

Ecography

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Supplementary material

Appendix 1: Description of coupled niche-population models

Study species

Six study species were chosen to represent a range of plant and animal life history types. The plant species were *Angophora hispida*, *Banksia baxteri* and *Xanthorrhoea resinosa*. The animals included two lagomorphs (*Oryctolagus cuniculus*, *Lepus timidus*) and a reptile (*Tiliqua adelaidensis*). Published spatially explicit population models exist for each species (Anderson et al. 2009, Fordham et al. 2012a, Fordham et al. 2012b, Fordham et al. 2012c). The following table summarizes the life history characteristics of the study species.

Species	Kingdom	Adult Survival	Rmax	Mean Dispersal (km)	Long Distance Dispersal (KM)	Published Model
<i>Angophora hispida</i>	Plantae	0.85	> 3.5	0.2	3	Fordham et al. 2012
<i>Banksia baxteri</i>	Plantae	0.95	2.7	0.5	5	Fordham et al. 2012
<i>Xanthorrhoea resinosa</i>	Plantae	0.96	1.6	0.5	4	Fordham et al. 2012
<i>Lepus timidus</i>	Animalia	0.7	1.4	2	20	Anderson et al. 2009
<i>Oryctolagus cuniculus</i>	Animalia	NA	1.5	1.8	9	Fordham et al. 2012b
<i>Tiliqua adelaidensis</i>	Animalia	0.77	2.5	0.1	0.5	Fordham et al. 2012c

Anderson, B. J., et al. 2009. Dynamics of range margins for metapopulations under climate change. - Proc. R. Soc. Lond. Ser. B-Biol. Sci. 276: 1415-1420; Fordham, D. A., et al. 2012a; Modelling range shifts for invasive vertebrates in response to climate change. - In: Brodie, J., Post, E. and Doak, D. (eds.), Wildlife Conservation in a Changing Climate University of Chicago Press, pp. Chapter 5. in press; Fordham, D. A., et al. 2012b. Plant extinction risk under climate change: are forecast range shifts alone a good indicator of species vulnerability to global warming? - Glob. Change Biol. 18: 1357–1371; Fordham, D. A., et al. 2012c. Managed relocation as an adaptation strategy for mitigating climate change threats to the persistence of an endangered lizard. - Glob. Change Biol. 18: 2743–2755.

Stochastic population models

Spatially explicit, stage-structured, stochastic metapopulation models were implemented in RAMAS GIS v5 (Akçakaya 2005).

Spatial structure

The spatial structure of the metapopulation models (i.e., location, size, and shape of populations) were

determined by the distribution of suitable habitat patches each year. Patch structure was defined using different ecological niche modelling (ENM) techniques and predictor variables:

- MaxEnt (Phillips et al. 2006) and four climate variables (maximum temperature in the warmest month, minimum temperature in the coolest month, summer and winter rainfall) for *A. hispida*, *B. baxteri* and *X. resinosa* (Fordham et al. 2012b);
- Bio-ensembles (an automated ensemble forecasting approach; Diniz-Filho et al. 2009) and four climate variables (maximum and minimum temperature of the hottest and coolest month, respectively; and annual and summer rainfall) for *O. cuniculus* (see Fordham et al. 2012a for details);
- General Additive Models (Hastie 1992) and four climate variables (mean temperature of the warmest month, mean temperature of the coldest month, annual precipitation and potential evapotranspiration) for *L. timidus* (see Anderson et al. 2009 for details); and
- Aggregated boosted regression trees (Elith et al. 2008) and three climate variables (mean annual rainfall, minimum winter temperature and maximum summer temperature) and two landscape variables (geological substrate and slope) for *T. adelaidensis* (Fordham et al. 2012c)

The suitability of habitat was defined by a threshold minimum value representing the 10th percentile of non-zero habitat suitability values produced by respective ENMs for each cell in the modelled landscape for *A. hispida*, *B. baxteri* and *X. resinosa*; and the 15th percentile for *T. adelaidensis*.

Alternatively, a land vegetation cover map was used to guide the setting of the habitat suitability threshold for *L. timidus* (Anderson et al. 2009). For *O. cuniculus* a climate suitability threshold was set based on present-day physiologically suitable habitat (Fordham et al. 2012a). Patch structure (size and location of habitat patches) was predicted for each species for each year between 2000 and 2080, using climate estimates from future climate scenarios (see Table 1). For all species, with the exception of *L. timidus* (see Anderson et al. 2009 for details), climate forecasts were generated using MAGICC/SCENGEN (Fordham et al. 2012d).

Habitat suitability functions included static as well as dynamic variables (Anderson et al. 2009, Fordham et al. 2012a, Fordham et al. 2012b). Habitat suitability for *T. adelaidensis* included a binary mask that reduced the suitability of regions that experience (or have incurred) high soil disturbance practices or urbanization to zero; and a mask that decreased the habitat suitability of regions not frequently grazed by live-stock by 50% (Fordham et al. 2012c). Each patch of habitat was defined as a cluster of nearby grid cells with suitable habitat, and was assumed to support one population. Adjacent populations were delineated by a neighbourhood distance threshold (see Akçakaya 2000 for details of method). Habitat suitability and neighbourhood distance thresholds are listed for each case study species in the table below.

Species	Habitat Suitability Threshold	Neighbourhood Distance (km)	Emission Scenario
<i>Angophora hispida</i>	0.12	1.5	High (WRE750; Wigley et al. 1996)
<i>Banksia baxteri</i>	0.22	2	High (WRE750; Wigley et al. 1996)
<i>Xanthorrhoea resinosa</i>	0.14	2	High (WRE750; Wigley et al. 1996)
<i>Lepus timidus</i>	0.25	3	Medium (A2; Nakicenovic & Swart 2000)
<i>Oryctolagus cuniculus</i>	0.25	6	High (WRE750; Wigley et al. 1996)
<i>Tiliqua adelaidensis</i>	0.20	1.2	High (MiniCAMRef; Clark et al. 2007)

Clarke L.E., Edmonds J.A., Jacoby H.D., Pitcher H., Reilly J.M. & Richels R. (2007). Scenarios of greenhouse gas emissions and atmospheric concentration. In. A report by the US Climate Change Science Program Washington, p. 154.; Nakicenovic N. & Swart R. (2000). Emissions Scenarios. Cambridge University Press, Cambridge.; Wigley T.M.L., Richels R. & Edmonds J.A. (1996). Economic and environmental choices in the stabilization of atmospheric CO₂ concentrations. *Nature*, 379, 240-243.

Demographic structure

Stage-based matrix population models were constructed for each of the three plant species, *L. timidus* and *T. adelaidensis*. A scalar type population modelling approach (Akçakaya 2002) was used for *O. cuniculus*.

Plant Models: Models for *B. baxteri* had a single seedbank stage in which some seeds remain dormant and some germinate (transition to seedling) during each time step. The remaining two plant species had no seedbank stage, meaning that all seeds either became seedlings or died during the same year that they matured (i.e. no storage between years). All three plant species had seedling stages in which individuals either progressed to the next stage or died (i.e. annual flux of individuals). Juvenile stages

were modelled similarly, with higher survival rates, with the exception of *B. baxteri*. Individuals within each juvenile stage for *B. baxteri* could either progress to the next stage or progress to the smallest adult stage (i.e., to simulate plants with accelerated growth). All species had several adult stages which were capable for producing seeds or, in the case of species with no seedbank stage, seedlings. The relative size and fecundity of individuals increased as they progress through the stages. Individuals in adult stages could either remain in the same stage, progress to the next stage or die during an annual time step. *B. baxteri* had a senescent stage, which was similar to adult stages but had lower rates of survival and fecundity. The stage matrices are provided by Fordham *et al.* 2012b.

Fecundity transitions from each of the mature and senescent stages represent the product of the mean number of seeds produced per plant, the proportional viability of newly produced seeds, and the proportion that escape predators during fruit ripening. For species without seedbanks, this was also multiplied by the proportion of individuals escaping post-dispersal predation, but for those with seedbanks post-dispersal predation was incorporated into the transition from.

Animal Models: Population models for *L. timidus* comprised 3 stage classes - leveret; one year old; and those two years or greater. Leveret survival was estimated at 50% and adult survival at 70%. Age of maturity was modelled at 1 year (Iason 1989) and fecundity was related to age, with an average litter size of 5 for 2-year olds and 6 for 3-year olds and older (Anderson *et al.* 2009). The stage matrix is shown in Anderson *et al.* 2009.

Models for *T. adelaidensis* consisted of five stage classes. Survival for juveniles was low and high for adults, 0.12 and 0.77 respectively (estimates based on Milne 1999). Clutch size was influenced by age/body size (Milne 1999). The stage matrix is described in detail by Fordham *et al.* 2012c

Population models for *O. cuniculus* were not stage structured, but generated using a scalar approach (Dunham *et al.* 2006) based on time series data (Fordham *et al.* 2012a).

Variability

Both demographic and environmental stochasticity were incorporated into the models. RAMAS implements demographic stochasticity by sampling the number of survivors from a binomial distribution, and the number of seeds/offspring produced from a Poisson distribution (Akçakaya & Root 2005). Environmental stochasticity was sampled from lognormal distributions with coefficients of variation set to 10% for survival and growth transitions, and 20% for fecundity and germination transitions for the three plants (Fordham et al. 2012b). For *T. adelaidensis* we estimated temporal variability in vital rates from a long term capture-mark-recapture study (1982 to 1991) of a congeneric and sympatric species, *Tiliqua rugosa* (Bull 1995). Coefficients of variation for survival and fecundity were set to 3.5% and 37% (Fordham et al. 2012c). For *L. timidus* variability in vital rates was estimated by trialling values until the fluctuations in the simulated populations matched those of the time series (Anderson et al. 2009). Time series data was used to estimate variation for *O. cuniculus* (Fordham et al. 2012a).

Correlation among fluctuations in subpopulations caused by environmental stochasticity depended on their spatial separation. Pairwise correlations were calculated using long-term annual rainfall (or winter rainfall data in the case of *X. resinosa*) from weather stations within the distribution of each species. An exponential function, $P = a \cdot \exp(D^{c/b})$, where D is the distance between centroids of population locations and a, b and c are constants, was then fitted to the data to calculate the pairwise correlations in vital rates between populations.

Catastrophes

Fires directly influenced the vital rates of *A. hispida*, *B. baxteri* and *X. resinosa*, by activating or enhancing the transition from seeds to seedlings, interrupting seed production and causing varying levels of mortality in some or all standing plant stages. These effects varied between the species and were modelled using stage-specific multipliers for the relevant vital rates (Fordham et al. 2012b). Wildfires through their interaction with food availability impacted *T. adelaidensis* body condition (Fenner and Bull 2007) and thus reproductive output. To account for this we modelled a 10%

reduction in fecundity in years following a wildfire.

Each habitat patch had a probability of ignition, which increased from a minimum value in the first year after fire to a maximum value ten years after fire according to the function $P(I) = 0.09/(1+120e^{-tsf})$, where $P(I)$ is the probability of ignition and tsf is the time in years since the last fire (McCarthy et al. 2001). This produced a mean fire return interval over 100 years of approximately 12 years, which overlaps with an independent estimate (Bradstock and Kenny 2003) for south-east Australia (where a number of the case study species occur). To maintain model structural simplicity, it was assumed that fires burnt entire patches (i.e. no fire heterogeneity within patches). Years that individual patches ignited were correlated according to their spatial proximity using the same correlation function given above for environmental stochasticity. Thus, multiple populations burnt in the same year determined by their respective times since last fire and spatial proximity.

Droughts have a strong negative impact on the vital rates of *T. rugosa* (in drought years fecundity can be reduced by as much as 100%; C. Michael Bull unpublished data) and are likely to have a similar impact on *T. adalaidensis*. Using long-term census data (1994 – 2007) for a subpopulation of *T. adalaidensis*, we found that on average drought cause a 57% reduction in abundance, occurring at a probability of twice in every ten years (Fordham et al. 2012c).

Density dependence

Density dependence for plant species was modelled in such a way that vital rates were reduced whenever the density of the population exceeded a ceiling threshold (Keith *et al.* 2008). Ceiling thresholds for each species were based on the mean area occupied by the canopy of a plant in the largest stage when unaffected by neighbours (maximum canopy size). During simulations, the space used by the population in each patch was calculated by summing the number of individuals in each stage, multiplying these numbers by a stage-specific weight, and summing across all stages. For each time step, where the space used exceeded the ceiling threshold, rates of survival, growth and fecundity were reduced by a user-defined (Fordham et al. 2012b).

Density dependence was implemented using a Scramble model (Logistic or Ricker type of density dependence) for *L. timidus* and *O. cuniculus* which determined the population growth rate at each time step (by modifying fecundity and leveret survival) as a function of the population size at that time step. Maximum rate of population growth (R_{max}) was calculated by fitting exponential and logistic models to time series abundance data from the Global Population Dynamics Database (<http://www3.imperial.ac.uk/cpb/research/patternsandprocesses/gpdd>). AIC model averaging was used to provide a weighted estimate of R_{max} . This procedure resulted in an R_{max} estimate of 1.34 for *L. timidus* and 1.5 for *O. cuniculus*. We also considered *O. cuniculus* R_{max} to vary spatially according to bioclimatic regions (Hone 1999).

The density dependence sub-model for *T. adelaidensis* was based on the Beverton-Holt equation, which models a process of diminishing returns that leads to contest competition. The primary factor limiting population growth for *T. adelaidensis* is availability of appropriate burrows (Fellows et al. 2009, Souter et al. 2004). Neonates normally disperse from their natal burrows within three – ten weeks after birth to secure an independent burrow (Milne 1999). Individuals that do not find a suitable burrow are likely either to die because of inadequate refuge conditions, or to have a higher chance of predation because they are exposed to predators for longer, or to disperse away from the population site. Souter et al. (2004) showed that addition of artificial burrows led to a significant rise in the number of neonates retained within plots in a subpopulation, indicating that competition for burrows is greatest amongst 0- 12 month individuals. Thus fecundity (a parameter that includes survival to 12 months) was treated as being regulated by density dependence. We used long-term density estimates for six relatively similar sized Australian lizards (*Ctenotus regius*, *C. leonhardii*, *C. schomburgkii*, *Diplodactylus conspicillatus*, *Rhynchoedura ornate* and *Lucasium stenodactylum*) that inhabit similar environmental conditions to *T. adelaidensis* (Read et al. 2012). By fitting a Ricker type density dependence equation to these time series we estimate R_{max} to be **2.56** (upper and lower 95% confidence intervals: 2.14 and 2.97)

Dispersal

The probability of seed dispersal between patches of suitable habitat and, animals moving between patches, during each time step was modelled with an exponential function, $P = a \cdot \exp(-D^c/b)$, where D is the distance between patch centroids and a , b and c are constants. When D exceeds a specified maximum distance (D_{\max}), P is set to zero. Mean (b) and maximum dispersal rates (D_{\max}) for each species are provided in the table above.

Carrying Capacity and Initial Abundance

Carrying capacity of a habitat patch (potential subpopulation) was modelled as a function of total (summed) habitat suitability; and abundance was initially set at 80% of carrying capacity. Specific functions used to calculate carrying capacity are provided by: (i) Fordham *et al.* 2012b for the three plants; (ii) Anderson *et al.* 2009 for *L. timidus*; (iii) Fordham *et al.* 2012a for *O. cuniculus*; and (iii) Fordham *et al.* 2012c for *T. adelaidensis*.

Simulation

Species' spatial abundance patterns and range limits were driven by demographic processes, climate change and, for some species catastrophic events (wildfires and droughts), and the interaction between these. All simulations were based on 1 000 stochastic replicates and run over a 81year period (i.e., 2000 - 2080). Depending on the case study, we provide the following indices:

1. The smallest population size that is expected to occur between 2020 and 2080 (Expected Minimum Abundance; McCarthy and Thompson 2001), which is a continuous metric reflecting risks of both declines and extinction risk.
2. Proportional changes in mean population abundance between 2020 and 2080
3. Mean population abundance and metapopulation occupancy between 2020 to 2080
4. Range movement between 2020 and 2080 based on a weighted mean of the latitudes of the most northern 10% of the metapopulation. Weights were the average population abundance of each

patch in each year, and latitude was taken from the geographic centre of the patch (Anderson et al. 2009)

We use the correlative SDM maps of habitat suitability, with a threshold applied to distinguish between potentially occupied and unoccupied sites (see above), to calculate annual change in range area between 2020 and 2080 (Buisson et al. 2010).

References

- Akcakaya, H. R. 2002. Estimating the variance of survival rates and fecundities. - *Anim. Conserv.* 5: 333-336.
- Akcakaya, H. R. 2005. RAMAS GIS: linking spatial data with population viability analysis, Version 5. - *Applied Biomathematics*
- Anderson, B. J., et al. 2009. Dynamics of range margins for metapopulations under climate change. - *Proc. R. Soc. Lond. Ser. B-Biol. Sci.* 276: 1415-1420.
- Bradstock, R. A. and Kenny, B. J. 2003. An application of plant functional types to fire management in a conservation reserve in south-eastern Australia. - *J. Veg. Sci.* 14: 345-354.
- Buisson, L., et al. 2010. Uncertainty in ensemble forecasting of species distribution. - *Glob. Change Biol.* 16: 1145-1157.
- Bull, C. M. 1995. Population ecology of the sleepy lizard, *Tiliqua-rugosa*, at Mt-Mary, South Australia. - *Australian Journal of Ecology* 20: 393-402.
- Diniz-Filho, J. A., et al. 2009. Partitioning and mapping uncertainties in ensembles of forecasts of species turnover under climate changes. - *Ecography* 32: 1-10.
- Dunham, A. E., et al. 2006. Using scalar models for precautionary assessments of threatened species. - *Conserv. Biol.* 20: 1499-1506.
- Elith, J., et al. 2008. A working guide to boosted regression trees. - *J. Anim. Ecol.* 77: 802-813.
- Fellows, H. L., et al. 2009. Spiders provide important resources for an endangered lizard. - *J. Zool.* 279: 156-163.

- Fenner, A. L. and Bull, C. M. 2007. Short-term impact of grassland fire on the endangered pygmy bluetongue lizard. - *J. Zool.* 272: 444-450.
- Fordham, D. A., et al. 2012a. Modelling range shifts for invasive vertebrates in response to climate change. - In: Brodie, J., Post, E. and Doak, D. (eds.), *Wildlife Conservation in a Changing Climate* University of Chicago Press, pp. Chapter 5. in press (accepted).
- Fordham, D. A., et al. 2012b. Plant extinction risk under climate change: are forecast range shifts alone a good indicator of species vulnerability to global warming? - *Glob. Change Biol.* 18: 1357–1371.
- Fordham, D. A., et al. 2012c. Managed relocation as an adaptation strategy for mitigating climate change threats to the persistence of an endangered lizard. - *Glob. Change Biol.* 18: 2743–2755.
- Fordham, D. A., et al. 2012d. Strengthening forecasts of climate change impacts with multi-model ensemble averaged projections using MAGICC/SCENGEN 5.3. - *Ecography* 35: 4-8.
- Hastie, T. J. 1992. Generalized additive models. - In: Chambers, J. M. and Hastie, T. J. (eds.), Chapter 7 of *Statistical Models in S* Wadsworth & Brooks/Cole.
- Hone, J. 1999. On rate of increase (r): patterns of variation in Australian mammals and the implications for wildlife management. - *Journal of Applied Ecology* 36: 709-718.
- Iason, G. R. 1989. Growth and mortality in mountain hares - the effect of sex and date of birth. - *Oecologia* 81: 540-546.
- McCarthy, M. and Thompson, C. 2001. Expected minimum population size as a measure of threat. - *Anim. Conserv.* 4: 351-355.
- McCarthy, M. A., et al. 2001. Theoretical fire-interval distributions. - *International Journal of Wildland Fire*: 73-77.
- Milne, T. 1999. Conservation and ecology of the endangered pygmy bluetongue lizard *Tiliqua adelaidensis*. - In, Flinders University PhD Thesis.
- Phillips, S. J., et al. 2006. Maximum entropy modeling of species geographic distributions. - *Ecol. Model.* 190: 231-259.

Read, J. L., et al. 2012. Booming during a bust. Asynchronous populations responses of arid zone lizards to climate variables. - *Acta Oecologia* 40: 51–61.

Souter, N. J., et al. 2004. Adding burrows to enhance a population of the endangered pygmy blue tongue