**Appendix S1: Online Extended Methods**

We used spatially explicit demographic models to simulate the metapopulation dynamics of *Chelodina rugosa* (northern snake-necked turtle) in two neighbouring water catchments in northern Australia: the Blythe and Liverpool river catchments. The region was chosen because the population dynamics of *C. rugosa* in these catchments is well established [[1-7](#_ENREF_1)].

*Survey data*

We surveyed 80 waterholes repeatedly over an eight year period (1999-2007) to determine: drying frequency, duration of inundation, extent of pig rooting, vegetation composition and rates of harvest carried out by the indigenous people. Turtle occurrence was assessed in the year 2000, between May and September, using hoop traps baited with fish [[8](#_ENREF_8)]. We used 10-15 traps (exact number depended on the size of the waterhole) which we re-baited and moved once over a 4 night trapping period. Traps were evenly spread throughout the waterhole to depths of 1.2 metres. Drying frequency (DRYFREQ) was averaged across years and classed as either: drying frequently (≥ 20% of years, typically drying every other year: DRYFREQ = 0.5), rarely drying (< 20 % of years, typically drying every 6 years: DRYFREQ = 0.166) or no drying. Period of inundation (hydroperiod) was calculated by recording the month of drying in each year. The average waterhole hydroperiod was classed as either: (i) too short to support *C. rugosa* or *Eleocharis dulcis* (water chestnut plant) (EPHEM = 1: dries before August 1) or sufficient to support these species (EPHEM = 0: dries on or after August 1). The extent of pig rooting at the end of the dry season (late October/ early November) was assessed both visually, and using direct measures described by Fordham *et al.* [[3](#_ENREF_3)]. Pig rooting intensity (PIGROOT) was classed as being high (PIGROOT = 1) if on average ≥ 20 % of the surface area of the waterhole was impacted by pig rooting in a given year; and low (PIGROOT = 0) if < 20 % of the area was impacted. The vegetation composition was surveyed and the area of the waterhole comprising *E. dulcis* was estimated and assigned one of two categories: high-to-medium (ELEOCHARIS = 1: ≥ 10 %), or low (ELEOCHARIS = 0: < 10%). An identical area-based assessment was made for *Melaleuca sp.* Harvest frequency (HARVFREQ) was assessed using direct observations and surveys (see [[7](#_ENREF_7)]).

*Environmental spatial data*

Wetland features were mapped using high resolution scans of 1:50,000 topographic maps produced by the Army Topographic Support Establishment (Department of Defence, Commonwealth of Australia) in 1999. Specifically, twenty four maps covering the Blyth and Liverpool catchments downstream of the Arnhem Land sandstone escarpment were scanned, geo-referenced and wetland features were digitised using ArcGIS 10. This produced a polygon map of waterhole features and their associated attributes: (i) VEG: vegetation density (1 = dense, 0 = sparse or absent); (ii) AREA: wetland area (ha); (iii) PERVEG: percent area of the waterhole that is vegetated; (iv) PERMANENT: whether the waterhole is inundated even during unusually dry years (1 = permanent, or 0 = ephemeral); and (v) SALTWATER: salinity content (1 = salt water, or 0 = fresh water). Wetland features were digitised at a 1:5,000 – 1:7,500 scale. A horizontal accuracy of 90% within ± 25m exists for features on the original topographic maps. We did not include the sandstone catchment area because *Chelodina burrungandjii*, not *C. rugosa*, is found in this region [[9](#_ENREF_9)].

Digital spatial layers describing major road networks, water courses and towns and outstations (satellite communities) were accessed from GeoScience Australia (GEODATA TOPO 250K Series 3 - Online [via MapConnect]; ANZCW0703008969). The layers were used to calculate distances from each wetland feature to the nearest road (DISTROAD), river (DISTRIVER), and outstation (or town; DISTTOWN) using the *Near* function in Spatial Analyst ArcGIS v10. The National Vegetation Information System broadly describes the present-day vegetation of the Northern Territory of Australia [[10](#_ENREF_10)] and was used to distinguish floodplain (SAVANNA = 0) and savannah (SAVANNA = 1) vegetation communities.

Elevation was based on a 30m Digital Terrain Model derived from the NASA SRTM project [[11](#_ENREF_11)]. From this, slope was calculated using the ‘DEM Surface Tools’ ArcGIS extension [[12](#_ENREF_12)]. A map of plant available water capacity (PAWC, a measure of soil water available to plants at 250 x 250m grid-cell resolution) was accessed from the Australian Soil Resource Information System (http://www.asris.csiro.au). National soil data was provided by the Australian Collaborative Land Evaluation Program ACLEP (www.clw.csiro.au/aclep). All spatial layers are available from the authors on request.

*Predictions of abundance and carrying capacity*

Three features of waterholes were previously identified as key determinants of occurrence, abundance and vital rates for *C. rugosa*: the density of *E. dulcis* at a waterhole, the drying frequency, and the period of inundation [[2](#_ENREF_2), [3](#_ENREF_3)]. We used binary logistic regression to model the relationship between mapped environmental variables and: (i) ELEOCHARIS: presence of *E. dulcis* (observed density ≥ 10 % or < 10 %); and (ii) EPHEM: whether waterholes dry or not before August (a hydroperiod too short to support viable populations of *C. rugosa*). We used ordinary linear regression to model the relationship between mapped environmental variables and drying frequency (DRYFREQ; described above) (we compared logit-transformed and untransformed DRYFREQ models and found that modelling untransformed DRYFREQ as the response variable resulted in a slightly lower cross-validation error). Models were built using survey information from 50 ephemeral waterholes within the study region. We hypothesised *a priori* that these waterhole features are affected by elevation, slope, waterhole area (AREA), PAWC, being in a floodplain or savanna environment (SAVANNA), and vegetation density (VEG; see above for a description of spatial features). The final predictor set was selected by first eliminating highly correlated variables from the predictor set (one variable was retained for each pair with Pearson correlation > 0.75) and subsequently selecting the best-performing combination of predictor variables using stepwise AICc (Akaike Information Criterion, with a correction for limited sample size [[13](#_ENREF_13)]). We calculated proportion of deviance explained (hereafter R2 [[14](#_ENREF_14)]) as a summary statistic to describe structural goodness-of-fit [[15](#_ENREF_15)].

We assessed the predictive performance of the selected models using leave-one-out cross validation and subsequently computing area under the receiver operating curve (AUC) [[16](#_ENREF_16)]. The best models for ephemerality (logit(EPHEM)= -0.17 - 9.4E-6\*[AREA] + 0.18\*[SAVANNA] – 18.7\*[VEG], R2 = 0.35) and drying frequency (DRYFREQ = 0.4 + 0.0028\*PERVEG – 0.23\*[SAVANNA]; R2 = 0.53) exhibited adequate model performance (AUC > 0.8) [[17](#_ENREF_17)]. However, cross validation for *E. dulcis* presence (i.e., ELEOCHARIS ≥ 10%) suggested poor predictive performance (AUC < 0.7). Therefore, we developed a simple set of rules based on field observations to predict *E. dulcis* presence in the Blythe and Liverpool catchments. We classified a waterhole as having *E. dulcis* if it dries regularly (> 20 % of years; DRYFREQ = 0.5) , but not before August [[3](#_ENREF_3)] (EPHEM = 0); and if it is not a marine swamp, or if it is not a sparsely vegetated waterhole (VEG = 0) in a savanna habitat (SAVANNA = 1).

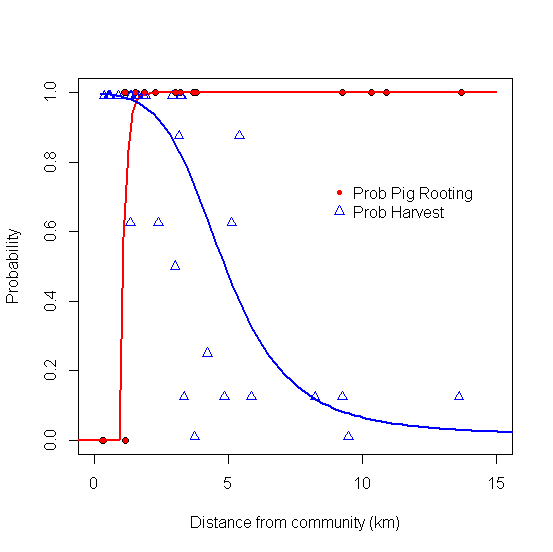
We used the spatial predictions of waterhole features (generated as described above) and area of the billabong (number of 50m x 50m pixels) to predict the initial abundance and carrying capacity for each of the 1,013 waterholes in the study region. We used published [[5](#_ENREF_5)] and unpublished capture-mark-recapture (CMR) estimates of abundance at waterholes differing in drying frequency and timing, density of *E. dulcis* and pig predation rates to calculate average abundance at waterholes. We set abundance as follows: (i) rarely drying waterholes (dries ≤ 20 % of years) with no *E. dulcis* = 5.5/ha; (ii) frequently drying waterholes (dries > 20 % of years) with *E. dulcis* = 9.5/ha; (iii) frequently drying waterholes with no *E. dulcis* = 5.5/ha; (iv) waterholes that dry annually before August = 0/ha; and (v) permanent waterholes = 3/ha. Predicted abundance was verified using an independent set of 10 waterholes with CMR estimates of average abundance. With the exception of one site, all model estimates agreed with CMR estimates of abundance (i.e., fell within the confidence bounds surrounding abundance). A subset of waterholes, with predicted turtle abundance = 0, were verified using occurrence survey data.

The size distribution of *C. rugosa* is strongly influenced by drying frequency and pig predation [[3](#_ENREF_3)]. Populations in waterholes that dry infrequently experience low levels of recruitment because densities are close to carrying capacity (K) [[5](#_ENREF_5)]. Populations in waterholes that dry frequently with *E. dulcis* tend to experience the highest levels of predation from pigs or harvested by humans, resulting in elevated recruitment and populations being maintained below K [[3](#_ENREF_3)]. We used these observations to estimate K for each waterhole: (i) rarely drying waterholes with no *E. dulcis* = 5.5/ha (i.e., K = initial abundance); (ii) frequently drying waterholes with *E. dulcis* = 14.25/ha (i.e., K = 1.5 \* initial abundance); (iii) frequently drying waterholes with no *E. dulcis* = 6.6/ha (i.e., K = 1.2 \* initial abundance); and (iv) permanent waterholes = 3/ha. Waterholes with K < 10 animals were treated as non-viable populations (i.e., K was set to zero) in the demographic model.

*Human mediated effects: pig predation and harvest rates*

We hypothesized that harvest frequency (measured at 32 waterholes) and pig visitation rates (measured at 50 waterholes) were strongly influenced by the distance to the nearest outstation. We fitted models using maximum likelihood, assuming a Gaussian error distribution for logit-transformed harvest rates and a binomial error distribution for pig predation impact (measured as high or low). Model selection was performed using AICc. Heavy pig visitation only occurred in wetlands with high *E. dulcis* abundance (*E. dulcis*, not *C. rugosa*, is the primary target for pigs [[2](#_ENREF_2)]), thus the relationship between pig visitation and outstation distance was modeled only for those surveyed sites with a high abundance of *E. dulcis* (*n =* 22).

Harvest frequency decreased with distance from the nearest outstation, and was modelled using four plausible functional forms: linear, exponential decay, logit-linear, and logit-exponential. We found the logit-exponential function to best describe the relationship between harvest and distance (logit(HARVFREQ) = 10.6\*exp(-[DISTTOWN]/5.57) – 4.43). Feral pig impact (PIGROOT) increased with distance from the nearest outstation, and was modelled using two plausible functional relationships: logit-linear or a "broken stick" logit-linear model (in which the relationship assumes different logit-linear slopes on either side of a threshold value). The modelled relationship between distance from village and probability of pig rooting and harvest was best described by the broken stick model (< 1.07 km: PIGROOT = 0, ≥ 1.07 km: logit(PIGROOT) = -0.38 + 9.28\*[DISTTOWN]).



*Fig. A1-1. Probability of extensive pig visitation (red) and probability of human exploitation (blue) at occupied waterholes as a function of the distance to the nearest outstation. Note that pig visitation is only displayed for waterholes with high abundance of E. dulcis.*

*Demographic model*

We constructed spatially explicit stage- and sex-based stochastic matrix models for *C. rugosa* in RAMAS GIS v5 [[18](#_ENREF_18)]. The model consisted of 4 female stages and 2 male stages: (i) Female S1 = < 140mm; (ii) Female S2 = 140-180mm; (iii) Female S3 = > 180-220mm; (iv) Female S4 = > 220mm; (v) Male S5 < 140mm; (vi) Male S6 > 140mm [[4](#_ENREF_4)]. Previously, Fordham *et al.* [[4](#_ENREF_4)] constructed three baseline *C. rugosa* stage matrices describing vital rates as a function of drying, the density of *E. dulcis* and the level of pig predation. Each waterhole (with *C. rugosa* abundance > 0) was assigned one of these baseline stage matrices: (i) rarely drying waterholes = matrix 1; (ii) frequently drying waterholes with *E. dulcis* = matrix 2; (iii) frequently drying waterholes with no *E. dulcis* = matrix 1; and (iv) permanent waterholes = matrix 3 (online Appendix S3). Frequently drying waterholes with *E. dulcis* and high pig rooting occurring in < 50 % years (because they are closely located to human settlements) were assigned matrix 1.

In years of high rainfall, frequently drying waterholes do not dry [[3](#_ENREF_3)]. This was simulated using the catastrophe function of RAMAS, whereby drying frequency was used to determine the probability of a wet year (based on Fordham *et al.* [[4](#_ENREF_4)]) and a local multiplier was used to account for the positive effect not drying in a given year can have on abundance. Harvesting of *C. rugosa* for indigenous consumption occurs in dry years at waterholes with *E. dulcis* that are close to outstations (< 15 km away) and where pig abundance is low. We simulated harvesting using a second catastrophe function, whereby the local probability of a harvest event was defined by the probability of waterhole drying multiplied by the probability that the waterhole will be harvested (i.e., based on distance from the nearest outstation). In savanna environments, we constrained the probability of harvest to waterholes with *E. dulcis*, where pig abundance is predicted to be low. In these waterholes harvest caused a 20% decline in stage classes > 140 mm [[4](#_ENREF_4)]. In floodplain environments, the indigenous people will catch turtles even in the presence of high densities of pigs, but the harvest rates are much lower (Fordham unpublished data). We simulated harvesting in floodplain environments up to 15 km from outstation settlements causing 10% decline in stage classes > 140 mm.

We modelled environmental variation affecting *C. rugosa* survival using CMR data [[3](#_ENREF_3)] analysed in Program Mark with sampling variation removed [[19](#_ENREF_19)]. We set the CV for survival in years when waterholes: (i) do not dry to 0.06 (SD = 0.057); (ii) dry with low levels of pig rooting (≤ 20%) to 0.04 (SD = 0.031); and (iii) dry with high levels of pig rooting (> 20%) to 0.15 (SD = 0.089). Underwater nesting, multiple clutching and embryonic diapause buffers *C. rugosa* fecundity against environmental variation [[1](#_ENREF_1)]. We modelled hatchling survival as density dependent. Density manipulation experiments (removal and supplementation) have shown that *C. rugosa* hatchling survival is strongly influenced by the density of turtles ≥ 140 mm [[5](#_ENREF_5)]. Previously we modelled this relationship using a density decay function [[4](#_ENREF_4)]. Here we use a similar approach through a function that multiplies maximum fecundities by the survival of hatchlings calculated as:

Φ0 = 0.9385 \* exp(-3.88 \* (N/K) ) Equation S1-1

Where N = population size, and K is the carrying capacity of animals > 140 mm.

*Dispersal models*

Dispersal was modelled as the proportion of individuals ≥ 140 mm [[7](#_ENREF_7)] dispersing between each pair of defined populations during the wet season (since dispersal does not occur at other times of the year [[2](#_ENREF_2)]). We used CMR field data, combined with expert knowledge, to model *C. rugosa* dispersal. Specifically, for any pair of populations, dispersal frequency was computed as the fraction of the total marked population at the source population that was recaptured at the target population. These observed dispersal rates were used to calibrate three different models for estimating dispersal rates between pairs of waterholes.

*1. Dispersal kernel – null model*: Dispersal was modeled as a function of centre-to-edge distance using a simple distance-based dispersal kernel, extrapolated from field-based estimates of average and maximum *C. rugosa* movement. Centre-to-edge was selected over the edge-to-edge method to soften the "drainage effect" by which very large populations are drained into smaller satellite populations nearby [[20](#_ENREF_20)]. We used the dispersal-distance function,

*a* \* exp(−Dij*c*/*b*) = 0.2 exp(−Dij2.4/1.8) (Equation S1-2)

where D is distance from the centre of the source population to the closest edge of the target population, with a maximum recorded dispersal distance of Dmax = 4 km (above which dispersal rate = 0). The intercept (*a* = 0.2) approximates the highest rate of immigration recorded from a single population. The constant *b* was set to 1.8 and represents mean distance between a subset of connected populations used to estimate dispersal (i.e., based on field observations). The constant *c* was set iteratively so that < 5 % (2.8 %) of the population moves 1.8km in a single year based on CMR data, in accord with dispersal rates observed in the field.

*2. Friction-based – structural connectivity*: Dispersal was modelled using a least-cost pathway method that combined a friction map (cost-of-movement raster) with the dispersal kernel described above [[18](#_ENREF_18)], allowing dispersal to be modelled as a function of structural connectivity. We used raster algebra ("raster" package for R) to generate a GIS cost-of-movement ("friction") raster for the study region. Specifically, we used CMR data, observations from radiotelemetry, and expert knowledge to develop the following rules to represent cost-of-movement through a 50 m grid cell: (i) savanna habitat = 1; (ii) floodplain habitat = 0.5 (i.e., cost of moving across two floodplain cells = cost of moving across one savanna cell); (iii) freshwater wetland and stream habitat = 0.33; (iv) waterhole habitat = 0.25; and (v) salt water and dense forest habitat = 100 (i.e., cost of moving across 0.01 % of a salt water or dense forest cell = cost of moving across one savanna cell). In the dispersal kernel (Eq. S1-2), D now represents the cost-distance (cumulative cost of movement along the least-cost path) from the centre of the source population to the closest edge of the target population. In RAMAS GIS, the value 1 represents the baseline value of the cost-of-movement raster: if all grid cells in the cost-of-movement raster were set to 1, estimated dispersal rates would match the distance-based estimates derived from Eq. S1-2 [[18](#_ENREF_18)]. Since savanna grid cell values were set to 1, and most surveyed waterholes were located in savanna environments in the cost-of-movement raster, the remaining parameters of the dispersal kernel (Eq. S1-2) were not altered for this analysis..

3. *Individual-based model – functional connectivity*: We modelled the interaction between turtle movement and habitat using a spatially explicit individual-based model (SIBM) parameterised in HexSim, which is a flexible software framework for individual-based ecological modeling and risk assessment [[21](#_ENREF_21)]. Our mechanistic SIBM dispersal model was designed to assess dispersal rates for a single wet season (the period during which long-distance movements occur for *C. rugosa* [[2](#_ENREF_2)]). All waterholes were initialized at carrying capacity (as described above). All turtle movements in the SIBM model were guided by an "attraction" surface (a map composed of hexagonal grid cells) that formalized four movement rules, described below. These movement rules were developed according to field observations of *C. rugosa*, and in recognition of the general dependence of this species on freshwater habitats for foraging and dispersal.

*Rule 1*: Dispersal movements occur during the wet season, during periods when streams are flowing and much of the floodplain is inundated [[2](#_ENREF_2)]. We therefore assigned "attraction" values such that: (i) streams represented important movement conduits; (ii) turtle movement directions were biased downstream( high flow rates in rivers and streams during the wet season tend to carry individuals downstream [[22](#_ENREF_22)]); and (iii) increased movement rates in floodplain environments.

*Rule 2*: Turtles tend to exit waterholes on the downstream side of the waterhole and move in a downslope direction when dispersing in savanna environments, presumably to locate wetlands.

*Rule 3*: Turtles are repelled by salt water, dense forests and the sandstone escarpment, which serve as reflective boundaries.

*Rule 4*: Turtles are free to leave the water catchments [[6](#_ENREF_6)] - so the edges of the study region were modelled as absorbing boundaries. We did not consider immigration from outside the two water catchments.

The SIBM model was developed to better capture such functional elements of connectivity by including mechanisms that allowed the following biologically realistic properties to emerge: (i) turtles occupying waterholes with greater edge to area ratio are more likely to leave their home patch; (ii) turtles are more likely to successfully disperse if a site has more and larger neighbouring waterholes, and disperse further if neighbouring waterholes are closely linked by streams and rivers; and (iii) dispersing turtles are more likely to return to where they came from if neighbouring sites are mostly small and distant (*Chelodina sp.* show strong homing behaviours to favourable habitat [[23](#_ENREF_23)])*.* A detailed description of the SIBM model is provided in Appendix S2.

*Model Simulations*

The influence of dispersal (captured using *null*, *structural connectivity*, and *functional connectivity* models) on spatial abundance patterns and local range limits was simulated using 10,000 stochastic replicates, run over a 101 year period (i.e., 2000 - 2100). Population viability was assessed using expected minimum abundance (EMA), a continuous metric reflecting risks of both declines and extinction risk [[24](#_ENREF_24)], calculated for the period: 2010 –2100. Range movement between 2010 and 2100 was calculated annually based on the latitude of the geographic centre of the most southern sub-population [[25](#_ENREF_25)]. Change in the number of occupied waterholes and total population abundance (of persistent model runs) over time were also investigated.

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