedded in the population viability analysis and estimates of minimum viable population size will result in a probability of extinction above the 10% threshold we have set as acceptable over the time horizon of 100 years;

(4) presence of toads within the target area, in any numbers, at any stage of the life cycle, for any substantial period will result in relevant increases in mortality;

(5) management authorities therefore adopt a "zero tolerance" approach, treating any increase in mortality as unacceptable (in fundamental conflict with the management goal) and so design exclusion and associated surveillance and response regimes to minimise probability of intrusions, discover minor intrusions promptly and eradicate them quickly;

(6) effective exclusion demands a combination of physical barriers to cane toad dispersal and regular monitoring of sites for the presence of toads using methods best suited to the physical location; and

(7) chosen sites lack toads at the time of construction, so no costs of cane toad removal are incurred. This option is rapidly being foreclosed for many of the more bio-diverse regions of the Northern Territory but remains realistic for parts of the NT and Western Australia.

It should be noted that in the case of artificial barriers to movement of toads, movements of many other animals will be inhibited and that this may have undesirable consequences for wildlife enclosed within those barriers. We do not provide estimates of the "cost" of such losses.

13. Results – the cost of exclusion

Estimates of the costs of exclusion are given in Tables 4 and 5.

Cobourg Peninsula

Table 4 shows the elements of the estimates of the total annual cost of excluding toads from the Cobourg Peninsula, and area large enough to support viable populations of most of the fauna we considered. It is important to understand the assumptions underlying those estimates. Important decisions regarding items requiring consideration that have substantial impact on the scale of those estimates are:

(1) Interest rate: An annual rate of 5% was applied to the cost of construction as an estimate of the cost of capital and to acknowledge that such an investment will divert funds from other conservation activity of potentially equivalent or greater benefit. Inclusion of this factor effectively doubled the cost of construction averaged over 15 years.

(2) Lifetime of structures: We have limited information on the life of the materials used for this barrier under the conditions they will experience. We have assumed that damage from tree fall, fire, storm damage, erosion, feral and burrowing animals will be frequent and that rapid repair will see a long term incremental degradation of the barrier that will be better managed by replacement than ongoing and increasing expensive repairs. We have assumed that barriers will last longer when footings of concrete are used and increased the estimated lifetime from 15 to 20 years.

(3) Monitoring: Any barrier capable of excluding toads and placed in the challenging north Australian environment is likely to suffer frequent damage that compromises its effectiveness. Close monitoring of the integrity of the barrier will be essential, and the frequency of examination will determine the costs and prospects of achieving control should intrusions occur following damage. We do not have the information needed to assess the relative costs of different monitoring schedules *versus* the costs of achieving and demonstrating control given various delays between intrusion and detection. It is therefore impossible to determine an optimal schedule and choices become essentially arbitrary. We have specified and based our calculations on twice weekly checks during the wet season - when damage is both more probable and likely to coincide with greater mobility of toads - and weekly in the dry season.

(4) Response: There have been no carefully documented and costed responses to control cane toad intrusions capable of achieving total eradication in a reasonably short period. We have assumed that a team of 10 people working for a minimum of 10 days will be necessary to actually achieve removal of all intruding toads and satisfy both management authorities and public that this result has actually been achieved. We have assumed that the probability of a real intrusion is quite low (at 10% per annum), but that a need to check reports of intrusions that prove unfounded will be more frequent at 5 event per year, and require 7 days (FTE) of staff time. We have assumed that methods will be based on manual capture plus some trapping and be focused on waterbodies in the region of suspected intrusion.

All of the variables used in calculations are included in annotated spreadsheets that are available from the authors on request.

The cost of constructing the 6 km barrier, averaged over the specified lifetime, is about 60% of the total annual cost. The balance covers maintaining, monitoring and responding to intrusions through. The estimated requirement of about $410,000 pa considerably exceeds the existing routine operational budget of Garig Banuk Barlu National Park, which meets all other conservation objectives. Nonetheless, the cost is a tiny fraction of the expenses involved in enclosing and protecting equivalent or smaller areas that require constructions and maintenance of a complete exclosure (Table 5).

Other terrestrial situations

Costs of exclosures for a range of fauna under a number of different assumptions regarding habitat quality are illustrated in Table 5. The estimates for enclosing an area large enough to maintain the northern quoll (a 59 km perimeter if an approximately square layout is assumed) are up to $3.2 million annually. In calculating this figure, costs similar to the Garig fence have been assumed, but reduced to take account of simpler gates and uniformly favourable terrain (e.g. no coastal margin). A 20% reduction in maintenance costs has been assumed to take account of economies of scale in securing the larger structure. Moreover, and despite the much larger perimeter, no increase in the probability of penetration of the barrier is assumed. We therefore regard the estimate as a conservative one.

We also examined the costs of providing equivalent protection to a quoll population occupying highly favourable habitat such that the area was capable of supporting 4 times the density of female territories found in typical habitat. Because the ratio of perimeter to area is higher for smaller areas (Figure 2), the cost is reduced to about half ($1.78 million pa). The estimate is slightly higher than half because it is assumed that costs of construction in the rocky areas that appear to provide superior quoll habitat will be 20% higher than in less rocky savanna.

We have also examined the costs of a completely artificial exclosure, namely maintenance of captive populations in the equivalent of a wildlife park. We estimate costs at about $510,000 pa for a population of 475 quolls, the number needed to avoid genetic problems (Frankham et al. 2002).

Costs of maintaining medium size lizards are considerably lower (Table 5), but still more expensive than the Garig barrier. Moreover, it is probable that a site selected to protect quolls would also protect a viable population of frill-necked lizards and some monitors.

Costs of enclosing and maintaining viable populations of the 2 snakes we considered are very high (around $6 million pa). It should be recalled that these estimates assume no increase in the risk of cane toad intrusion or costs of eradicating intruding populations. This is probably unrealistic for such large perimeters (exceeding 100 km) and hence we regard the estimates as very conservative especially as we have assumed a 40% reduction in maintenance costs for these longer fences.

Given that the risk of failure of exclosures is presently unspecifiable, we have also considered the costs of "mixed" strategies that include "insurance" through intensively maintained captive populations. Captive populations can also fail – for example through a disease outbreak - but in general will be organised as a number of sub-populations maintained in widely separated locations, so that the risk of complete failure through catastrophe or otherwise is extremely low. Maintaining substantial and hence secure captive populations costs very much less than well-maintained exclosures in remote sites, irrespective of optimistic assumptions of low risk of failure of barriers around "natural" habitats.

Islands

We have limited direct experience of the costs of maintaining islands free of toads because formal and explicitly resourced "quarantine" arrangements have never been implemented. A number of the larger islands with substantial human populations and regular access by boat and air (Groote Eylandt and Bathurst Island) have already been invaded by toads. Other islands of substantial size close to major estuaries have also been invaded, such as the Edward Pellews group adjacent to the MacArthur River.

The estimates we have provided relate to middle-sized islands (see Figure 3), often in very isolated sites, and requiring access by a mix of road travel, boats and less frequently by light aircraft using bush airstrips. They are necessarily approximate because there are no data about frequency of access by land owners (mostly Aboriginal) other users of the coastal region (e.g. commercial fishermen) or recreational access (fishers or pleasure boaters).

We envisaged 9 visits per year by groups who would survey for toads, plus additional work interviewing the region's boat users to assess risks and promote awareness.

We generated an estimate of about $60,000 pa per island. However, there would obviously be scope to reduce this cost by linking visits to neighbouring islands if included in an exclusion exercise. Whilst this cost appears relatively modest compared with construction and maintenance of barriers in terrestrial settings, effectiveness is presently unknown, and most NT islands are too small to maintain viable populations of one or more relevant species over the long term (Figure 3). By definition, islands used for introductions of mainland "stock" to provide protected populations will lack resident populations of the species of concern, and their absence under natural conditions will occur probably because they are too small. This means that an island strategy would probably need to be associated with maintenance of viable captive populations, at a cost (for quolls) of several hundred thousand dollars per annum.

Captive populations

Providing estimates of the cost of maintaining captive breeding populations was not part of the project brief, but given the ambiguity inherent in estimates of the cost of untested systems of exclosure, inspection and response, we thought it useful to provide at least a crude comparison with this more conventional approach. Our estimates are based on modules sufficiently large to maintain groups of 9-12 animals and assume placement within existing wildlife parks or similar facilities. Thus they represent a conservative estimate of total costs, but include provision for staff salaries, food, veterinary care and connection to infrastructure carrying utilities like water and power supply, and drainage. They are best treated as broad indicators of costs rather definitive estimates for particular species, for which costs will be highly context dependent, depending, for example, on availability or otherwise of skills for maintenance of that taxonomic group among existing staff.

Costs for maintaining populations large enough to maintain genetic variation are far from trivial, but are nonetheless cheaper than semi-natural exclosures.

Table 4: Costs of exclusion of cane toads from an area (Garig Gunak Barlu National Park, Cobourg Peninsula) large enough to support populations of the Northern Quoll and other co-occurring vulnerable species with equivalent or lower area needs (northern sand goanna, black headed python, mangrove monitor, blue winged kookaburra, frill neck lizard and northern death adders). For instance, the minimum area of savanna required to support a viable population of quolls and at a probability of persistence of 90% over 100 years, without supplementation from other populations, is 220 km^2 (Table 2), where as the total area of Garig is 2207 km^2. Total costs are calculated over 15 years (for options 1 and 3) or 20 years (option 2). Option 4 is a cheaper structure based partially on second hand materials and using some volunteer labour. An interest rate of 5% is applied to the capital cost over the life of the project and incorporated in estimates of average annual cost.

Situation	**Area (km^2)**	**Item**	**Quantity**	**Description**	**Total cost (of structure)**	**Annual cost**
Peninsula (Garig Gunak Barlu National Park)	2207	Construction of fence Including labour)	6 km	Based on structure of compressed fibre board on steel supports and with metal capping, with 15-20 year life (before total replacement required) and 5% interest rate. Costs include initial clearing of line. Option 1 involves trenching to bury panels to 30 cm and repacking with earth. Option 2 uses concrete footings throughout. Option 3 uses concrete in vulnerable areas and rammed earth over most of length	(1) \$3.45 million (2) \$5.86 million (3) \$3.64 million (4) \$1.81 million	(1) \$229,700 (2) \$292,900 (3) \$242,400 (4) \$121,200
		Construction of gates	2	Double gate and associated structures over main access road to permit vehicle entry but limit toad access, plus gates over separate access track.		

Situation	Area (km^2)	Item	Quantity	Description	Total cost (of structure)	Annual cost
		Tidal zone protection	2	Barrier for tidal zone at both northern and southern margins of the peninsula		
		Maintenance of firebreaks	12 km	Annual maintenance of firebreaks to limit damage by tree fall or fire		$4,800
		Repairs and maintenance of structures	6 km	Repairs to major and minor damage from floods, vehicle damage, feral animal damage and tree fall.		$54,741
		Inspection and surveillance of structures and surrounds	6 km	Regular inspection to promptly detect breaks and mobilise repairs, as well as identify and intervene in potential sources of damage (e.g. developing drainage changes)		$28,141
		Surveys for detection of toads inside barrier, including in the absence of known breaches in barrier	weekly	Inspections of entire fenceline using pitfalls and other traps and inspections of all known waterbodies persisting during dry within 2 km of fenceline. More frequent (twice weekly) inspections during wet season. Also includes ad hoc inspections and interviews with visiting boats, commercial and recreation users of the park		$39,854
		Responses to entry of toads	As necessary	Includes aggressive interventions to control intruding toads, plus comprehensive investigation of all reports. Methods to include hand capture at waterbodies plus trapping.		$28,006

Situation	Area (km^2)	Item	Quantity	Description	Total cost (of structure)	Annual cost
				Assumes probability of significant and well established entry is low, and figures average high cost responses over long periods.		
		Surveys of populations of vulnerable fauna	Annual	Surveys to provide assurance that populations of fauna of concern are actually being maintained		$11,196
		Maintaining captive populations	Ongoing	Maintaining captive populations of relevant provenance as "insurance"	$1.69 million	$510,127 for quolls
TOTAL				without "insurance" with "insurance" for one (most vulnerable) species	~$3.6 million ~$5.70 million	~ $410,000 ~ $920,000

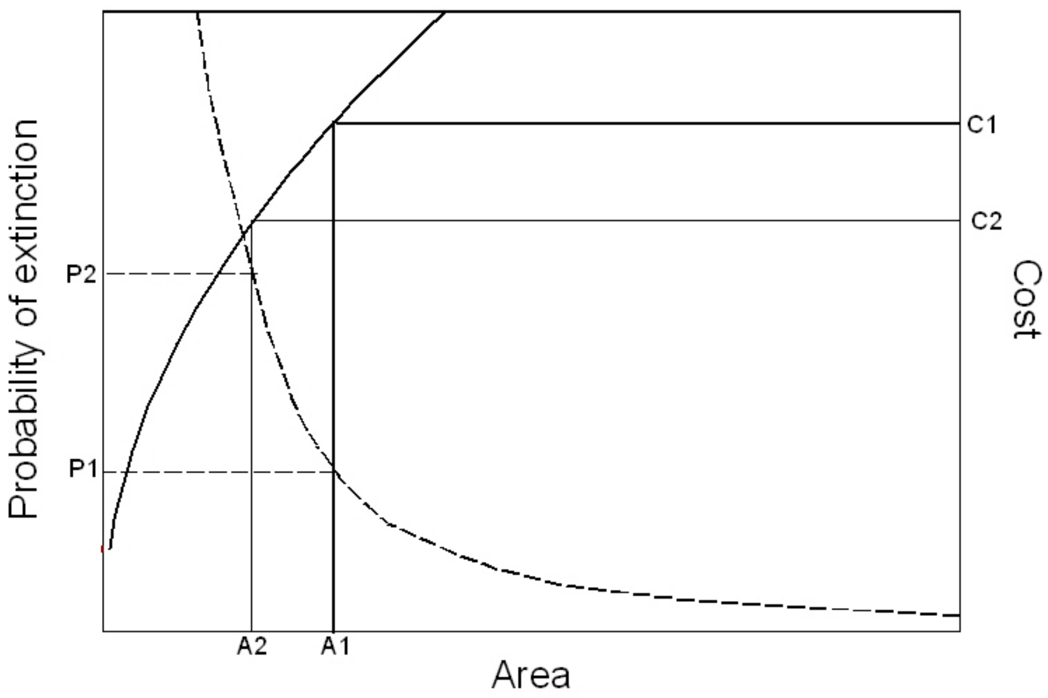

Figure 2. Illustration of the asymmetry of impacts of reduced area (A1-A2) on costs of maintaining a perimeter (continuous heavy line) and probability of extinction (dashed line). Relatively modest reductions in cost (C1-C2) are associated with very substantial increases in risks of extinction (P2-P1). The lines are based on a hypothetical "average" vertebrate with a home range of 5 ha. The probability of extinction axis is from zero to 40%.

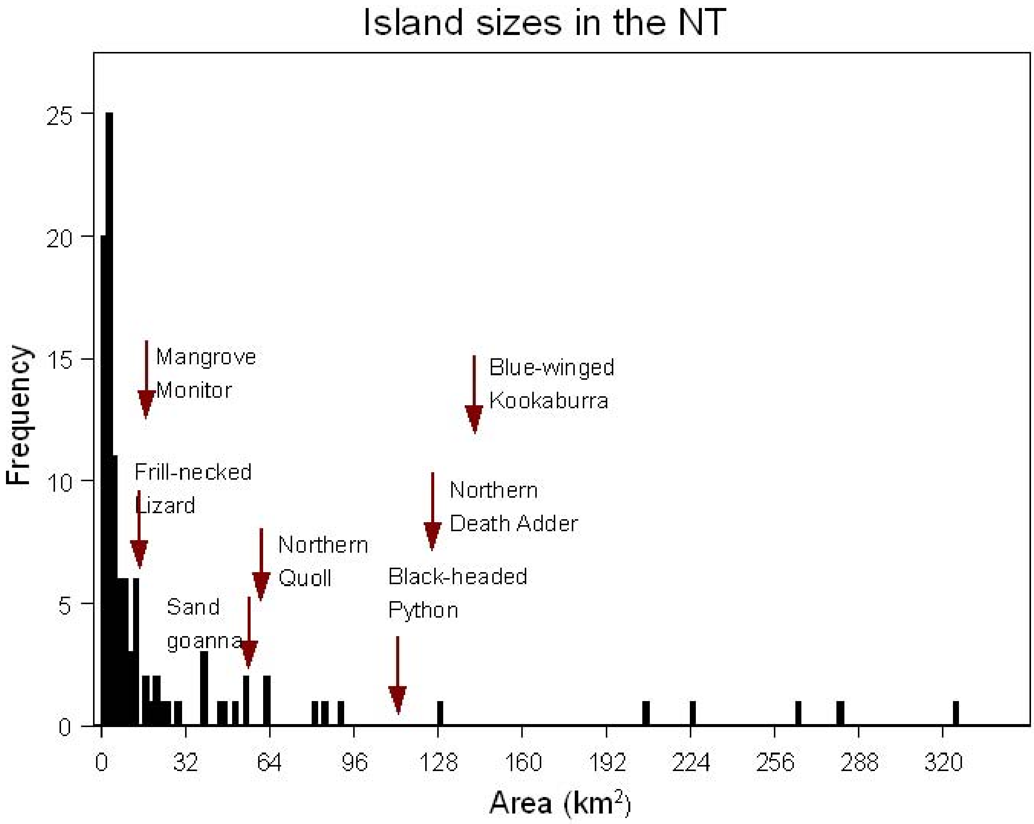

Figure 3. Island sizes in the Northern Territory in relation to MHAs (<20% probability of extinction in a period of 20 years) for a number of fauna. Neither the largest islands (Bathurst = 1707 km^2; Melville = 5821 km^2; Groote Eylandt = 2285 km^2) nor very small islands less than 2 km from the mainland are included. Very small islands (<1 km^2) are also excluded. Clearly there are very few NT islands capable of sustaining a fauna similar to the mainland and few that can sustain viable populations of common vulnerable fauna unless active population management, including supplementation, is undertaken and maintained.

Table 5: Comparison of annual costs for a number of exclusion scenarios for a range of vulnerable species in northern Australia.

Situation	**Vulnerable species**	**Annual costs ($000)**	**Issues**
Peninsula	Northern quoll and varanids, snakes and frill-necked lizards, non-migratory birds	410	Probability of higher rates of toad entry, by both sea and land, than some alternatives. Considerable uncertainty regarding ability to effectively "close" coastal margins of cross-peninsula barrier If exclusion is successful, such sites likely to be large enough to be confident of protecting many species In the example used, costs are contained by access to existing infrastructure and staffing in an existing national park. This will not always be the case, and costs could be considerably higher in other situation.
Non-estuarine island	Northern quoll (introduced)	198	Difficult to institutionalise inspections of boats because of lack of facilities in remote locations, so heavy dependence on regular on-site surveys for toads Frequency of use or residence on islands increases with size, so larger islands face greater risk of cane toad introductions Island selected should be isolated from major rivers so that risk of toads reaching them in wet season floods is low Small uninhabited islands face lower risk of cane toad establishment (e.g. Astell Island at 12.7 km^2) as example, but un likely to support populations of wildlife viable over the long term, therefore requiring insurance of captive population
Mainland exclosures of "average" habitat	Northern Quoll Varanids Snakes	3,121 860-1023 5902-6424	Calculations of area required for MVPs (e.g. northern quoll 220 km^2) are based on average habitat which will mostly be made up of patches of favourable habitat in a matrix of marginal or even hostile habitat. Hence costs of enclosing a single block of typical savanna habitat for quolls may be high.

Situation	Vulnerable species	Annual costs ($000)	Issues
	Non-migratory birds	7364	Yet even exclosures of this size may fail to enclose viable populations of other affected species even if site also contains habitat favourable for those species
Mainland exclosures of high quality habitat	Northern Quoll	1,775	Areas required for MVPs may be greatly reduced in areas of highly favourable habitat in which high densities may be attained, including rocky areas for quolls. A need to take account of higher costs of both construction and maintenance in some favourable habitat types (rocky areas, wetlands, mangroves).
Captive populations alone	Northern Quoll Reptiles	510 104	For comparison with costs and benefits of exclosure strategies, and to permit exploration of "hybrid" strategies (below). Captive population large enough to avoid significant genetic risks.
Mixed strategy – exclosures plus captive breeding insurance	Northern Quoll	1,782	Tradeoffs between robustness of exclosure populations and costs of protecting their integrity may warrant consideration of mixed strategies that take advantage of lower cost options. For purpose of calculations assume an area of the most favourable habitat half that required if no supplementation from captive populations

Table 6: Risks and collateral benefits associated with different "exclusion" tactics for management of cane toad impacts. Categories are necessarily somewhat arbitrary, but are thought to provide a useful summary. The "impact of exclusion failure" column assumes that strategy is the sole or dominant strategy.

Dominant Response	**Conservation benefit for vulnerable species**	**Costs**	**Collateral conservation benefits**	**Risks of exclusion failure**	**Impact of exclusion failure**
Large (MVP+) mainland exclosure with high intensity maintenance	High	Very High	High	Low	Very major
Large (MVP++) peninsular exclosure with high intensity maintenance	Very high	High	Very high	Low	Very major
Large (MVP++) islands with high intensity quarantine	Very high	Moderate	Very high	Low	Very major
Small (MVP-) non-estuarine islands with moderate intensity quarantine	Time limited	Low	High	Substantial	Very major
Small (MVP-) non-estuarine islands with moderate intensity quarantine and large captive populations as "insurance"	High	Moderate	High	Substantial	Low
Small (MVP) mainland "focus" sites of unusually high quality habitat for one or more species and high intensity monitoring	High	High	Substantial	Low	Major
Small (MVP) mainland "focus" sites of unusually high quality habitat with captive breeding "insurance"	High	High	High	Low	Major
Captive breeding only for a range of vulnerable species	Moderate	Moderate	Low	Low	na

14. Discussion

We have calculated minimum habitat areas for a diverse range of taxa, spanning at one extreme species that are relatively small, mostly sedentary and highly vulnerable to cane toads and, at the other extreme, highly mobile species that are much less likely to be affected by the presence of toads in the landscape. We present this span of results to illustrate the numerous challenges facing the adoption of exclusion as a strategy for reducing the impacts of cane toads in north Australia, and potential responses to those challenges, given the present state of knowledge.

Context and limitations

We do not represent these analyses as management prescriptions for general application across the landscapes of northern Australia. In part this is because the information available on the interactions between toads and many species of conservation interest is too weak to permit such an approach (van Dam et al. 2002). However, we also believe that the search for general prescriptions is misguided. All proposals for protection of native fauna through toad exclusion require detailed individual analysis that takes account of local or regional conditions. Rather than a set of recipes, we consider that the particular value of our contribution is to set out many of the issues that will need to be considered and the parameters to be measured or estimated in performing a rigorous analysis.

A particular limitation of our approach is that we have considered the presence of toads in any numbers to be a serious threat to the persistence of vulnerable species in the landscape. With the possible exception of quolls (M. Oakwood, unpublished data) and some goannas (AJ Griffiths, unpublished; D Holland, unpublished data), our choices of species are based on no quantitative assessment of the impact of cane toad exposure. It is therefore impossible to predict changes in the size and dynamics of populations even over the short term, let alone to demonstrate unacceptable risk of local or regional extinction. There is no consensus regarding the probability of adaptation, through individual (and "cultural" through imitation (Dugatkin 2000)) learning from sublethal interactions with toads, or through selection for improved tolerance of bufotoxins or selection for heritable disinterest in preying on toads. Many note the persistence, albeit at reduced densities, of putatively highly vulnerable species like northern quolls in Queensland (S Garnett, pers. comm.) and goannas in Territory sites (WJ Freeland, unpublished) despite large populations of toads.

When the risks of leaving the putative threat unmanaged are effectively unknown, it is difficult to do more than approximately rank options for different responses. An optimal approach is unspecifiable. There is a risk that an overenthusiastic response to toads, if it diverts a substantial proportion of the funds that would otherwise be used for demonstrably robust conservation work, will weaken conservation performance.

To illustrate this point, consider the hypothesis that the effect of toads is an initial acute and then chronic reduction of densities rather than the total elimination of local populations. Subsequent fragmentation of habitats at an otherwise relatively benign level might put dispersed populations of "resistant" individuals at great risk. Should available investments be directed at seeking to maintain landscape integrity at very large scales, rather than building expensive localised exclosures of unknown efficacy? How many species will be maintained by expenditure on the functional equivalent of extraordinarily large cages compared with equivalent expenditures on keeping very often marginal land out of frequently economically marginal production (see Holmes 1996).

Making choices

Under circumstances of continued uncertainty, we consider that relatively modest expenditures on exclusion strategies with a high prospect of long-term success, despite a boisterous climate, are to be preferred over more ambitious approaches. This seems to us to suggest use of natural islands under relatively tight quarantine, backed by captive breeding of vulnerable fauna, to be the optimal mid-term strategy, while biological or other effective controls of toad density are sought.

If expenditures on long artificial barriers are at all justified, they are best designed to complement relevant natural barriers. In the case of the cane toad, this means making use of peninsulas with a morphology that permits exclosure of large areas by a relatively short barrier (as illustrated by Cobourg Peninsula). But this conclusion is also contestable. *Bufo marinus* is capable of surviving considerable periods in seawater, and is regularly observed swimming in saline waters in mangrove habitats, even when freshwater flows from local rivers are too low to significantly dilute these waters (AJ Griffiths, unpublished observations). The design of inter-tidal barriers capable of retaining integrity despite high tidal flows, frequent storms and irregular cyclones, deposition of natural and anthropogenic debris and rapid corrosion, remains uncertain. If such a barrier was to be built, it would be desirable to conduct trials to also allocate funds to assess durability and effectiveness of different designs in limiting the likelihood of toads simply swimming or hopping around them.

Operational considerations

The exercise of producing these estimates of extinction probabilities and costs of exclosure has raised a number of important issues regarding the operational choices associated with such a management tool. Among the most significant are:

(1) The ratio of perimeter to area enclosed decreases with increasing size of exclosures. For example, for a square exclosure, doubling the perimeter increases the area enclosed four-fold. The area enclosed increases rapidly with increased investment in perimeter fencing.

(2) Conversely, the probability of extinction rises approximately exponentially as the enclosed area - and hence the population protected – decreases. As a consequence, risks of failure increase very rapidly if attempts are made to extract savings from reduced expenditure on a perimeter (Figure 2).

(3) Dependence on a single population and hence the integrity of the structure that protects it is an inherently high-risk approach. In many areas of favourable cane toad habitat (especially coastal habitats), cyclones or other high intensity storms are frequent. There is a small but measurable risk of catastrophic failure. In regions subject to such storms, following severe events, attention to conservation issues is likely to be accorded lower priority than urban and other infrastructure like utilities, roads, and housing. As a consequence, repair is unlikely during ensuing periods of high rainfall and, presumably, high rates of dispersal of cane toads at all stages of the life cycle. cane toad invasions of exclosures of substantial size may be effectively irreversible if delays in repair are protracted. Non-estuarine islands of reasonable elevation, and hence low susceptibility to flooding during storm surges of up to several metres above normal tidal limits, are likely to provide much more robust protection in the face of cyclonic conditions.

(4) Other catastrophic events, such as outbreaks of disease, also threaten single populations. Responses to such risks might include erection of multiple "natural" exclosures or maintaining genetically relevant captive populations, or both. Obviously the cost of individual structures and the increasing perimeter to area ratio with smaller exclosures (Figure 2) will inhibit strategies like subdivision to reduce risk of simultaneous catastrophic failure. The maintenance of captive breeding colonies will in most cases appear to be a considerably lower cost option than multiple exclosures, but captive populations are not immune from risk of disease, destruction in storm or fire. The

desirability of maintaining a number of genetically distinct stocks may also increase the desirability of multiple captive populations.

(5) A widely applied rule of thumb for determining the size of captive populations needed to maintain genetic integrity (retaining 90% of genetic variance over 100 years) is 475/L, where L is the generation length. We have applied this rule of thumb in our estimates. However, it should be noted that with close management, especially of family size, this can be approximately halved, so that for quolls the minimum requirement will be about 250, and for large reptiles of the sort we considered, 100 plus (Frankham et al. 2002).

(6) The most effective conservation strategies that take account of uncertainty and risk of vulnerable fauna will require "hybrid" responses, rather than a search for a single cost- or conservation-optimal design.

(7) Given high costs of management of the toad threat and finite resourcing for conservation, responses to the issues raised by toads have the potential to constrain other conservation activity. We sought to reflect this additional "cost" by incorporating interest payments in the estimates of the annual cost of the large capital expenditures on structures, which serves to help emphasise the scale of such investments compared with many other conservation programs. However, we have not provided equivalent estimates of the collateral benefits of toad-based investments that secure sites against other threats. For example, good biological "quarantine" systems for islands may return multiple conservation benefits protecting against a wide range of potential threats.

(8) We have taken no account of the cost of placing barriers to the movement of small mammals, reptiles and amphibians across large parts of the landscape, nor the loss of habitat in wide firebreaks. Risks associated with such issues would need to be assessed case by case and inform judgments about the net benefits of such interventions.

15. Conclusions

This new work on deriving minimum viable population sizes combined with estimates of costs of building relevant exclosures against cane toads has implications that extend beyond the immediate cane toad management problem. Areas required for minimum viable populations of important elements of the north Australian fauna are large and the costs of excluding threats are accordingly very high. The minimalist goal of avoiding conservation disasters (extinctions) by such interventions is not self-evidently cheaper nor demonstrably more effective than conservation strategies that seek to maintain landscape integrity rather than respond to individual threats. It remains an open question whether a more rational approach to the mid to long term problem of cane toads is to seek to better manage other processes that will exacerbate effects of increased mortality of vulnerable species, especially habitat fragmentation.

To depend on enclosing small spaces - which remain vulnerable and require intense ongoing intervention - is to retreat from the larger issues confronting northern Australia and to avoid the difficult long term questions. It seems reasonable to suggest that under some circumstances, such exclosures may provide a useful adjunct to more comprehensive strategies. However, those comprehensive strategies must address, realistically and preferably quantitatively, the situation that will apply to toads after an achievable biological and other control program. A sober appraisal based on the situation with other species (for example the rabbit) that have been subject to focused attention for decades, backed by resources for control that are unlikely to ever be devoted to species that do not threaten agriculture or human health, is that the pest will remain common in the landscape. Thus the long term strategy must address the interaction of the toad threat with other processes like land clearing and habitat fragmentation, fire management, and other invasive species. A magic bullet for the toad, no matter how expensive and how effective, is only one part of a much larger problem confronting many more elements of the north Australia biota.

Appendix: The candidate species

(i) *Canis lupus dingo* (dingo)

Dingos are remain relatively common in Australia and are found in scattered groups across Southeast Asia. With the exception of Tasmania they were formerly found throughout the entire Australian continent but are now absent from densely settled parts of the south-east and south-west (Menkhorst 2001).

Pressures on the dingo population include habitat loss and culling by humans. They compete with foxes and feral cats for small animal food sources but have shown greater success with large prey, especially in times of drought. The dingo has also benefited from access to stock watering points and the provision of abundant non-native food (Corbett 1995). The greatest current threat to pure populations of dingo appears to be hydrisation due to interbreeding with domestic/wild dogs (Corbett 1995; Menkhorst 2001; Hintze 2002). Dingo populations in both the Borroloola and the Roper River region have been shown to be adversely affected by the cane toad at least in the short term (Catling et al. 1999). Dingos were ranked among the 10 highest at risk species from the cane toad in Kakadu National Park (van Dam et al. 2002).

Undisturbed by humans, dingos form packs of 3-12 individuals. Dingos are monogamous and cooperative breeders. Packs have distinct male/female hierarchies with only the dominant pair breeding successfully. They breed once per year and average 5 pups per litter. Dominant females will kill the young of other females within the pack. Dingos obtain sexual maturity around 22 months and pair during their 3rd year, often mating for life (Corbett 1995).

Dingos are opportunistic carnivores (Menkhorst 2001), 60% of their diet being mammals, with reptiles and birds making up the remainder (Strahan 1983). Although they eat a diverse range of prey they tend to specialise on abundant species, changing hunting strategy to maximise success (Corbett 1995).

A pack typically remains in the territory of their birth, traveling 10-20km per day. Home ranges vary from 10 to 77 km^2 depending on the environment. Home ranges are larger in arid regions and smallest in the moist forested mountains of E and SE Australia. Home range is a function of the reliability and regularity of food availability and terrain rather than pack size (Corbett 1995).

(ii) *Dasyurus hallucatus* (northern quoll)

The northern quoll's range has become increasingly smaller and fragmented. Currently it is restricted to six main areas, the Harmersley Range and the Kimberly in Western Australia, northern and western parts of the Top End in Northern Territory, northern Cape York, Artherton in the Cairns area and the Carnarvan Range in the Bowen areas, Queensland. Formerly they were found across northern Australia from the northwest cape Western Australia to southeast Queensland (Menkhorst 2001). The northern quoll is considered to be near threatened (Parks and Wildlife Commission NT 2003).

Although the reason for the species decline is not well understood, impacts that degrade the habitat including changes to grazing and fire regimes which remove shelter and increase their vulnerability to predation are likely causes. In combination with increased predator abundance (feral dogs and cats) and road kill, such habitat change may have lead to the broad scale decline of the northern quoll (Oakwood 2000; Department of Environment and Heritage 2004). Recent research in Kakadu has shown northern quoll populations declining at alarming rates prior to cane toad arrival due to unknown factors (van Dam et al. 2002). There is a large amount of anecdotal evidence of local population declines in areas following the introduction of the marine toad (Burnett 1997; Phillips et al. 2003). This species has been assigned the highest priority in the risk assessment of cane toads in Kakadu National Park (van Dam et al. 2002).

Quolls have a naturally short life span, the northern quoll reaching sexual maturity at 11 months in both sexes, most females surviving 2 breeding seasons and reaching a maximum age of 3 years while males rarely survive the breeding season, reaching a maximum age of 14 months (Oakwood 2000). Litters average six young, with around a third lost prior to independence. Whereas most females remain in the area they were born, most males disperse from the natal area by the age of 6-8 months (Strahan 1983).

The northern quoll is an aggressive predator. Diet is varied and may include small mammals, reptiles and insects, as well as figs and other soft fruit (Strahan 1983)

The northern quoll is most common in rocky , sparsely vegetated areas and open woodlands (Department of Environment and Heritage 2004). There is a large difference in the size of reported home ranges for the species. Schmitt et al recorded home ranges between 0.2ha to 3.5ha for both sexes (Schmitt et al. 1989). Oakwood reported the female home range to be 35ha with some overlap of foraging ranges when the density was high (3-4 females/km2), and the male home range to be similar, expanding during the mating season to more than 100ha (Oakwood 2002).

(iii) *Epippiorhynchus asiaticus* (black necked stork/Jabiru)

The black necked stork is found in north and northeastern Australia from the Pilbara, Western Australia to eastern Queensland and southern New Guinea (del Hoyo et al. 1992). In the past it was found along the coastal strip as far south as the Hunter River, although at present there are few records from New South Wales (Garnett and Crowley 2000). The species global status is least concern (Garnett and Crowley 2000). However, it has shown decline in the southern end of it's range and is listed as endangered in New South Wales and rare in Queensland (Dorfman et al. 2001).

Black necked storks are highly susceptible to disturbance and prefer areas little visited by humans. Habitat loss is by far the greatest threat over their distribution range. This includes felling of nest trees, encroachment of agriculture and aquaculture (Birdlife International 2003), and the degradation of wetlands through drainage, invasion of weed species, salinisation and siltation (del Hoyo et al. 1992). Other threats outside Australia include capture for pet trade and zoos, and in Australia and elsewhere, collision with electricity wires and the introduction of the cane toad (Sundar 2003). Although this species is ranked as an uncertain (high) risk in the risk assessment of cane toads in Kakadu National park it is likely to consume native anurans and exhibits foraging behaviour that will probably maximise exposure to the cane toad metamorphlings and possibly adults (van Dam et al. 2002).

The black necked stork rarely occurs in groups being dispersed as single birds or pairs or loose family flocks (Sundar 2003). Larger aggregations may occur when severe drought reduces suitable habitat (Simpson et al. 1999). Outside of the breeding season, flocks of up to 100 birds form (Australian Museum 2003). Pairs remain together for many years and tend to use the same sites repeatedly (Simpson et al. 1999) and are known to use the same nest in successive breeding attempts (Sundar 2003). Clutches of 2-4 eggs are produced and 2 or 3 chicks may be raised successfully to fledging (Simpson et al. 1999). Young birds stay with the adults for a considerable time and do not disperse far (del Hoyo et al. 1992).

Black necked storks are completely carnivorous feeding mainly on fish, but also taking frogs, snakes, turtles, crabs, prawns, molluscs, beetles and arthropods (del Hoyo et al. 1992).

The storks prefer comparatively undisturbed freshwater wetlands (del Hoyo et al. 1992). They forage in river pools, swamps, irrigated crops, dry floodplains and open grassy woodland but are less often found along the coast, occasionally in mangroves and rarely on coastal mudflats (del Hoyo et al. 1992; Simpson et al. 1999; Birdlife International 2003).

There are few data on the size of the home range, but a belief that each pair require a large territory (del Hoyo et al. 1992).

(iv) *Dacelo leachii* (blue winged kookaburra)

The range of the blue winged kookaburra includes Northern Australia and New Guinea. It is found in northeast and northern Queensland, northern Northern Territory and northern Western Australia (Simpson et al. 1999; del Hoyo et al. 2001).

The major threat to the kookaburra is habitat destruction resulting from the clearance of woodland and forest for farming. They are not globally threatened and are fairly common over most of their range (del Hoyo et al. 2001). There is a risk of the cane toad affecting their population as they occupy a broad range of habitats that will more than likely see them encounter the cane toad (van Dam et al. 2002).

The family group consists of up to 8 individuals composed of a pair and its offspring from previous years. The male bird and the auxiliaries (of which a larger number are male) assist in the preparation of the nest, feeding of the breeding female prior to laying, incubation, feeding the chicks and territorial defense. Clutch sizes are up to 5 eggs although generally only 2 chicks survive. A third chick may survive if food is plentiful. In general the kingfishers are quite long-lived, surviving up to 12 years in the wild and greater than 15 years in captivity (del Hoyo et al. 2001). Breeding pairs form long term bonds which are probably life long (Marchant and Higgins 1990).

Their diet consists of mainly invertebrates, small vertebrates (including frogs – need to clarify this in all cases, including discussion of nocturnality etc), small birds, birds eggs and small mammals. In Kakadu it consisted of 59% invertebrates and 41% vertebrates (del Hoyo et al. 2001)

The blue winged kookaburra prefer open tropical and subtropical Euclayptus forests, woodlands (Marchant and Higgins 1990) and paper bark swamps (Simpson et al. 1999) avoiding areas with dense understorey (del Hoyo et al. 2001). In the Northern Territory they are found at a density of 0.08-0.72 birds per hectare (Marchant and Higgins 1990). Their mean territory size is $0.4km^2$ (del Hoyo et al. 2001).

(v) *Aquila audax* (wedge tailed eagle)

The wedge tailed eagle is found throughout mainland Australia, Tasmania and southern New Guinea (Australian Museum 2003). However, the population in Tasmania is considered a different sub species (del Hoyo et al. 1994). Although the mainland subspecies is widespread and common the Tasmanian race is endangered and has been reduced to 60-80 breeding pairs (del Hoyo et al. 1994).

Supposed impact on domestic stock has given rise to a long history of human persecution. Even now the eagle is subject to illegal shooting and poisoning. Local declines in southern Australia have been attributed to habitat disturbance, especially in heavily settled and farmed areas (del Hoyo et al. 1994; Simpson et al. 1999). Although the eagle may benefit from thinning of tree cover, introduction of the rabbit and access to abundant carrion from road kills etc, intolerance of human activity leads to nest abandonment and therefore threatens breeding success (del Hoyo et al. 1994).

Eagles are sexually mature at 3 years and although they may pair in immature plumage, they seldom breed before adult plumage at 6 years (del Hoyo et al. 1994). They are monogamous and apparently mate for life unless one bird of the pair is killed, after which the survivor will find a new mate (Australian Museum 2003). They lay 1-3 eggs rarely 4, and usually rear only one young per clutch, although in a good year, two chicks may fledge in some nests. They are a fairly long-lived species with records of birds living to 40 years in captivity. Although predominantly carrion eaters, the wedge tailed eagle also takes live prey like rabbits, small macropods, reptiles and birds (del Hoyo et al. 1994). Although there is no direct evidence of wedge tailed eagles taking cane toads in Queensland and the eastern half of the Northern Territory, their documented willingness to prey on other herpatiles (del Hoyo et al. 1994), and the conspicuousness and abundance of the cane toad in their range, suggests that some predator-prey interactions between these two species are highly likely. They may also feed on the carrion of individuals of other species (e.g. goannas) that have been killed by ingesting cane toads.

Wedge tail eagles can live in most terrestrial habitats but avoid areas of dense human population and dense rainforest (del Hoyo et al. 1994). They prefer wooded and forested land and open country (Australian Museum 2003). Established breeding pairs will defend the home range around their nest sites from other eagles but will hunt for food in a larger territory that they do not defend (Australian Museum 2003). Home ranges differ according to the region, breeding pairs in temperate regions occupy from 30-35 km^2 and in arid lands 3-6 birds may occupy 100km^2 (Zoological Parks and Gardens Board of Victoria 2003). Other ranges recorded are 28-32km^2 for the eastern highlands (NSW), 53km^2 in arid NSW and 32-108km^2 in arid Western Australia (Sharp et al. 2001). Nesting densities were calculated for western

NSW to be one pair per 3-9km^2 and in the semi arid zone one pair per 40-48km^2 (Sharp et al. 2001).

(vi) *Ixobrychus flavicollis* (black bittern)

Black bitterns are found in the Moluccas, New Guinea, Bismarck Archipelago and Australia (del Hoyo et al. 1992). Within Australia they are widely distributed in the near coastal region, from southern New South Wales north to Cape York and along the entire northern coast to the Kimberley region, also in the south western corner of Western Australia (Marchant and Higgins 1990). Although the northern Australian population is apparently secure, declines have occurred along the southern margins of its range (Garnett and Crowley 2000) coinciding with clearing for agriculture and increased salinity of rivers (Marchant and Higgins 1990).

Threats include habitat loss and predation by feral cats on eggs and young. A wide range of activities have affected habitat availability including clearing, grazing and trampling of riparian vegetation and salinisation, siltation and pollution of wetlands and waterbodies (National Parks and Wildlife Service NSW 1999). There are some reports of deaths of this species after ingesting juvenile cane toads and, due to it's broad range of habitats it will more than likely encounter cane toads (van Dam et al. 2002).

Usually solitary during the non-breeding season the black bittern can sometimes be found in small colonies. During the breeding season they are seen in pairs (Marchant and Higgins 1990; del Hoyo et al. 1992). They are monogamous and both parents incubate and tend the young until fledging (Marchant and Higgins 1990; del Hoyo et al. 1992). They are generally single-brooded and produce between 3 to 6 eggs, normally 4 (National Parks and Wildlife Service NSW 1999).

The black bittern are nocturnal and generally feed at dusk and at night (Marchant and Higgins 1990). They feed on fish, frogs, molluscs, crustaceans and insects (del Hoyo et al. 1992).

Typically the black bittern is found in areas where permanent water and dense vegetation are present. Preferring densely forested freshwater streams and pools or other wetlands. In Australia it also frequents mangrove and Melaluca swamps, margins of estuaries, lagoons, tidal creeks and mudflats (del Hoyo et al. 1992; National Parks and Wildlife Service NSW 1999; Simpson et al. 1999). Currently there is no available information on the home range of the black bittern

(vii) *Ardeotis australis* (Australian bustard)

The Australian bustard is found in all states but is generally rarer or absent in the south, especially in the southeast. It is also found in southern Papua New Guinea ranging into Irran Jaya (del Hoyo et al. 1996). The Australian population is thought to number at least 100 000 birds with the majority occurring in northern Australia (Birdlife International 2003), where heavy rains reduce access of humans whilst they are breeding (Simpson et al. 1999). Their range has contracted markedly since the first settlement (Marchant and Higgins 1990) and they are considered endangered in New South Wales and Victoria and vulnerable in South Australia (Stanger et al. 1998). Their global status is near threatened (del Hoyo et al. 1996; Garnett and Crowley 2000).

Heavy hunting for food and sport, up to at least 1940 when they were formally protected, greatly affected their distribution. Other factors in their decline include habitat destruction (including intensive agriculture and invasion of pastoral land by woody weeds), and the impact of introduced animals (particularly the fox) (Marchant and Higgins 1990; del Hoyo et al. 1996; Simpson et al. 1999; Birdlife International 2003). Traditional and illegal hunting is still considerable and pesticides have been responsible for local extinctions (Garnett and Crowley 2000). There may be some increase in abundance in response to clearing, but this effect dissipates as agriculture intensifies (Birdlife International 2003).

The Australian bustard is loosely gregarious with single birds or small groups of 2-6 in sight of others at favorable feeding and breeding sites. Non-breeding birds appear sparsely scattered in small groups of 2-10, and much more rarely in flocks of hundreds where food and water are abundant (Marchant and Higgins 1990). They are polygamous and lay 1-2 eggs (del Hoyo et al. 1996; Simpson et al. 1999) and are hatched with a covering of down and open eyes, capable of leaving the nest alone within a few days (Simpson et al. 1999). Only the female incubates and stay with the young (Marchant and Higgins 1990).

Nomadic omnivores, the Australian bustards eat myriapods, arachnids, insects, reptiles, young birds, small rodents, molluscs, shoots, roots, leaves, flower-heads, seeds and berries (Marchant and Higgins 1990; del Hoyo et al. 1996; Simpson et al. 1999), and thus although they may not consume cane toads directly, would likely still compete with them for food. Individuals feed during the day on the ground in open grasslands (Marchant and Higgins 1990).

Australian bustards are generally confined to areas where the upper canopy cover is less than 10% or under 2m high or near areas where grasses are dominant (National Parks and Wildlife Service NSW 1999). They are found mostly in grassland dominated by tussocky forms, also in sparse low shrubland, savanna, grassy woodland, artificial landscapes such as pastural land, crops and golf-courses (del Hoyo et al. 1996; Simpson et al. 1999; Garnett and Crowley 2000). The species is highly nomadic and moves in response to rainfall (National Parks and Wildlife Service NSW 1999).

(viii) *Aspidites melanocephalus* (black headed python)

Black headed pythons are found in the northern third of Australia except in the extremely arid regions (Mirtschin and Davis 1992; Barker and Barker 1994; Cogger 2000). Humans, cats dogs and foxes are the main threat to populations

Sexual maturity is obtained in 3 years for females and 18 month in males. Clutch size ranges from 3-18 eggs, (Mirtschin and Davis 1992; Barker and Barker 1994; Torr 2000; Geer 2003), with an average of 12 (G. Bedford, CDU, pers com).

The black headed python is primarily nocturnal and preys mainly on reptiles including a wide variety of lizards and snakes (including venomous species) and occasionally on small mammals and birds (Barker and Barker 1994; Torr 2000). Although there is no direct evidence of predation by black headed pythons upon frogs, they may opportunistically choose to ingest cane toads once they become numerically abundant in the black headed python's habitat. There may also be indirect effects of toads on the python if the presence of toads drives a change in the abundance of small mammals, their primary prey item.

They are most often found in woodlands, open forest and rocky areas but also inhabit grassland and shrubland and are reported from a wide range of other habitats. They spend a considerable time underground, using the burrows of mammals and goannas or occasionally excavating their own burrow (Barker and Barker 1994). Bedford's previous studies on the home range of four other python species measured homeranges between 0.3ha to18ha. Based on his observations he estimates the black headed python to have a home range likely to be double that of the greatest measured for the other species (G. Bedford, CDU pers com)

(ix) *Acanthophis praelongus* (northern death adder)

The northern death adder is found in the subhumid to humid areas of the Kimberley Ranges, northern Northern Territory, northern Queensland and possibly in southern New Guinea (Storr et al. 1986; Mirtschin and Davis 1992).

Habitat destruction appears to be the most significant factor in the decline of death adders. However they have been observed to disappear from areas shortly after the introduction of the cane toad (Grigg et al. 1985). The northern death adder has been identified in Phillips study as one of 49 snakes that are at risk from the invasion of the cane toad based on the overlap of their distribution with the toad and their dietary composition. It was shown that they have the ability to ingest a single toad large enough to be fatal (Phillips et al. 2003). This species has also been identified as one of the 10 species of high risk in the risk assessment of cane toads in Kakadu National Park (van Dam et al. 2002).

These snakes are live bearers (Cogger 2000). Litter sizes vary from 13-33 with an average of 23.1 (Webb, Shine and Christian, unpublished data)

Northern death adders are nocturnal ambush feeders that use caudal luring to attract prey (Webb et al. 2002). They eat amphibians, lizards, small mammals and birds (Mirtschin and Davis 1992; Cogger 2000).

They inhabit a range of habitats prefering grasslands, woodlands (wet and dry eucalypt forests), rocky ranges and outcrops (Mirtschin and Davis 1992). There is no information to date regarding the size of their home range.

(x) *Varanus panoptes* (northern sand goanna)

The northern sand goanna is found in the Kimberley and arid western regions of Western Australia and Northern Territory (Cogger 2000).

There is little information on the impact man has on this species. Generally the conservation status of all Australian varanids is sound. The extent of traditional use of the goanna for food varies; they comprise only a small part of the diet of residents of coastal regions but make up a larger proportion of the meat in the diet of desert dwellers. The presence of the cane toad poses a major threat to many northern Australian species of goanna (King and Green 1999). There is some anecdotal evidence that declines have occurred following the appearance of the toad (Burnett 1997; Phillips et al. 2003). Long term sampling of the northern sand goanna in Booroloola prior to and after the invasion by cane toads showed population decline. Survivors of the initial decline are thought to “seed” a recovering population that doesn’t attack the toad (Freeland unpublished; van Dam et al. 2002). This species was ranked among the 10 highest risk species in the risk assessment of cane toads in Kakadu National park (van Dam et al. 2002).

The northern sand goanna has clutch sizes of 7-13 (Geer 2003). Clutches are buried deep in the soil, especially along the margins of creek beds (King and Green 1999).

This ground dwelling monitor feeds on a variety of prey, largely on insects and small terestrial vertebrates (Cogger 2000). As they are found mainly in riparian habitats their diet consists of large amounts of aquatic prey (Shine 1986).

The northern sand goanna occurs in a variety of habitats including beaches, beach dune grasslands, grasslands, mangroves, woodland, monsoon forest, open forest and vine forest (Geer 2003). Home range sizes of 3ha have been recorded on the Adelaide river floodplain (T. Madsen, pers com).

(xi) *Varanus indicus* (mangrove monitor)

The Mangrove monitor is found in the rainforest and coastal mangrove habitats of eastern Cape York Peninsula and the islands of the Torres Strait, coastal mangrove forests of Arnhem Land, Northern Territory, New Guinea and other parts of the Indo-Papual Archipelago (Cogger 2000). Within this large distribution there is much variation in size, pattern and scalation (Bennett 1998).

On the Marshall Islands the marine toad proved toxic to the lizard (Bennett 1998), and a decline of the species on Kayangel Atoll was noted following the introduction of the cane toad (Burnett 1997). The mangrove monitor was ranked among the 10 highest risk species in the risk assessment of Kakadu National Park. Although it does not appear to occupy key cane toad habitats it often forages around the back swamps and paleochannels of the floodplain where it will probably be exposed to them (van Dam et al. 2002).

In varanids the clutch size is generally related to body size (King and Green 1999). The clutch size of the mangrove monitor is probably smaller than would be expected for a medium sized lizard. However, observation suggest that when food is abundant that they may reproduce frequently, producing a large number of small clutches of 1-6 eggs (Bennett 1998).

Diet of mangrove monitors consists predominantly of frogs, lizards, crabs, fish, insects and small mammals (King and Green 1999) and also includes birds and their eggs and the eggs and young of turtles and crocodiles (Bennett 1998). Mangrove monitors are always found close to water (Bennett 1998). They are restricted to coastal mangroves in northern Australia and rainforest and coastal mangrove on the eastern Cape York Peninsula and Torres Strait islands (Cogger 2000). Home ranges of this species have been estimated in the Northern Territory to be 0.9ha for females and 0.4ha for males (J. Smith, CDU, pers com}

(xii) *Chlamydosaurus kingii* (frilled neck lizard)

The frilled neck lizard is found in the Kimberley district, Western Australia through the top end of Northern Territory to the Cape York Peninsula, eastern Queensland and southern New Guinea (Cogger 2000).

Lizard numbers in southeastern Queensland have diminished as a result of land clearing, whilst predation by cats and death in fires have resulted in direct losses. There is some anecdotal evidence that decline in numbers has followed the introduction of the cane toad (Phillips et al. 2003).

This lizard has an early maturing and multiple brooded life history strategy. Double clutching has been recorded. However, not all females reproduce in each reproductive season or in consecutive years (Griffiths 1994). Reportings of clutch size varies considerably, ranging from 3 to 23 with a positive correlation between female SVL and clutch size (Bedford et al. 1993; Geer 2003). In captivity this species has lived for at least six years. However, given the large size of the species it is likely that it could live much longer (Greer 2003).

Frilled necks are sit and wait predators that eat insects such as lepidoptera larvae, termites and ants, and more rarely small vertebrates (Shine and Lambeck 1989; Greer 2003), though there is no record of them consuming frogs or tadpoles, and so they may be at low risk from cane toads except indirectly via competition for common food. Primarily arboreal frilled neck lizards and are found in savanna woodlands (Greer 2003) and dry sclerophyll forests (Cogger 2000; Savage 2001). Their home range size differs between sexes and during the wet and dry seasons. Adult male home range size during the dry is recorded as 1.96±0.57 ha (n=16) and during the wet 2.53ha, adult female home range size in the dry is recorded to be 0.634±0.12 ha and during the wet 0.68ha (n=7) (Griffiths 1994).

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