Predicting Risk to Biodiversity as a Function of Aquifer Pressure in GAB Mound Springs

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Abstract: Plants and aquatic invertebrates endemic to mound springs of the Great Artesian Basin (GAB) of Australia are of national biodiversity significance. Springs occur in groups, and current survey data suggest that local extinction and recolonisation within a group does occur naturally. Mound spring fauna are therefore good examples of true metapopulations, and any study of the environmental impacts on these systems must take this into account. Use of GAB water leads to drawdown of the pressure in the aquifer. This can potentially impact mound spring flora and fauna in two ways. First, as pressure decreases, so does flow from spring vents, and this causes the area of each of the patches in the spring group to shrink. If local extinction probability of a patch increases with decreasing area, this will increase the likelihood of extinction of a species in the whole group. Second, when pressure drops far enough, entire vents may cease to flow, decreasing the number of patches in the system. This change can lead to increases in extinction risk of a species, because the nature of the patch system is changing qualitatively. We model this system as a 1st order Markov chain, and predict the risk of quasi-extinction under two different scenarios for modelling the effect of drawdown on wetland areas.

Keywords: Markov chain, Quasi-extinction risk, Artesian aquifer

1. INTRODUCTION

Peering into the future and trying to predict how uncertain processes are going to turn out is the business of forecasting. Mathematical models are a familiar and comfortable part of predicting the future in engineering and financial market analysis. This same capability is needed in ecology, especially in the area of environmental impact assessment. The tools are not as well developed, and must be custom designed for each application. Our research for the last two years has focussed on the development of ecological forecasting methods for Great Artesian Basin (GAB) mound springs. The pure science component of the project is constructing new models of metapopulation dynamics, and testing metapopulation theory against data on mound springs fauna. The applied component will contribute to operational management of mounds springs of the GAB. Western Mining Corporation (WMC) operate the Olympic Dam mine which is a significant user of water from the GAB. These models will contribute to WMC's ecological monitoring programs through improved design, and by forecasting the effect of water extraction impacts on mound springs fauna.

WMC and other users of the GAB need tools for forecasting the ecological effects of water extraction, especially on mound springs fauna. Critics of WMC operations make statements such as "a 20% loss of wetland results in a 70% loss of snails" [Mudd, 2000] in public journals and Internet forums. The dynamics of ecological populations through long periods of time often respond in non-intuitive ways to changes in the environment. To analyse such statements, or any other statements about extinction risk, we need to use quantitative models to explicitly predict risks. This does not eliminate subjective components to risk estimation, but it does make the process transparent and explicit [Burgman, 1993]. In the remainder of this introduction we consider the impact of draw down on the metapopulation dynamics of a hydrobiid snail species, and then we introduce the model in the methods section.

Fonschochlea accepta is a small aquatic species of hydrobiid snail that is obligate to a single complex of springs associated with the GAB. In 1995, *Fonscochlea accepta* was present in 9 out of 13 spring vents in the Sulphuric Springs group at Hermit Hill (Figure 1).
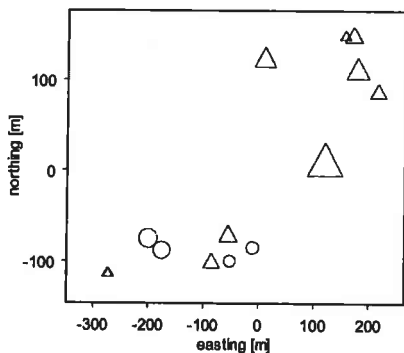
Figure 1 A map of sulphuric springs with symbol size proportional to wetland area in 1995, and wetlands occupied by *F. accepta* indicated with triangles.

The extraction of water from the GAB leads to a reduction in aquifer pressure. If the draw down of aquifer pressure occurs near spring vents it can have two effects. First, it can reduce the flow rate, and this leads to a reduction in the area of the wetland associated with the spring. Second, it can cause a spring to cease flowing altogether. A reduction in area is expected to reduce the possible maximum size of the snail population in that wetland, increasing the probability that it will become extinct in that one spring. The loss of an entire spring wetland also affects the ability of snails to move between groups, because the missing wetland would have contributed colonising snails to neighbouring wetlands, and potentially acted as a stepping-stone between more distant wetlands.

2. METHODS

2.1 Model Structure

We set the problem up as a spatially explicit metapopulation model similar to previous patch occupancy models [e.g. Hanski, 1994; Day and Possingham, 1995], with each patch being either unoccupied or occupied by *F. accepta* (Figure 1). We assume that the internal population dynamics of each patch occur on a relatively rapid time scale, so the main processes in the model are the local extinction of occupied patches, and the recolonisation of unoccupied patches from other occupied patches in the system. When all patches are locally extinct, no further recolonisation is possible, and the system is globally extinct. The state of the model is the occupancy pattern of the metapopulation, e.g. patches 3 and 4 are occupied and all other patches are unoccupied. We model this using a first order Markov chain, because we assume that the transition prob-

abilities depend only on the state of the metapopulation in the previous time step.

We assume that local extinction rates (u_i) decrease with wetland area A_i according to the formula

$$\mu_i = e / A_i^x \qquad (1)$$

where e and x are parameters to be estimated from the data.

Colonisation probabilities ($p_{i,j}$) depend on the state of the system. An occupied patch i contributes successful colonists to an empty patch j with probability

$$p_{i,j} = e^{-\alpha d} \qquad (2)$$

where d is the Euclidean distance in metres between the vents, and α (units of m^{-1}) is the rate at which colonisation declines with distance. When $d = 0$ the probability of colonisation is 1. The probability that an empty patch i receives at least one colonist is:

$$\lambda_j = 1 - \prod_{i=1, i \neq j}^{n} \left(1 - p_{i,j}\right) o_i \qquad (3)$$

where o_i is 0 if patch i is unoccupied and 1 if it is occupied. We assume that the number of colonists arriving does not affect the probability of colonisation or local extinction.. This assumption is reasonable because the number of colonists is likely to be orders of magnitude less than the numbers of individuals in a fully occupied patch.

2.2 Parameter Estimation

We assume that a patch with an area less than 1 m^2 has a local extinction probability of one, thus $e = 1$. Given the range of variation in A_i is large, and some large patches are not occupied by *F. accepta*, it seems reasonable to suppose also that x is less than one, so that extinction rate decreases less than linearly with increasing area. We presume x to be larger than 0.1, because this leads to a high local extinction rate (>0.9) across the full range of patch sizes. Likewise, if x is bigger than 0.5, then large patches (~5000 m^2) are almost immune to extinction.

When $\alpha = 0.015$ the probability of colonisation is less than 0.05 if the distance is greater than 200 m. If $\alpha > 0.05$ then the probability of colonisation beyond 100 m is <0.01, while if $\alpha < 0.01$ the probability of colonisation beyond 500 m is <0.01.

We set the expected value of the parameters equal to those estimated using the incidence function approach for this species in a different spring group $\alpha = 0.015$ and $x = 0.34$ [Tyre et al., 1999]. Because the

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incidence function method yields a "psuedo-likelihood" measure, it is not clear that we could use that method to formally derive distributions for the uncertainty in the parameters. We are currently exploring the use of Markov Chain Monte Carlo methods to formalise this step.
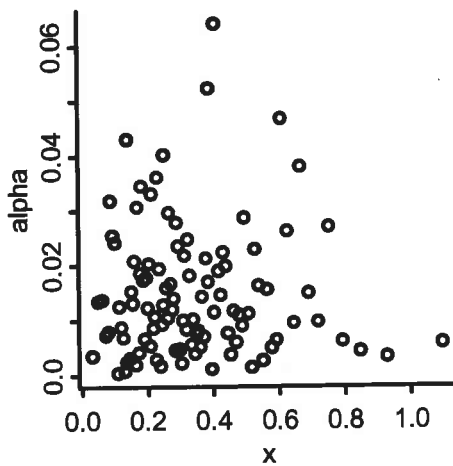

Figure 2 Distribution of parameter values from the latin hypercube. α (alpha) ~ gamma(1.5, 100) with an expected value of 0.015, and x ~ gamma(2.72, 8) with an expected value of 0.34.

2.3 Sensitivity Analysis and Scenarios

We incorporate the uncertainty about the parameter values into the model predictions with a Latin Hypercube sensitivity analysis [Blower and Dowlatabadi, 1994; Sanchez and Blower, 1997; Rushton et al., 2000a; Rushton et al., 2000b]. Uncertainty in each parameter is represented with a gamma distribution with parameters a and b. We chose a and b so that the expected value of each distribution matched that from the parameter estimation, while the 2.5% and 97.5% quantiles matched the ranges established above. Each of these distributions is then divided into 100 equi-probable intervals. A replicate run of the model is conducted by choosing an interval at random without replacement from each of these lists. Thus, we run the model with 100 different parameter combinations where each sub-range of each variable is used only once (Figure 2). The range was chosen to give a reasonable spread of values following the reasoning given above.

In the absence of other information about how pressure relates to wetland area, we have assumed that a reduction in pressure will lead to a linear reduction in wetland area for all springs of equal elevation and at the same location along the aquifer pressure gradient. There is a linear relationship between wetland

area and spring flow [Fatchen, 2001]. We further assume that if the pressure surface is below the height of the spring vent, then the wetland area will be zero, i.e. the spring will be extinct. Thus, if pressure drops to 60% of the previous pressure the area of a spring will shrink to 60% of the initial area. We consider two scenarios for how draw down of aquifer pressure affects wetland area. The observed variation in wetland areas for each vent is created by a number of factors including variation in the underlying geology, particularly characteristics of the conduit between the aquifer and the surface, and small differences in elevation. These details are unknown at present, and the two scenarios each assume that either there are no geological differences between springs, or that there are no elevation differences between springs. Both are obviously false assumptions, but we believe that they bound the range of possible outcomes.

First, if variation in wetland area is created only through geological differences, then all springs in the group are at the same elevation, and have the same pressure underneath them. Variation in elevation among vents at Sulphuric Springs is less than 2 metres, and although there is a gradient of decreasing pressure from north-west to south-east across the group it is likely to be very small compared to the effects of draw down. We call this the "geological differences" scenario, and as draw down only reduces wetland areas all vents cease to flow simultaneously when aquifer pressure reaches 0% of initial levels.

Second, if variation in wetland areas is due only to differences in elevation, then the smallest spring wetlands are at the highest elevation, and will cease to flow first. When draw down reaches 60% of 1995 levels then only the largest wetland in the group remains. We call this the "elevation differences" scenario.

We ran the model 10,000 times at each of the 100 different parameter combinations in two different scenarios at 40 different levels of draw down to estimate the probability that the number of occupied springs drops to one or less over a 50 year period (quasi-extinction risk). The initial state for all runs corresponded to the distribution of snails among vents observed in 1995.

3. RESULTS

The average quasi-extinction risk over fifty years increases with decreasing aquifer pressure in both scenarios (Figure 3, Figure 4). In the "elevation differences" scenario the increase is roughly linear, with a brief acceleration between 85% and 75% draw down, corresponding to the cessation of flow

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of several springs (Figure 3). Once aquifer pressure reaches 60% of its original level, only the largest patch in the system remains, and the risk of quasi-extinction over fifty years rises sharply to one, by definition. However, given the degree of uncertainty in the parameters, the 10th and 90th percentile risks are 0 and 1 except at the very lowest amounts of draw down.

In the "geological differences" scenario the average quasi-extinction risk over fifty years still rises linearly, but at a much lower rate (Figure 4). This linear increase persists down to very low aquifer pressures on the order of 20% of the original (curve not shown). The lower bound on risk is still zero, while the upper bound increases from 0.6 to 0.8.

The total area of wetland available over all vents decreases much more in the "elevation differences" scenario than in the "geological differences" scenario, but this is not the only reason why the mean risk rises faster (Figure 5). If total area was all that mattered, then the risk would increase at the same rate with decreasing total area in both scenarios. However, the risk rises much faster when spring vents cease to flow altogether, because this also reduces the ability of the snails to colonise empty patches.
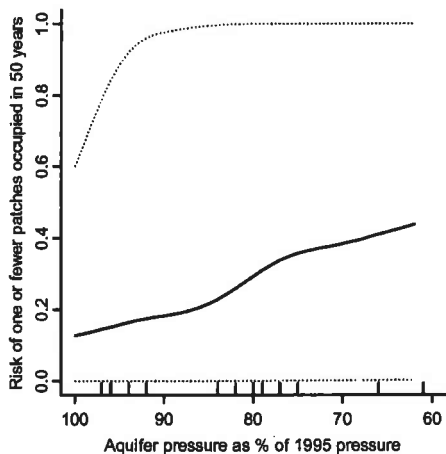
clude clear statements of risk for spring fauna. The 10th and 90th percentile predicted quasi-extinction risks range from zero to one – which is tautological. Thus the continued development of statistical methods to extract rates of colonisation and local extinction) from spring survey data is warranted. In addition, it would be helpful if future biodiversity monitoring was designed with the goal of estimating these rates.
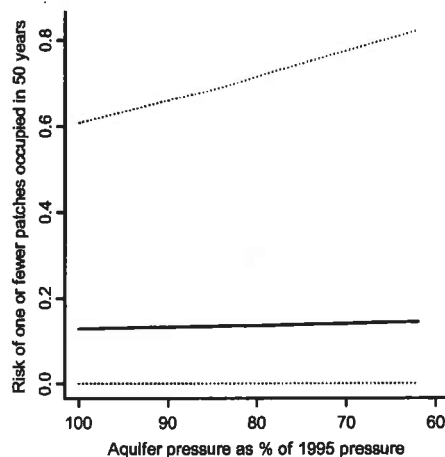

Figure 4 Average quasi-extinction risks over fifty years for the "geological differences" scenario. Lines are as for Figure 3.
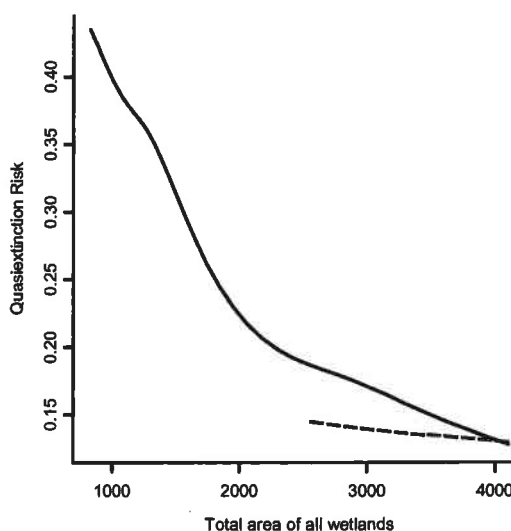

Figure 3 Average quasi-extinction risk over fifty years for "elevation differences" scenario. Vertical bars at the bottom of the plot indicate where individual vents cease to flow, with smallest vents disappearing first. Dotted lines indicate 10th and 90th percentile risks from the sensitivity analysis.

Figure 5 Average quasi-extinction risk over fifty years as a function of the total area of all wetlands in m^2. "elevation differences" scenario is the solid line, and "geological differences" scenario is the dashed line. Risk increases much faster when individual springs cease to flow than when only the area decreases.

4. DISCUSSION

The most important conclusion from this work is that the current levels of parameter uncertainty pre-

The second important result from this work is that the processes leading to variation in wetland areas between spring vents critically affect the response of quasi-extinction risk to draw down of aquifer pressure. Existing hydrogeology models for the region make predictions at a large spatial scale and are not designed to explicitly predict the loss of flow in a specific spring [Kinhill Engineers 1997]. At this stage we do not have information about spring elevation for this spring group. The most plausible assumption is that differences in elevation are a relatively small contribution, and so the "geological differences" scenario is possibly a good description.

Another possibility is to ignore the hydrogeology altogether, and base expected risk estimates on trends in observed wetland areas. Wetland area has been monitored independently of flow and biodiversity on at least a five-year basis since the inception of the Olympic Dam project. It is also possible to use aerial photography to get remotely sensed measurements of vegetated wetland area [Niejalke et al., 2001]. Thus we could use the trends in wetland area with time to extrapolate future patterns of reduction in area. With sufficient data for individual spring vents it may also indicate which vents are likely to cease flowing first.

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