Population Viability Analysis for Squirrel Gliders in the Thurgoona and Albury Ranges Region of New South Wales

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Prepared for: Albury Conservation Company

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EXECUTIVE SUMMARY

The Squirrel Glider is a small arboreal mammal that is threatened with extinction in NSW. On the south-west slopes of NSW, it occurs in dry woodland that has been heavily cleared for agriculture, and more recently for urbanisation. The Thurgoona and Albury Ranges areas to the north of Albury are currently being developed for a range of residential, commercial and industrial land-uses. A conservation network has been proposed, and this report evaluates the likely success of the conservation network by evaluating the population viability of Squirrel Gliders.

A spatially-explicit individual-based simulation model was used to investigate the viability of the Squirrel Glider population in the Thurgoona and Albury Ranges region of southern NSW. The results of the "best estimate" model scenario suggest that the Squirrel Glider should persist in the region for at least one hundred years. However, these results should be interpreted with caution due to some uncertainty surrounding the model inputs and hence outputs.

The areas set aside as part of the conservation network, including the relatively large blocks of woodland and the narrow roadside and creek-side reserves appear to be able to sustain the glider population. The strategies put in place by the Albury-Wodonga Corporation, and in particular much of the forward tree planting conducted 20 to 30 years previously, has established a solid baseline from which the conservation of Squirrel Gliders can be achieved.

The population size was found to be particularly sensitive to survival rates and management should take measures to ensure that survival rates of Squirrel Gliders do not get too low. This might involve, for example, controlling predation by cats and reducing the rate of animal-vehicle collisions. It was also found that creating and maintaining connections in the habitat to facilitate Squirrel Glider movement would have a positive impact on population size. This might even help to offset the negative effects of the Hume FWY which may be draining animals from the region. Furthermore, it was found that the carrying capacity of habitat can have a large influence on population size. Therefore maintaining high quality habitat by conserving large hollow bearing trees, providing nest boxes and planting an understorey might be an effective management strategy to make the population robust against extinction - as this would help to increase the carrying capacity and hence the population size.

Further research and monitoring would be useful to get a better understanding of the system and to keep a check on the population size and survival rates. New information generated from field studies could help better inform management and feed back to improve future modelling efforts. Furthermore, the monitoring of the population size in selected areas would help to detect population decline before it is too late.

Key Principles for Management

1. Maintenance of a connected habitat network, including narrow strips, larger blocks and small patches will be critical at both large and small spatial scales.

2. Monitoring and managing (i.e. keeping as low as possible) mortality rates across the landscape and at locations adjacent to negative land-uses (e.g. roads, industrial and residential land – the latter due to cats) is important.

3. The Hume Freeway *may* be acting as a sink, and draining animals out of the population. Therefore, restoring connectivity to areas that connect to the freeway would be a lower priority than connecting up areas away from the freeway.

4. Installation and ongoing maintenance of quality nest-boxes in areas of forward tree planting and more recent revegetation sites would allow Squirrel Gliders to colonise those areas about 100 years before natural hollows form.

5. Ongoing management of the woodland habitat is critical, including what people plant in residential areas that adjoin patches woodland. For example, planting species of trees and shrubs in residential areas that flower during winter will likely be of benefit to gliders. However, it should be noted that these benefits may be reduced if the rate of predation by cats in backyards is high.

6. Thurgoona and the Albury Ranges are currently being urbanised. Opportunity exists to set up a long-term experimental research and monitoring program that has all the components of a good scientific research program, namely data collected before, during and after development, at sites being developed as well as at sites remaining undeveloped for comparison. There may be opportunity from state and federal governments and philanthropic trusts to establish and undertake this long-term program.

ACKNOWLEDGEMENTS

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INTRODUCTION

The Squirrel Glider (*Petaurus norfolcensis*) is a small arboreal marsupial native to Australia. It is patchily distributed from northern Queensland to Victoria along the east coast of the continent (van der Ree and Suckling 2008). It occupies woodland and forest, and in the south its preferred habitat is dominated by *Eucalyptus* species, often with an Acacia midstorey. Throughout its range, much of its habitat has been cleared, with extensive clearing for agriculture in south-east Australia (Claridge and van der Ree 2004). More recent urban expansion along the coast in NE NSW and southern Queensland is threatening its survival there (Goldingay and Sharpe 2005). Consequently, the range and occurrence of the species has declined considerably since European settlement. It is now listed as endangered in Victoria, threatened in NSW and of conservation concern in Queensland. Despite extensive loss and fragmentation of habitat across its range, Squirrel Gliders still persist in many regions. This is most likely because much of the remaining habitat is of high quality and is sufficiently connected. The majority of this habitat forms a network of linear strips along roads, in road reserves and along streams, as well as numerous patches of varying size and shape (van der Ree 2002). However, the extent to which these small and often isolated populations are viable in the long-term is unclear.

The Squirrel Glider is likely to be particularly sensitive to the loss and fragmentation of habitat because its main form of locomotion is by gliding from tree to tree, and gaps between trees greater than a glider's gliding distance may act as a barrier to its movement and dispersal (van der Ree *et al.* 2003). Roads in particular have a large impact on the social structure of populations, form barriers or filters to movement (van der Ree 2006) and reduce survival rates (McCall *et al.* in review). Other threatening processes include increased rates of predation by native and introduced predators and reduced habitat suitability. The majority of studies on the Squirrel Glider have been undertaken in agricultural or forested situations, with the exception of recent work in Brisbane (e.g. Goldingay and Sharpe 2005). Consequently, the specific effects of urbanisation on the species are largely unknown.

The combined cities of Albury and Wodonga on the Murray River are a rapidly growing regional centre. Much of the growth in the region is occurring on the outskirts of the towns, converting primarily agricultural land into residential and industrial land-uses. Squirrel Gliders are widespread around Albury and Wodonga, and frequently occur in remnant and regrowth Eucalypt woodland across the region. In particular, the urban expansion occurring in Thurgoona and the Albury Ranges coincides with prime Squirrel Glider habitat (van der Ree 2003). A regional conservation strategy has been prepared for both Thurgoona and Albury Ranges, and the needs of the Squirrel Glider were a primary driver in that process (Davidson *et al.* 2004; Davidson *et al.* 2005). Areas of land were set aside as habitat for the species, including large blocks of woodland as core habitat zones and linear strips along roads and streams as movement corridors. The effectiveness of the strategies is largely unknown, and the medium-term effectiveness is unlikely to be known for at least 30 to 50 years. In the mean time, decisions are being made about which land to keep as habitat for the species, and which can be converted to residential or industrial land-uses. Objective, scientifically-based information is required to inform this decision-making process.

The aim of this study was to investigate the viability of the Squirrel Glider population in the Albury Ranges and Thurgoona area. Simulation modelling was used to predict how population size would be affected by different scenarios involving different parameter values and different versions of the landscape. The results of the simulations were used to inform a discussion about the viability of the population in the region.

METHODS

Study Area

The study area is located in the Thurgoona and the Albury Ranges region of southern NSW. It is currently a mix of residential, industrial and agricultural land-uses (Fig. 1), with further conversion of agricultural areas to urban and suburban uses planned for the next 20 years. The existing wooded vegetation is comprised of relatively large blocks to the west, and numerous linear strips and smaller blocks to the east. The type and quality of the woodland varies across the region; the woodland in the large blocks to the west primarily occur on rocky soils on slopes while that to the east typically occurs on more fertile alluvial floodplain soils.

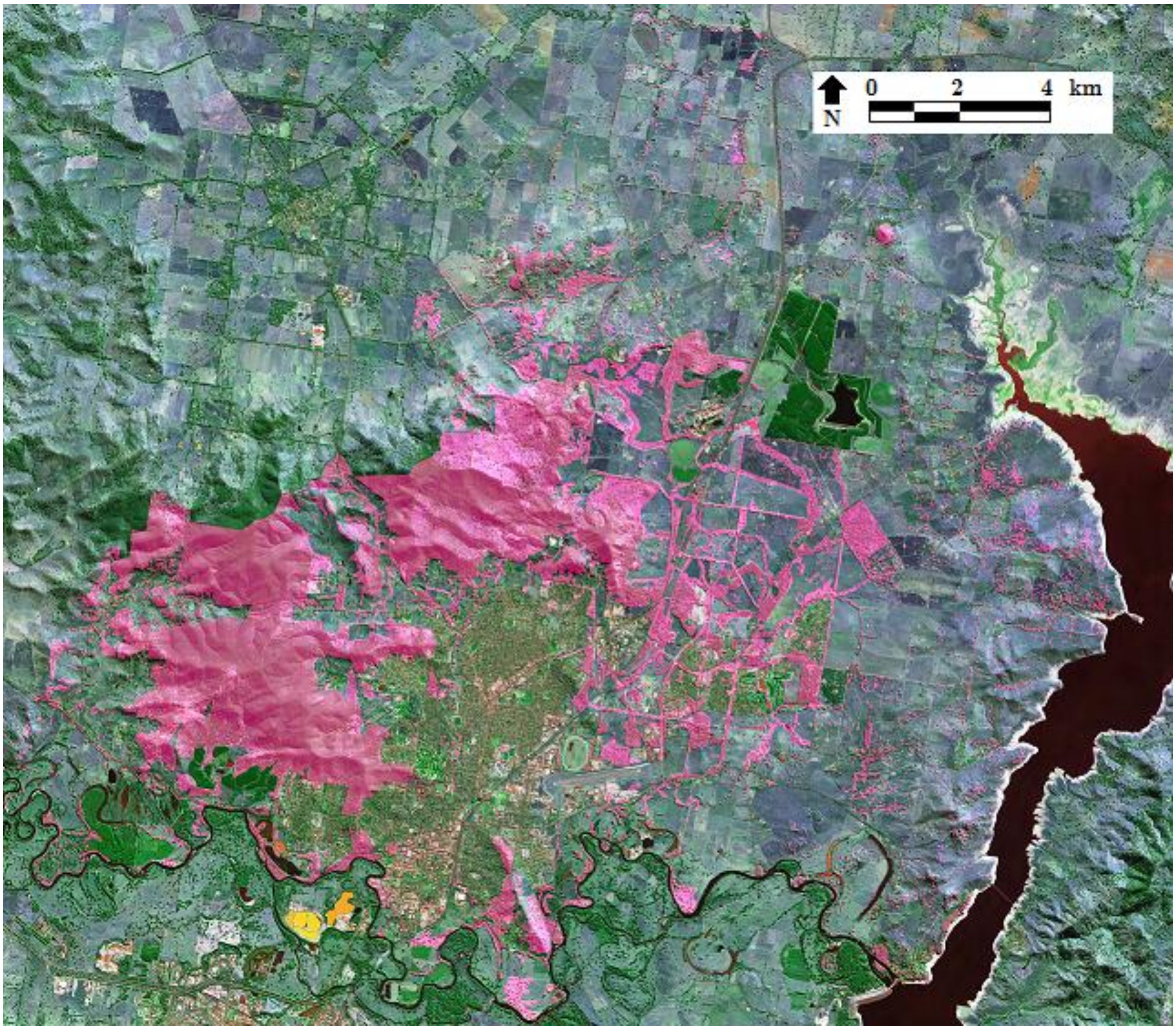

Figure 1. Albury Ranges and Thurgoona study area. Lake Hume is to the east and the Murray River separates the city of Albury from Wodonga to the south. The areas coloured pink represent the wooded vegetation considered in the study. The large blocks of woodland and forest in the central to western portions of the map are located in the Albury Ranges and the network-like vegetation in the central to eastern portions of the map represents the primarily roadside strips of woodland in Thurgoona.

Selection of map input data

The suitability of different vegetation communities across the study area for Squirrel Gliders varies significantly. For example, higher densities of Squirrel Gliders occur on the floodplains to the east than in the steeper terrain to the west. The study area has been classified into four regions of suitability for Squirrel Gliders (Table 1), based on the current understanding of the habitat preferences of Squirrel Gliders. These habitat types (Table 1) were mapped using a vegetation mapping GIS layer from DECCW and their distribution across the study area is shown in Figure 2(a).

Table 1. Regions of different suitability for the Squirrel Glider in the study area.

Habitat Type	Description	Vegetation communities
Non-habitat	Cleared farmland and urban areas. It is assumed that Squirrel Gliders cannot inhabit or disperse through this region type.	Cleared farmland, exotic/native plantings in residential areas
Dispersal-only habitat	Vegetation on the rocky slopes that the Squirrel Glider can probably disperse through, but is unlikely to establish home ranges in.	Open forest/woodland
Low quality habitat	Woodland that can support low densities of Squirrel Gliders.	South West Slopes Box-Gum Woodland
High quality habitat	Woodland that can support high densities of Squirrel Gliders.	Box-Gum Woodland, Colluvial Box-Gum Woodland, River Red Gum Forest

The distribution of habitat suitable for dispersal and supporting resident (high and low density) populations was mapped using maps generated by the AWDC in their conservation strategies for Thurgoona and the Albury Ranges. Hence, these GIS layers include existing and proposed habitat, without distinguishing between the two, and the results of our model assume that all the suitable woodland vegetation proposed for the area has been established.

Major arterial roads are known to have a major impact on Squirrel Gliders (McCall *et al.* in review). The Hume FWY was recently built to bypass the centre of Albury, which now dissects Thurgoona and Albury Ranges. The effects of roads and traffic can extend beyond the edge of the road itself. Similarly, the effects of other land-uses, such as predation by cats that are resident within suburban areas may extend into adjacent bushland. In Figure 2(b) the residential and the Hume FWY infrastructure zones are given; the residential zone is coloured red and the road zone is coloured grey. The relative positions of the residential and road zones and the vegetated regions are shown in Figure 2(c).

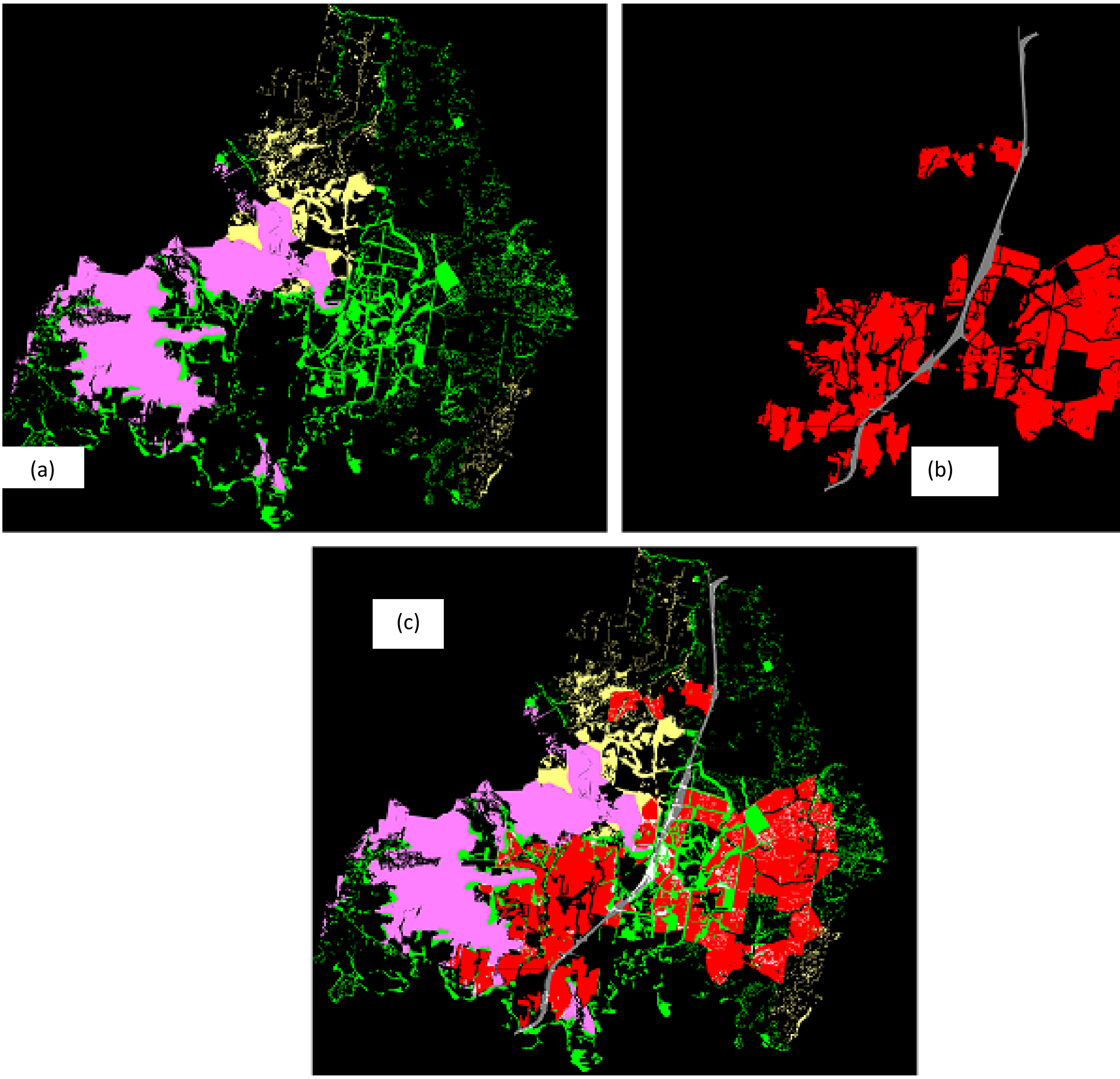

Figure 2. The Thurgoona study area: in (a) the vegetation is colour coded to reflect the different regions of suitability for the Squirrel Glider with black representing non-habitat, pink representing dispersal-only habitat, yellow representing low quality habitat and green representing high quality habitat; in (b) the residential zone is shown as red and the Hume FWY infrastructure zone is shown as grey; in (c) the maps are superimposed and vegetation overlapping with either of the zones from (b) is coloured white.

Population Model

The model used in this study belongs to a class of models known as simulation models. These models differ from other classes of models in that they often incorporate more detail and explicitly account for many of the fundamental functions and processes of the real system being modelled. They can be thought of as simplified analogues of real systems. Simulation models often model systems that have dynamic (i.e. time-varying) and/or stochastic (i.e. random) processes. Most simulation models are computer-based and need to be "run" to investigate system-level behaviour. Simulation models are often run on different sets of parameter values to investigate how the real system might behave under different scenarios. They are particularly useful for the study of systems that are hard to test in real life.

The Squirrel Glider population in Thurgoona and the Albury Ranges can be thought of as a dynamic and stochastic system. It is dynamic since its population size can vary through time and it is stochastic since many of the processes determining its population size are random (e.g. there is a chance associated with surviving a given year). Moreover, it is a spatial system since its population is distributed across space. To model it we used an individual-based spatially-explicit simulation model. The model is individual-based since we account for individual animals and it is spatially-explicit since we account for where they occur in the landscape.

OVERVIEW OF MODEL

Model Parameters and Inputs

The model requires two types of maps as inputs; these are a “region map” and an “other-land-use map”. The region map defines the distribution of different types of habitat in the landscape, namely: non-habitat, dispersal-only habitat, low quality habitat and high quality habitat (Table 1, Fig. 2a). In this study we use this particular map (Fig. 2a) for some of the model runs and altered versions of it for other runs. The map of other land-uses defines the distribution of land-uses that could affect the survival of Squirrel Gliders. For our model we used the Hume FWY and residential zones as two land-uses that could influence survival rates, either directly through mortality via collision with vehicles (Hume Fwy) or indirectly via predation by cats (residential zone) (Fig. 2b).

The maximum number of individuals a landscape can support is known as its carrying capacity, which is abbreviated to the letter “K”. The maximum density of individuals permitted in the low and high quality habitat regions of the region map are set by two parameters called the “low K_density” parameter and the “high K_density” parameter respectively. The study landscape will only be at carrying capacity when its low quality and high quality regions are uniformly at low K_density and high K_density respectively. Therefore, a “K_density” can be thought of as a “carrying capacity density”.

The “social parameter” defines the maximum number of mature individuals that a social group can contain assuming that all social groups are equally sized (where size refers to the number of individuals). It is used to calculate the number of social groups that the model will allocate to the landscape. By dividing the landscape’s carrying capacity into social groups with sizes set to the value of the social parameter, the number of social groups to be allocated is determined. In the model, however, not all social groups are necessarily of this size. The actual sizes of social groups are determined by the relative spatial positions of their centres (which are determined randomly) and the quality and area of the habitat they occupy. However, the social parameter indirectly affects the mean social group size because the more social groups there are in the landscape, the smaller on average they must be (assuming that the carrying capacity is kept constant).

There are three parameters in the model which allocate three annual survival probabilities to regions of habitat in the study area. These parameters are labelled: “broad-scale survival”, “road zone survival” and “residential zone survival”. The regions allocated to these parameters overlap in parts, and in these areas the model assigns the lowest survival probability of those allocated. The broad-scale survival parameter is used to allocate an annual survival probability to all low and high quality habitat. The other two survival parameters are used to allocate survival values to regions associated with the zones of the other-land-use map. The value of the road zone survival parameter is allocated to low and high quality habitat within 20 m of the Hume FWY infrastructure zone. The value of the residential zone survival parameter is allocated to low and high quality habitat within 100 m of the residential zone. For the majority of model runs undertaken in this study, the residential zone survival parameter was set equal to the broad-scale survival parameter. Therefore it will always be assumed in this report, unless stated otherwise, that these parameters are equal.

The “birth number” is a parameter that sets the expected number of offspring a female will produce in a given year if she mates that year. (Note that the birth number must be a number between 1 and 2.) The probability that a female will produce 1 offspring is calculated as 2 - (birth number) and the probability that she will produce 2 offspring is (birth number) - 1. For example, if the birth number is 1.7, then the probability that a female will produce 1 offspring will be 0.3 (i.e. a 30% chance) and the probability she will produce 2 offspring will be 0.7 (i.e. a 70% chance).

The “dispersal distance” parameter defines the expected dispersal distance of a juvenile. Specifically, the distance covered in a dispersal event is determined by a random variable taken from an exponential distribution with a mean set equal to the "dispersal distance" parameter.

A final input parameter for the model is the “environmental stochasticity” parameter. This parameter defines a percentage for varying the birth and death processes to simulate the effect of environmental influences (e.g. rainfall). This parameter description and the others given are summarised in Table 2.

Table 2. Summary of input parameter and input map descriptions.

Input/Parameter	Unit	Description
Region map	-	Defines location of non-habitat, dispersal-only habitat, low quality habitat and high quality habitat.
Other-land-use map	-	Defines location of Hume FWY infrastructure zone and residential zone.
Low K_density	Individuals per ha	Maximum density of individuals permitted in low quality habitat.
High K_density	Individuals per ha	Maximum density of individuals permitted in high quality habitat.
Social parameter	Individuals per social group	Maximum number of mature individuals that a social group can hold assuming that all social groups are equally sized.
Broad-scale survival	0 – 1	Annual survival probability allocated to all low and high quality habitat.
Road zone survival	0 – 1	Annual survival probability allocated to low and high quality habitat within 20 metres of the Hume HWY road infrastructure zone.
Residential zone survival	0 – 1	Annual survival probability allocated to low and high quality habitat within 100 metres of the residential zone.
Birth number	Offspring per female per year	Expected number of offspring a female will produce in a given year if she mates that year.
Dispersal distance	Km	Expected dispersal distance of a juvenile.
Environmental stochasticity	Percent	Variation in birth and death processes due to environmental factors.

Model Structure

The model has two core components: a “spatial module” and a “population dynamics module” (shaded gray in Figure 3). The spatial module distils the landscape into a network of seeds and links; where the seeds represent the centres of social groups and the links represent the dispersal roots among social groups[1]. The spatial module also determines a survival probability for the individuals of a social group associated with a particular seed and a "ceiling value" which determines the maximum number of mature individuals that the social group can support. Thus, the spatial module produces three outputs: a network of seeds and links, a list of seed survival values and a list of seed ceiling values. These outputs are shown under the heading “spatial information” in Figure 3. Details of the spatial module can be found in Appendix A.

Using the spatial information generated from the spatial module and some additional input parameters (i.e. dispersal distance, birth number and environmental stochasticity) the population dynamics module is run. Starting at an initial population size at around half the carrying capacity, population dynamics are simulated on the network using random birth, death and dispersal processes. As a result, Squirrel Glider numbers are projected for each social group at yearly intervals for 100 years. This procedure is repeated 100 times and then averaged across social groups; altogether this constitutes one run of the population dynamics module. The output of the population dynamics module (i.e. mean social group size through time for each social group) is stored as an output file in a folder; the folder is shown as a rectangle with a tag at its top left corner in Figure 1. Details of the population dynamics module can be found in Appendix A.

The seeds of a particular network are distributed randomly across the landscape by the spatial module. In order to avoid a possible bias in results arising from a particular spatial distribution of seeds, the spatial module and population dynamics module procedures are repeated multiple times - we use N=60 iterations; this creates many output files in the folder which are then averaged over the iterations to give a final output. The flow of procedures is shown in Figure 3. A variable labelled i is used to count through the iterations. To start with i is initialised with a value of one. This is shown with the notation $i \leftarrow 1$ (below the word “START”). At the start of each iteration the value of i is checked; this check is represented by the diamond. If i is less than or equal to a number N (which is set to be 60), then the spatial module and population dynamics module are run and the value of i is incremented by one; the action $i \leftarrow i+1$ in Figure 3 indicates that i is incremented. After i is incremented its value is checked again. Hence, a loop is involved where the spatial module and population dynamics module are run until i is greater than N. When this happens the loop terminates, the folder contents are averaged and then the model ends. The endpoint is marked with the word "FINISH" in the Figure 3. In total, 6000 population dynamics projections are used to produce the final output. This results from 60 iterations of the population dynamics module and 100 projections for each population dynamics module run. We refer to the process of producing the final output as "one run of the model".

[1] In the field of "network theory" seeds are referred to as nodes; we use the term "seed" to be consistent with Stewart and van der Ree (2009).

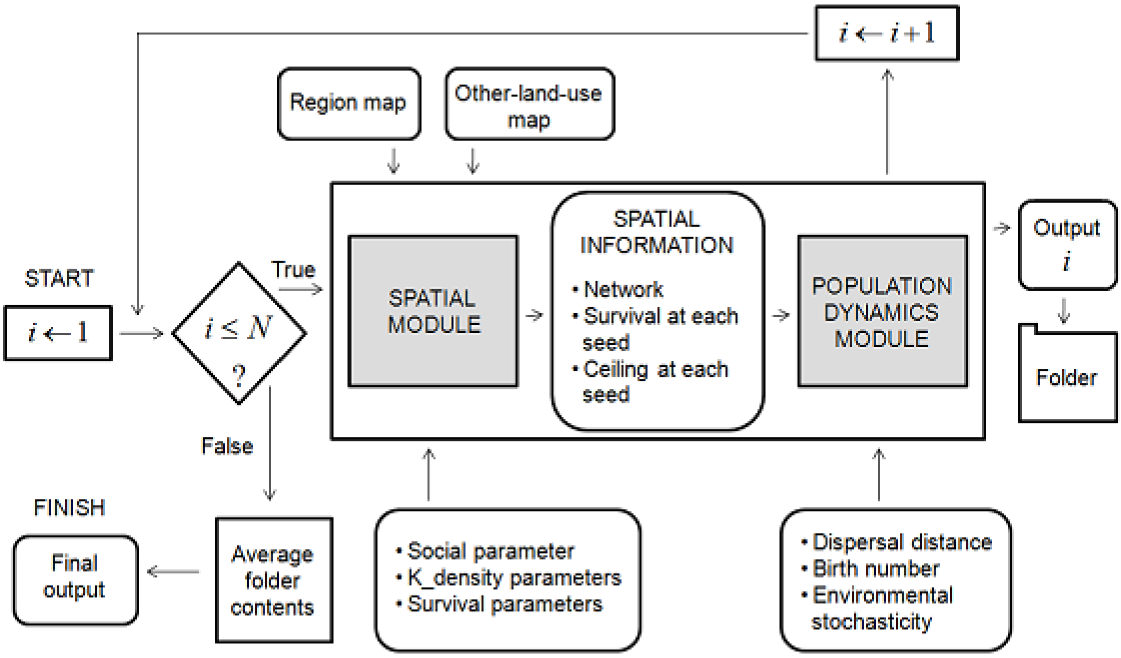

Figure 3. The structure of the simulation model used in the study. The two core components – the spatial module and the population dynamics model – are shaded gray. Inputs and outputs are shown as rectangles with rounded corners whilst "actions" are shown as standard rectangles (with the exception of the diamond). The diamond represents a "value check" on the counter *i* and the square with a tag at its top left corner is a folder that is used to store output files.

Model runs

Best Estimate

We ran the model for a set of parameter values that we thought might best reflect the population in the study area. The values chosen were based on field estimates from nearby populations as well as from values described in the scientific literature. We refer to these values as "best estimate" values. They are given in the second column of Table 3.

Endpoint Comparisons

There was some uncertainty about the best estimates of some parameter values. We chose to reflect this uncertainty by presenting an interval over which the true value could lie. These intervals, labelled "exploratory intervals", are given in the third column of Table 3. With the exception of the broad-scale survival parameter, each parameter's exploratory interval has a midpoint equal to its best estimate. For these parameters we were unsure of whether their true values would be larger or smaller than our best estimate of them; thus we centred their exploratory intervals on their best estimates. On the other hand, our best estimate for broad-scale survival was used as an upper bound on its exploratory interval. This was because it was based on data for an area that is not in close proximity to an urban zone and the effect of the urban area is more likely to be detrimental than beneficial to the Squirrel Glider. Therefore, if the true value of broad-scale survival in the study area is not equal to our best estimate of 0.7, then it is probably going to be a lower value due to urban pressures. We refer to the set of exploratory interval midpoints as "base case" values. They are given in the fourth column of Table 3. The base case may give a more realistic projection of population size through time than the best estimate scenario as it accounts for a possible negative impact from the residential zone by incorporating a lower broad-scale survival value. However, we are uncertain about the true value of this parameter and it is best determined by field studies in the study area.

We used the base case values as a central point in the "parameter space" about which we could investigate the sensitivity of the model to changes in the input parameters. By varying the parameters across their exploratory intervals one at a time, we explored their impact on the model output. Parameter impact values were calculated by subtracting the minimum model evaluation from the maximum model evaluation over each particular interval. The model was computationally expensive to run, and therefore it was only evaluated at three points for each interval: the left endpoint, the midpoint and the right endpoint[2]. As the exploratory intervals were all centred on their base case values, only one run of the model was needed to determine all the midpoint evaluations - namely a run using the base case values. In total, 17 runs were undertaken in this part of the modelling process; two endpoint runs for each of the eight exploratory intervals plus the base case run. We refer to these runs as the "endpoint comparisons", since the impact of the input parameters on the model output was primarily determined by endpoint evaluations (excluding the base case). The endpoint comparisons can be thought of as a basic type of sensitivity analysis; to see how they relate to sensitivity analysis in general, see Appendix B.

Different widths of exploratory intervals may be suitable for different uses of a sensitivity analysis (see Appendix B). The widths of the exploratory intervals we used were primarily chosen to reflect “state of knowledge” uncertainty; this enabled us to determine which parameters it would be most important to acquire accurate estimates of in future field studies in order to improve future modelling. Furthermore, our choice of intervals enabled the robustness of the base case scenario to be explored and were also mostly wide enough to give an indication of model behaviour and to provide some information useful for management. The exception, however, was the road zone survival parameter. Its interval [0.2, 0.4] was not wide enough to account for complete road mortality mitigation. To include the effect of mitigation would require the right end point of the interval to be set at a value of around 0.7 rather than 0.4. We chose to keep a thin interval for road zone survival in order to adequately reflect "state of knowledge" uncertainty. Mitigation scenarios were explored in other runs of the study.

Varying Broad-Scale Survival

As we thought that broad-scale survival might have a big impact on the model output we undertook some additional runs to gain further insight into the effect of varying this parameter. In all, six runs were used to look at the effect of varying broad-scale survival; these covered the broad-scale survival values of 0.65, 0.66, 0.67, 0.68, 0.69 and 0.7. Apart from the broad-scale survival parameter which was varied, all other parameters were set to their best estimate values (second column of Table 3).

[2] When we say that the model is evaluated at a point on a particular parameter's exploratory interval, we mean that the parameter of interest is set to the point's value and that all other parameters are set to their base case values; the evaluation is then the value of the model output when it is run. The output value referred to here is mean population size.

Table 3. The values of the parameters used in the best estimate simulation, in the endpoint comparisons and in the model runs used to investigate the impact of varying broad-scale survival are given. In the endpoint comparisons the model was evaluated at each of the exploratory interval endpoints and for the base case.

Parameter	Best Estimate	Exploratory Interval	Base Case	Varying Broad-Scale Survival
Low K_density	0.5	[0.4,0.6]	0.5	0.5
High K_density	1.5	[1.1,1.9]	1.5	1.5
Social parameter	5	[4.5,5.5]	5	5
Broad-scale survival	0.7	[0.65,0.70]	0.675	0.65, 0.66, 0.67, 0.68, 0.69, 0.7
Road zone survival	0.3	[0.2,0.4]	0.3	0.3
Birth number	1.7	[1.6,1.8]	1.7	1.7
Dispersal distance	3	[2,4]	3	3
Environmental stochasticity	10	[5,15]	10	10

Landscape Alterations

After examining the results from the model runs looking at the effect of "varying broad-scale survival", we were motivated to investigate the effect of altering the landscape. We noticed in the results for "varying broad-scale survival", lower densities were predicted for some regions of the landscape than for other regions[3]. We wondered whether these low densities could be increased by adding or removing vegetation in certain areas. We investigated the effect of two types of landscape alteration which we thought might have a positive impact on population density: (i) a group of joins linking gaps in the vegetation and (ii) cuts either side of the road infrastructure zone creating breaks in the vegetation. The base map we used assumed that any gap > 5 m between patches of habitat represented a barrier to gliders. While we know that gliders can regularly traverse gaps of 30 – 40 m, we retained this 5 m barrier size for a number of reasons. First, we wanted to ensure that population size was not overestimated, and we wanted to ensure our estimates were conservative. Second, we do not know what size gaps gliders are willing to cross while dispersing, and decided it safer to not guess the thresholds and keep it at the scale of the underlying maps. Finally, the aim of this stage of the model was to test the in principle importance of cutting or joining habitat. In some respects it is irrelevant whether the specific location we joined or cut was or was not a barrier to dispersal; the important outcome is what effect the management action had on population size after 100 years.

The joins were placed strategically in the landscape to eliminate "dead ends" (gaps > 5 m on the habitat maps) in the woodland. This was done to help facilitate movement in the patch by enabling more pathways for dispersing individuals. When individuals can move more easily through the landscape, locally extinct regions are more likely to be re-colonised and regions with low numbers can be bolstered by immigrants. This will often increase the viability of a population. Of the joins chosen, some were used to bridge a break in the vegetation defined on the region map across Old Sydney Rd. This break may not be an actual barrier for the Squirrel Glider in real life; however, by comparing the results of the scenario with joins to the scenario without joins we can get a sense of the importance of having links in

[3] These results are given and discussed in detail later in this report.

this region of the landscape. The vegetation gaps that we chose to join have been enclosed with red loops in Figure 4 and the proposed joining vegetation - just visible - is white. To incorporate the joins in the model we used an altered version of the region map shown in Figure 2(a) as a model input, with high quality vegetation added to the join positions.

Cuts were used to investigate the effect on population size of preventing animals from accessing the Hume Fwy, thus isolating the low survival zone adjacent to that road. The cuts are shown as white strips in Figure 4; a red arrow points to one of the cuts. To incorporate the cuts into the model we once again altered the region map of Figure 2(a) by removing the strips of vegetation shown in Figure 4.

We compared landscape alteration scenarios with and without joins for broad-scale survival values of between 0.65 and 0.7 and with road zone survival set either to the broad-scale survival parameter or to 0.3. By setting road zone survival equal to broad-scale survival, we were able to simulate scenarios without road mortality (i.e. without mortality due to collisions with vehicles). The relative effects of cuts and joins were examined for broad-scale survival values of 0.67 and 0.68 and with road zone survival set to 0.3 (i.e. in the presence of road mortality). Apart from the survival parameters, all other parameters in this stage of the modelling were set to their best estimate values.

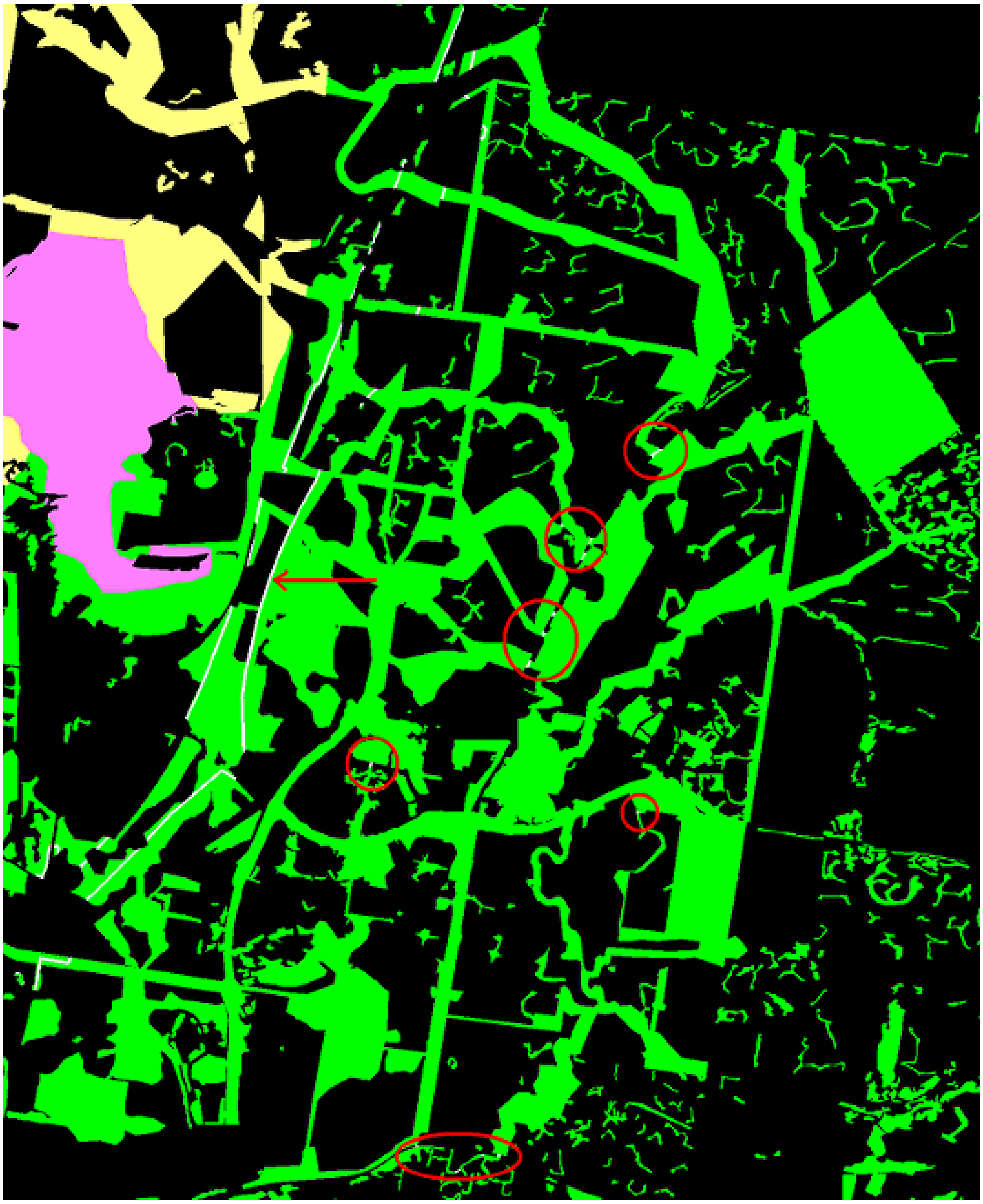

Figure 4. The location of landscape alterations (cuts and joins in habitat) in Thurgoona and Albury Ranges. The red arrow points to a cut on the east side of the Hume FWY infrastructure zone and the red loops enclose joins. The cuts and joins are shown as white. The three top-most groups of joins bridge a break in the vegetation across Old Sydney Rd defined by the region map. This break may not be an actual barrier for the Squirrel Glider in real life, but by creating joins across it we can investigate the importance of maintaining connections in this part of the landscape.

Localisation

We compared two scenarios with relatively low survivals localised around the residential zone (red region in Figure 2(b)) to scenarios with low widespread survival. For the scenarios with localisation we set the residential zone survival parameter equal to 0.65 (in one run) and 0.66 (in another); broad-scale survival (and thus survival away from the residential zone) was set to 0.7. These model runs were the only occasions on which we used a different residential zone survival value to the broad-scale survival value. For the scenarios with low widespread survival, broad-scale survival (and residential zone survival) was set to 0.65 and 0.66. In these different scenarios, we investigated whether the population could support relatively low survival rates if they were localised in a buffer around residential zone and not too widespread.

Region of Interest

When making comparisons between different scenarios in our results section we mainly focus on the mean population size in a "region of interest" on the east of the Hume Freeway in the hundredth year of the simulations. The region of interest is shown as white in Figure 5 and roughly corresponds with the Thurgoona portion of the study area. Whilst there is a focus on the results of the region of interest, the model runs still cover the whole landscape. Therefore the results for the region of interest can still indirectly be influenced by other parts of the landscape through dispersal.

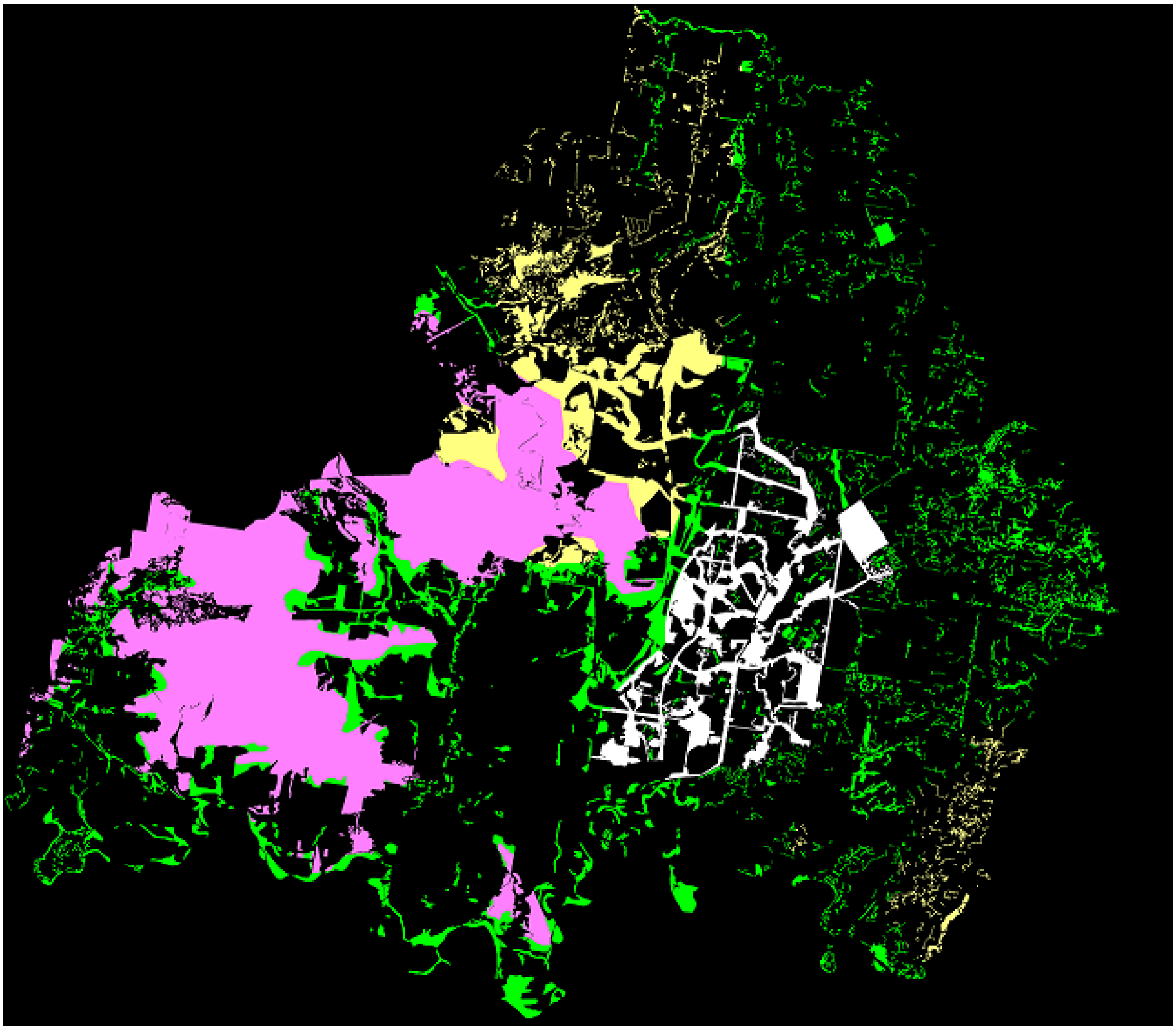

Figure 5. The vegetation of the Thurgoona and Albury Ranges landscape with the region of interest shaded white. The black regions represent non-habitat; the pink, dispersal-only habitat; the yellow, low quality habitat; and the green, high quality habitat. The habitat in the region of interest (shaded white) is high quality.

Comparison of Best Estimate Model Run with Field Data

We compared the results of a field survey with the best estimate model results. The field surveys at 13 sites in Thurgoona and the Albury Ranges (Table 4, Fig. 6, and see Appendix C for results) aimed to determine the presence of Squirrel Gliders at eight sites and density at five sites.

We used model densities taken from the hundredth year of the best estimate run to compare with the field data; these densities corresponded to the spatial locations of the field sites. Three of the sites (Urana Rd, Thurgoona Dve and Central Reserve Rd), however, had locations that lay in regions with a carrying capacity of zero according to the region map. For these sites, model predictions would necessarily be zero (which could be unrealistic given that in the field survey it was thought that these sites could possibly support gliders) and so model predictions were also made for these sites at the closest map point with a non-zero carrying capacity. At Urana Rd and Central Reserve Rd this was done with shortest path distance; the closest points are marked with red arrows in Figure 6. At Thurgoona Dve a straight line distance was used; in this case the closest point was only five meters away.

Table 4. Details of sites trapped in November – December 2007 to determine the presence and density of Squirrel Gliders. The "Conservation Strategy Regions" and "Precincts" correspond to "Threatened Species Conservation Strategies" published by the Albury-Wodonga Development Corporation, 2004.

Site Number	Site Name	Conservation Strategy Region	Precinct	Type
1	Bells TSR	Thurgoona	C	Density site
2	Old Sydney Rd	Thurgoona	C	Density site
3	Mitchell Park	Thurgoona	D	Density site
4	Mr. Brown’s	Albury Ranges	H	Density site
5	Urana Rd	Albury Ranges	J	Density site
6	Thurgoona Dve	Thurgoona	-	Presence-absence
7	Olympic Way	Albury Ranges	H	Presence-absence
8	Central Reserve Rd	Albury Ranges	H	Presence-absence
9	Bungambrawatha Creek	Albury Ranges	I	Presence-absence
10	Centaur Rd	Albury Ranges	J	Presence-absence
11	Pearsall St	Albury Ranges	K	Presence-absence
12	Nail Can Hill Reserve	Albury Ranges	L	Presence-absence
13	Bowna Reserve	-	-	Presence-absence

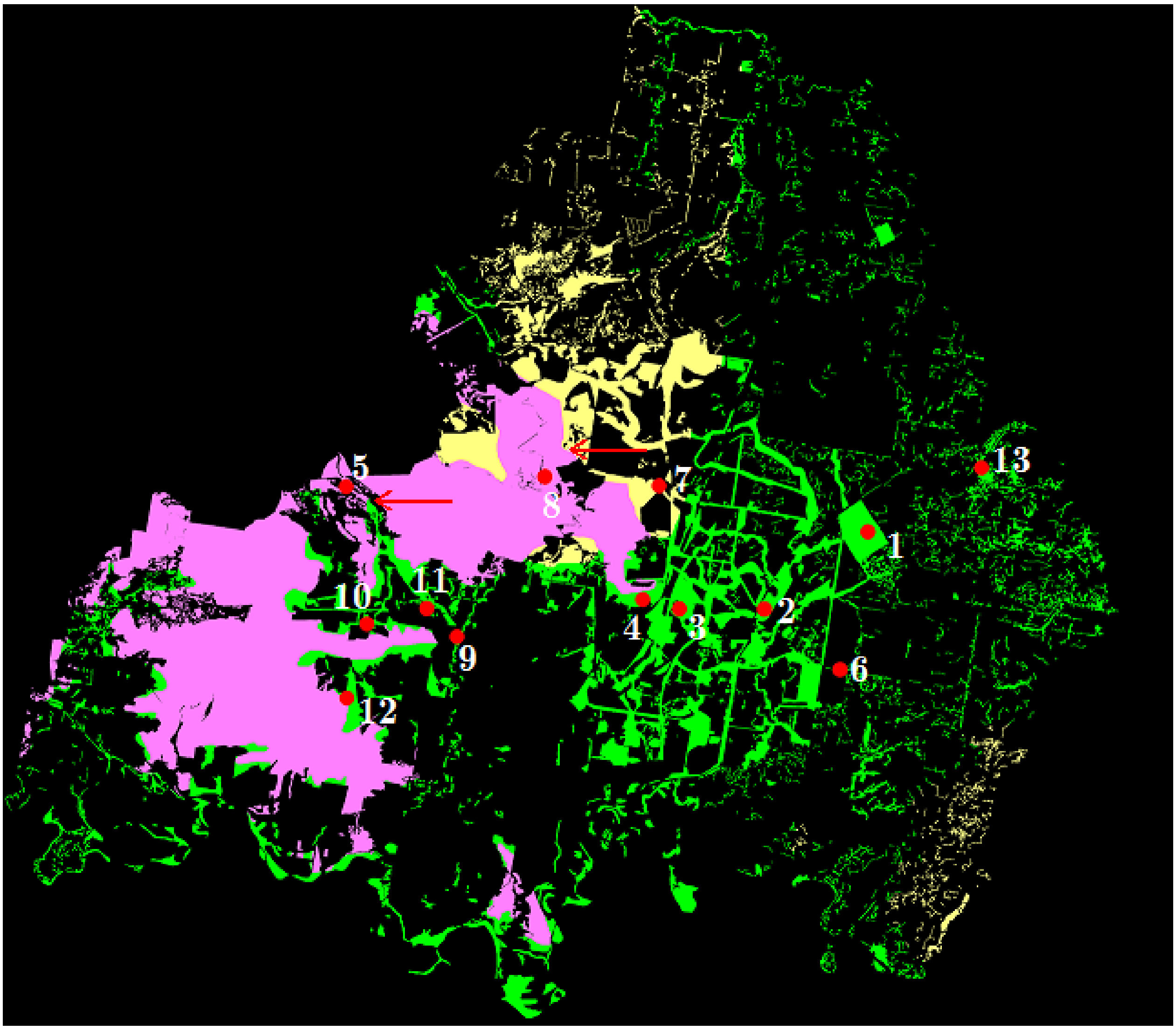

Figure 6. Map of study area with field survey sites shown as red dots. The numbers correspond to the site numbers in Table 4 and the arrows point to the closest points to Sites 5 and 8 (Urana Rd and Central Reserve Rd, respectively) with a non-zero carrying capacity. The black regions represent non-habitat; the pink, dispersal-only habitat; the yellow, low quality habitat; and the green, high quality habitat. The pink and black regions both have a carrying capacity of zero.

RESULTS

Best Estimate

Using the best estimate parameter values, the model outputs suggest that Squirrel Glider populations in the Thurgoona and Albury Ranges region should persist over the next 100 years. This is evident in Figure 7, where the population curve for the region of interest (bottom curve) stabilises at an equilibrium of around 1021 individuals by the hundredth year. The mean population size for the whole landscape (top curve) shows some decline which is due to the populations in the small and isolated patches going extinct.

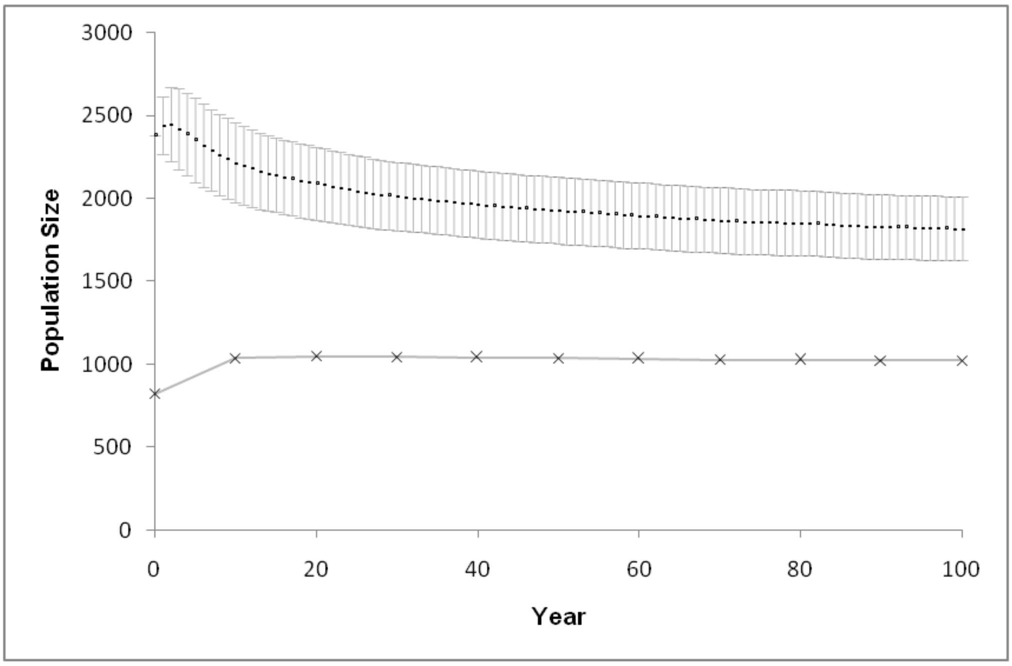

Figure 7. Mean population size through time for the whole landscape (top line) and for the region of interest (bottom line) for the best estimate scenario. The error bars on the projection for the whole landscape indicate the mean standard deviation in population size averaged across the 60 networks used in the run.

The density of Squirrel Gliders in the hundredth year of the best estimate model run is fairly high across most of the region of interest (Fig. 8) with a mean of 1.03 individuals per ha[4]. However, the population did not stabilise at carrying capacity, but rather at 68.57 percent of carrying capacity in this region. In other parts of the landscape, density is mostly high in the high quality habitat, except for the isolated patches, which have low predicted densities. Note that the low quality habitat of Figure 2(a) (coloured yellow in Figure 2(a)) can never be white in a density map as its carrying capacity density is always less

[4] In all the density maps presented in this report (Figs. 8, 10, 11, 14), white represents areas of high density, while black denotes a density of zero.

than that of the high quality habitat (coloured green in Figure 2(a)). Moreover, the carrying capacity of the dispersal-only habitat (coloured pink in Figure 2(a)) is 0.0, and therefore it necessarily appears black in Figure 8.

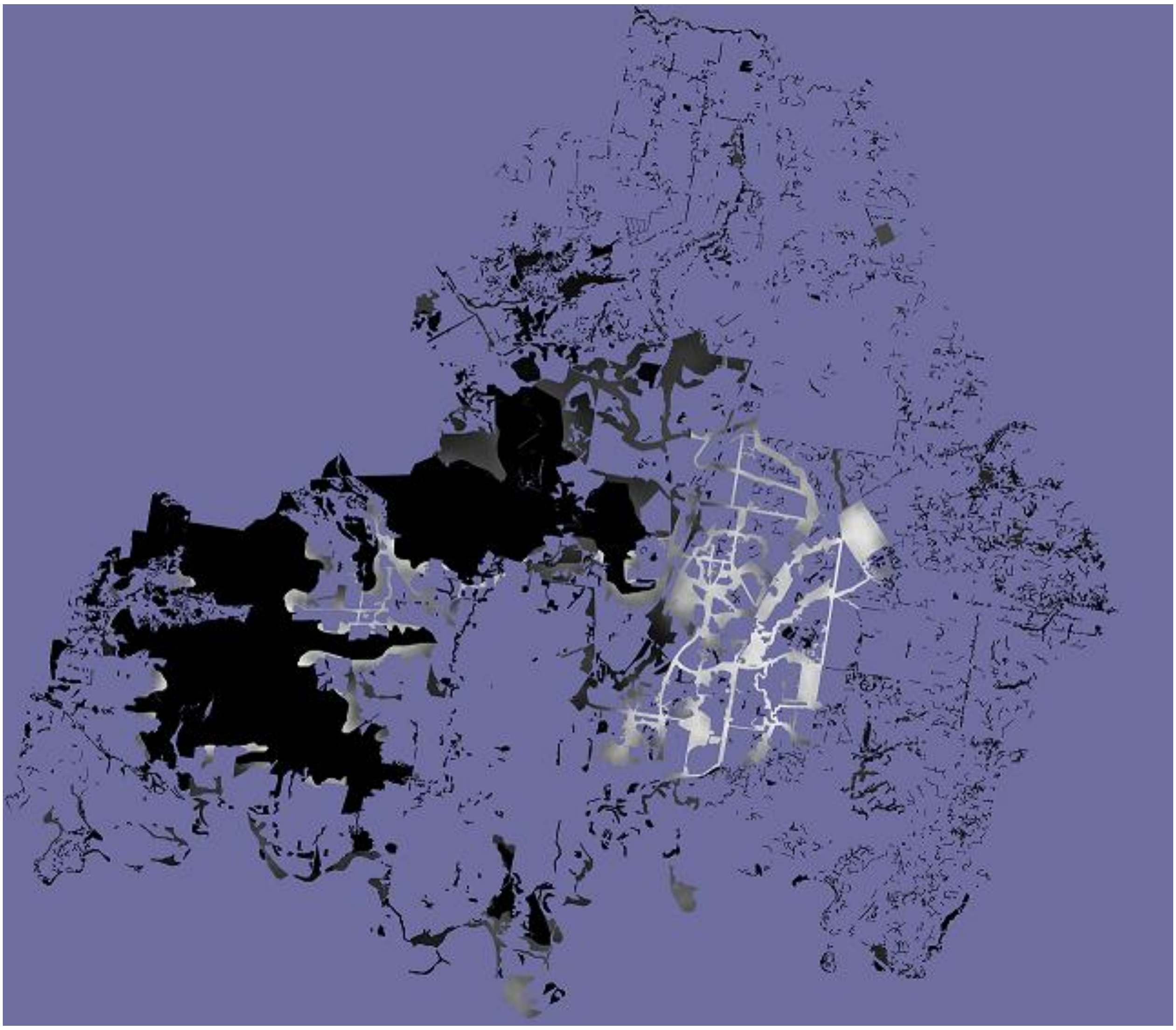

Figure 8. Map of the predicted density of Squirrel Gliders across the whole study area after 100 years using the best estimate parameter values. White represents high K_density and black a density of zero. Blue areas are regions of non-habitat.

Endpoint Comparisons

In the base case scenario, the model projected a population size of 657.31 ± 21.40 (s.e.) individuals for the region of interest[5]. The model evaluations at the exploratory interval endpoints show considerable variation about this value (Table 5). For example, the value of the model evaluated at the left endpoint of the exploratory interval for broad-scale survival is 148.15 individuals which is 77.46 percent less than

[5] In this section population sizes correspond to year 100 in the region of interest.

the base case evaluation; furthermore, the evaluation at the right endpoint is 1021.50 individuals which is 55.41 percent greater than the base case evaluation. The variation in the model output is less when the other parameters are varied, which is shown in the impact column of Table 5. Following broad-scale survival, the birth number, social and high K_density parameters all have large impacts; environmental stochasticity has a moderate impact; while the road zone survival, low K_density and dispersal distance parameters all have small impacts.

For every exploratory interval, the value at the midpoint evaluation (i.e. the value for the base case scenario) was found to lie between the values at the endpoint evaluations. For example, the midpoint evaluation of the environmental stochasticity exploratory interval was 657.31, which is less than 766.86, the value of the left endpoint evaluation, and greater than 486.93, the value at the right endpoint evaluation. This suggests that population size may have a negative relationship with environmental stochasticity. The type of relationship, either positive or negative, that population size will have with a particular parameter, as predicted by the endpoint comparisons, is given in the last column of Table 5. The "+" symbol indicates a positive relationship and "-" a negative one. Note that the impacts of the low K_density and dispersal distance parameters are low relative to the standard error of their model evaluations; this means that their predicted relationship types should not be considered significant[6].

[6] We did not undertake significance testing as we were primarily interested in ordering the parameters and were less concerned with their relationship with the output. Insignificant impacts should be small any way due to the sample sizes used in the simulations.

Table 5. Results of the endpoint comparisons; the parameters are ordered from highest impact to lowest impact. The "Left Endpoint Evaluation" column gives the model evaluations at the left endpoints of the exploratory intervals and the "Right Endpoint Evaluation" column gives evaluations for the right endpoints. Similarly, the "Left s.e." and "Right s.e." give the standard errors for the endpoint evaluations (calculated for sample sizes of 60 - the number of networks used in each model run). The "Impact" column gives the value of the minimum model evaluation subtracted from the maximum over each interval. The "Type" column shows the predicted relationship type between the input parameter and the output population size.

Parameter	Exploratory Interval	Left Endpoint Evaluation	Left s.e.	Right Endpoint Evaluation	Right s.e.	Impact (max-min)	Type
Broad-scale survival	[0.65,0.7]	148.15	14.74	1021.50	9.56	873.35	+
Birth	[1.6,1.8]	353.46	18.29	896.43	11.63	542.97	+
Social group	[4.5,5.5]	401.54	24.76	874.61	13.81	473.07	+
High K_density	[1.1,1.9]	438.70	16.29	888.19	23.99	449.49	+
Environmental stochasticity	[0.05,0.15]	766.86	18.21	486.93	20.37	279.93	-
Road zone survival	[0.2,0.4]	643.53	17.89	690.16	20.17	46.63	+
Low K_density	[0.4,0.6]	668.94	18.86	636.40	22.23	32.54	-
Dispersal	[2,4]	650.20	19.18	666.51	19.46	16.31	+

Varying Broad-Scale Survival

The plots of mean population size through time reveal markedly different results for the broad-scale survival values of 0.65 and 0.7 (Fig. 9). The curve for the value of 0.65 shows a relatively steep decline, whilst the curve for 0.7 stabilises at an equilibrium. The curves for the values in between 0.65 and 0.7 bridge the gap between the two extremes.

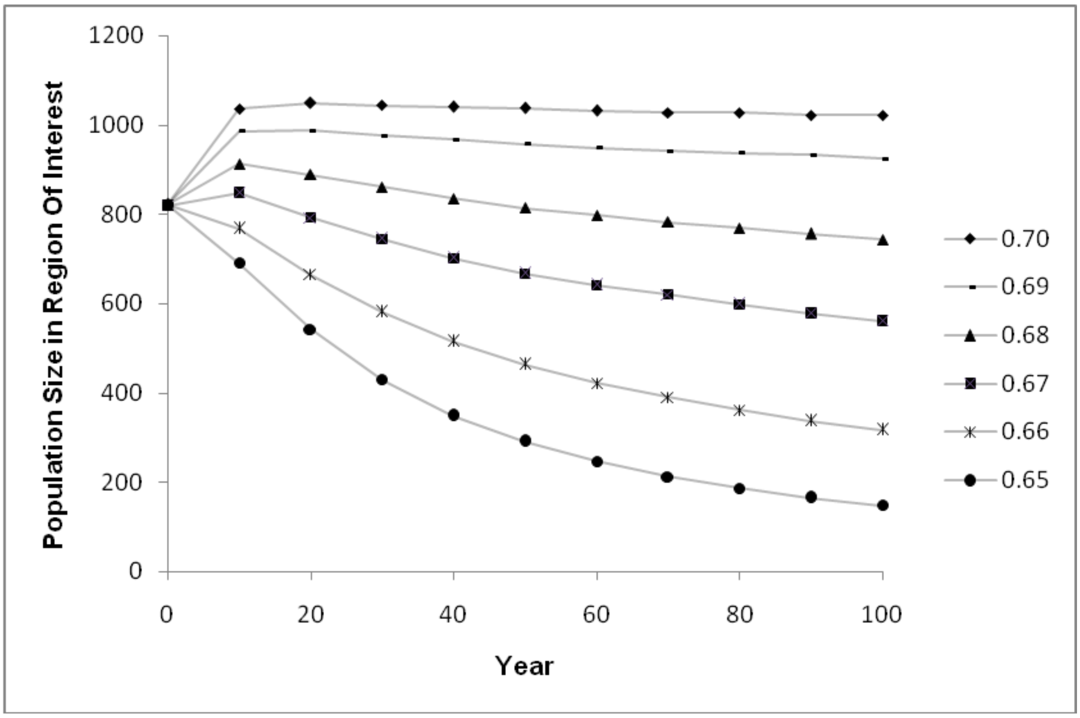

Figure 9. Mean population size in the region of interest through time for broad-scale survival values between 0.65 and 0.7. Best estimate values are used for all other parameters.

As mentioned in the "Best Estimate" results section, the equilibrium population size in the region of interest reached when broad-scale survival value was 0.7 was around 1021 individuals which is only 68.57 percent of carrying capacity; the region of interest which is 993 ha can actually support around 1489 individuals at carrying capacity (i.e. at 1.5 individuals per ha). The population sizes and the corresponding densities and percentages of carrying capacity for the other broad-scale survival values are given in Table 6.

Table 6. Mean population size, mean density and % of carrying capacity at year 100 in the region of interest for different values of broad-scale survival.

Broad-scale survival	Mean population size	Mean density	% of carrying capacity
0.70	1021.50	1.03	68.57
0.69	924.30	0.93	62.04
0.68	743.84	0.75	49.93
0.67	561.60	0.57	37.70
0.66	316.82	0.32	21.27
0.65	148.15	0.15	9.94

The density maps of Figure 10 show mean glider densities in the region of interest after 100 years for survival values ranging between 0.65 and 0.7. Whilst there is considerable variation in density between the maps, there is also spatial variation within each map. This is most noticeable for the maps corresponding to the survival values of 0.67 and 0.68 where the eastern portion of the region of interest shows higher densities of Squirrel Gliders than the habitat closer to the Hume FWY.

A time series of density maps calculated using a broad-scale survival of 0.675 is shown in Figure 11. We present it because it shows the trajectory of the midpoint of the two extremes (0.65 and 0.70) of broad-scale survival we investigated. The maps in Fig. 11 show an initial peak in density and then a steady decline as time progresses. The density in the central portion of the region of interest decreases the quickest.

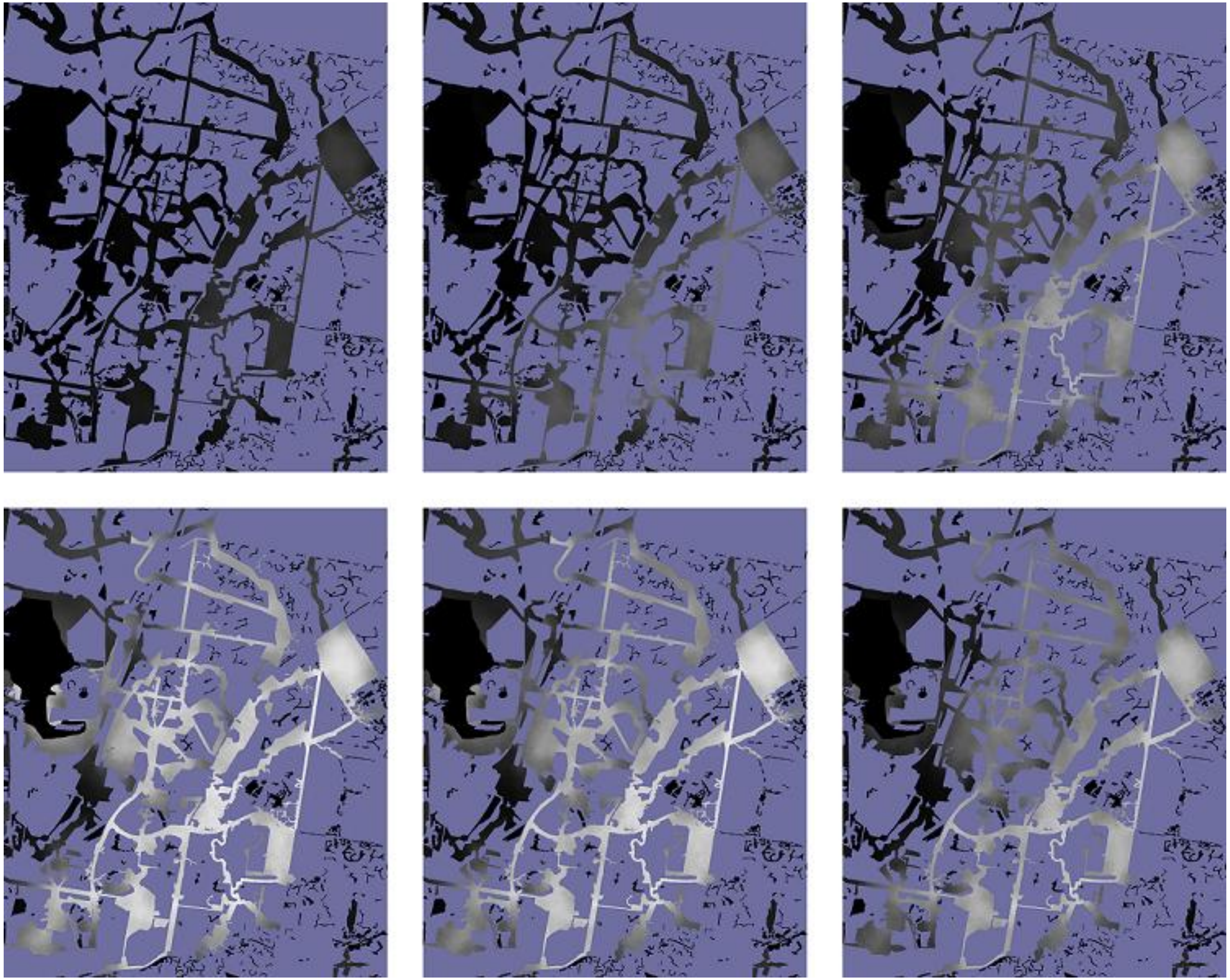

Figure 10. Density of Squirrel Gliders at year 100 in the region of interest. Clockwise from the top left, the broad-scale survival values corresponding to the maps are 0.65, 0.66, 0.67, 0.68, 0.69 and 0.7. All other parameters were set at their best estimate values. White represents high K_density and black a density of zero. Blue regions represent non-habitat.

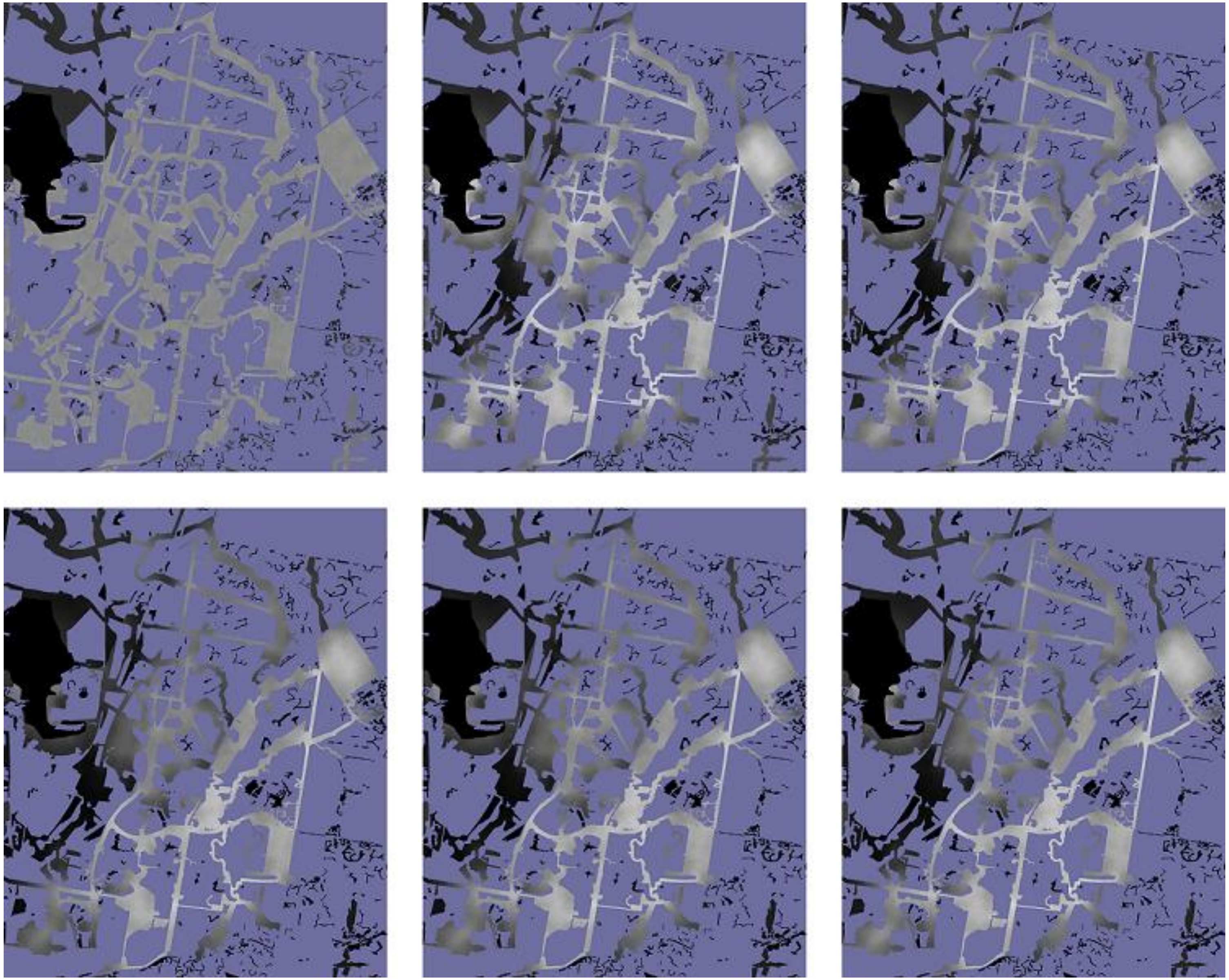

Figure 11. A time series of density maps for the base case scenario using a broad-scale survival rate of 0.675. Time progresses at 10 year intervals clockwise from the top left. Density maps for years 0 to 50 are shown on this page, 60 to 100 on next page. White represents high K_density and black a density of zero. Blue regions represent non-habitat.

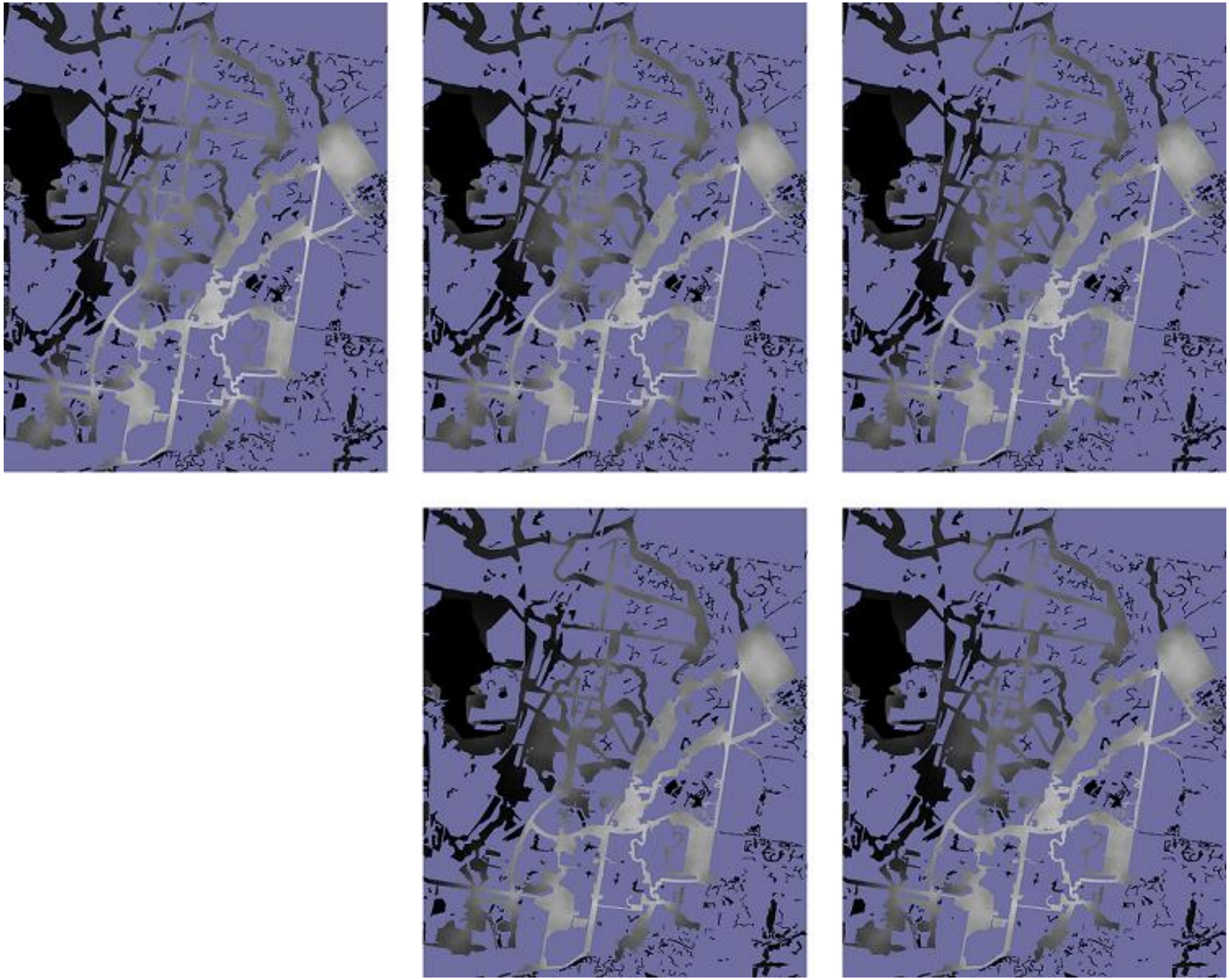

Figure 11 continued. The time series continues at 10 year intervals clockwise from top left. This page shows density maps for years 60 to 100.

Landscape Alterations

When joins were added to the landscape, the population size[7] after 100 years was found to be higher than without joins. This was true across all the broad-scale survival values considered (i.e. 0.65 to 0.7) and in both the scenarios with and without road mortality (Fig. 12).

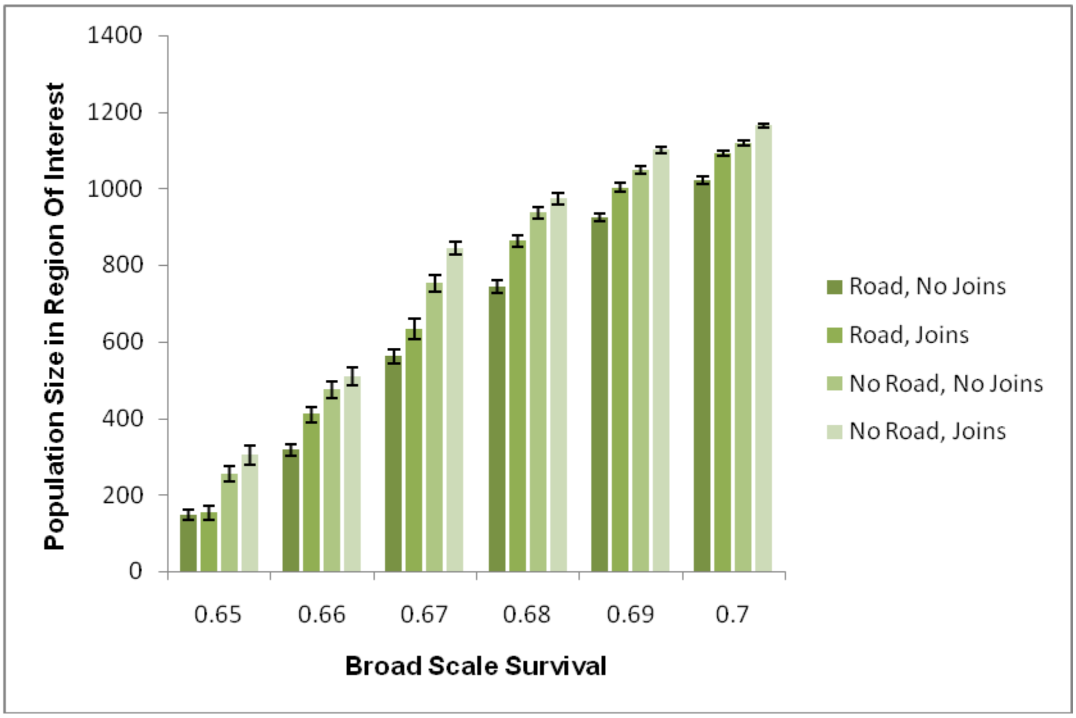

Figure 12. The effect of creating joins in the landscape on the size of the Squirrel Glider population in region of interest in Thurgoona. Broad-scale survival groupings are given on the horizontal axis. The "Road" and "No Road" categories correspond to model runs with and without road mortality, respectively. Similarly, "Joins" and "No Joins" indicate runs with and without joins, respectively.

The model runs investigating the relative effects of joins and cuts show that using only cuts would increase population size more than only joins; whilst using both cuts and joins would increase population size the most (Fig. 13). The trend holds for both the broad-scale survival values investigated. Whilst cutting the landscape is more effective, this is unlikely to be the best conservation strategy (see discussion).

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[7] In this section of the results and in the following section, when we talk of population size, we refer to mean population size in the region of interest in the hundredth year of the simulations.

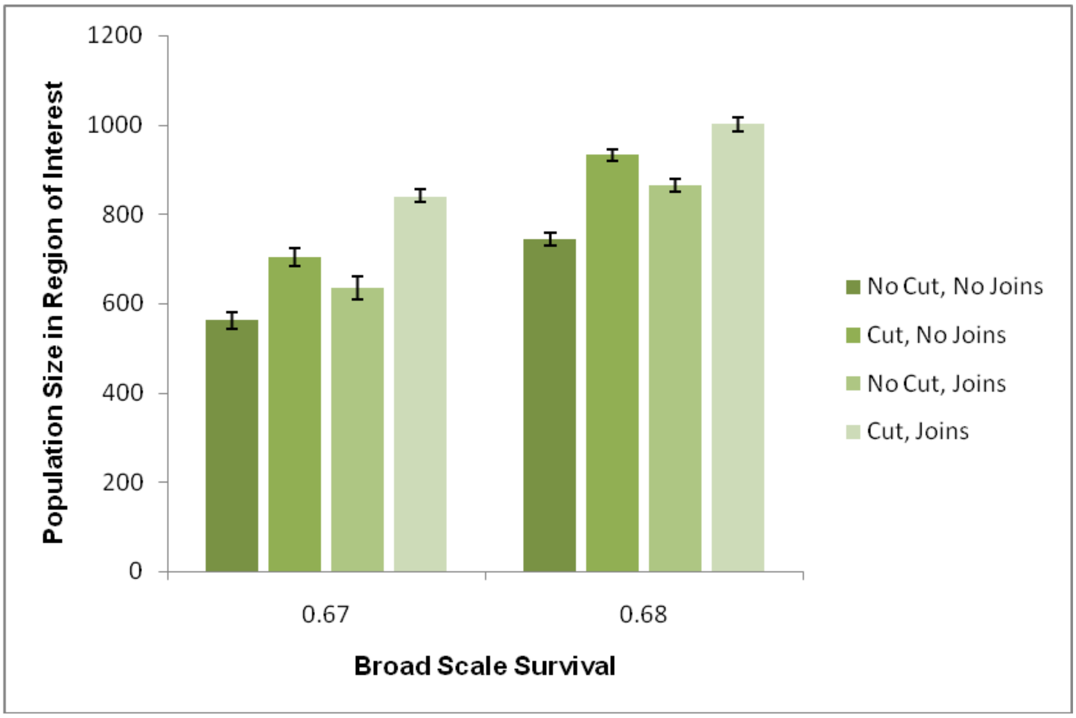


Figure 13. The relative effects of cuts and joins on the size of the population of Squirrel Gliders for broad-scale survival values of 0.67 and 0.68. The categories "No Cut" and "Cut" indicate when cuts are absent and present in model runs respectively. Similarly, the categories "No Joins" and "Joins" indicate when joins are absent or present respectively.

The effects of cuts and joins on population density are shown in Figure 14. The figure shows density maps for model runs with a road zone survival of 0.3 and a broad-scale survival of 0.68. Cuts and joins both increase the glider density in the central portion of the region of interest. The joins alone increase population size in the region by 16.11 percent.

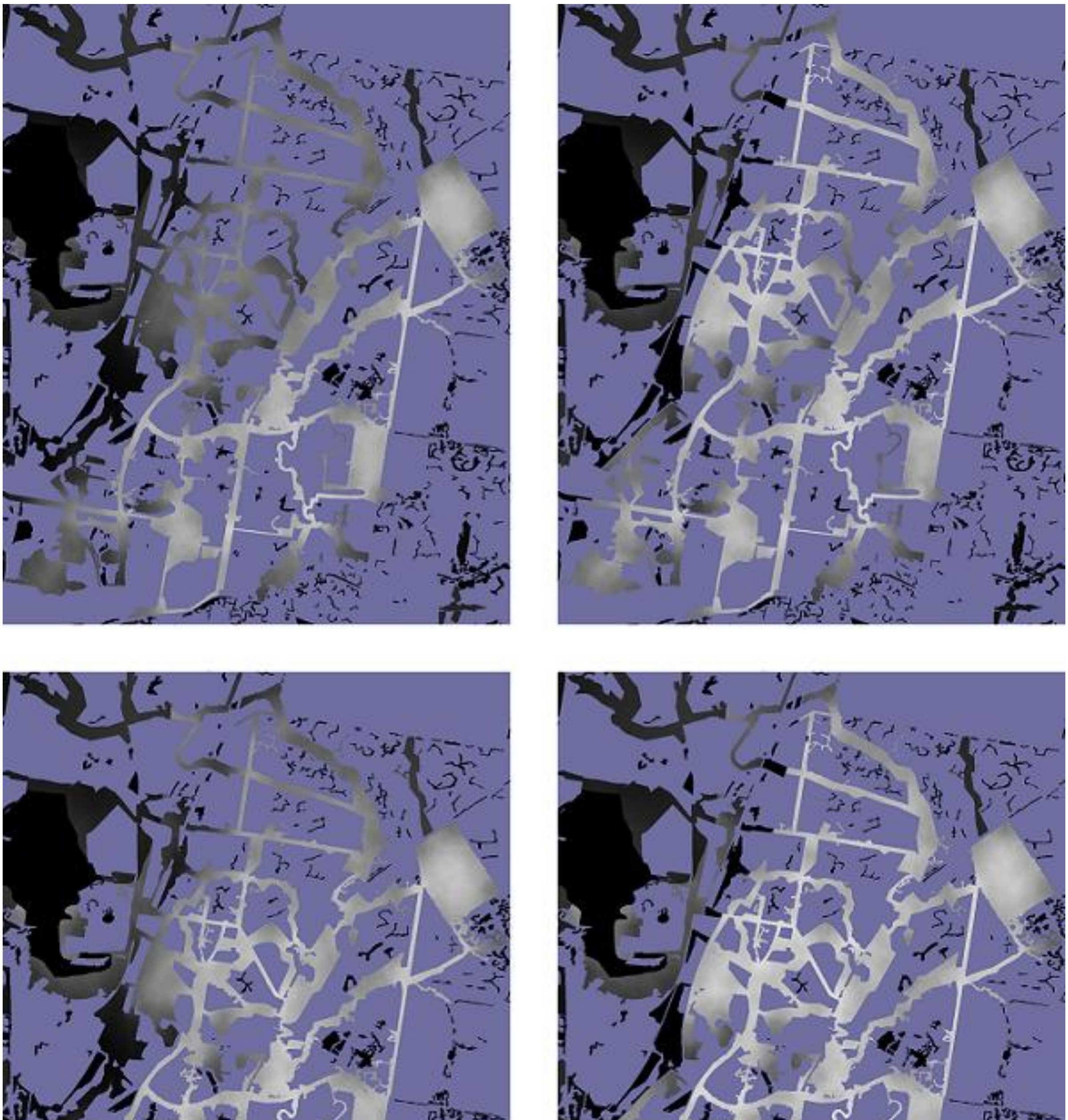


Figure 14. Density of Squirrel Gliders across the region of interest after 100 years with a broad-scale survival rate of 0.68. The top left panel corresponds to no cuts and no joins; top right to cuts and no joins; bottom left, no cuts but with joins; and bottom right shows cuts and joins. White is high K_density, black is a density of zero and blue is non-habitat.

Localisation

When the relatively low survival rate of 0.65 and 0.66 were localised to within 100 metres of the residential zone, the population size within the region of interest was much greater than when broad-scale survival was set to 0.65 or 0.66 (Fig. 15).

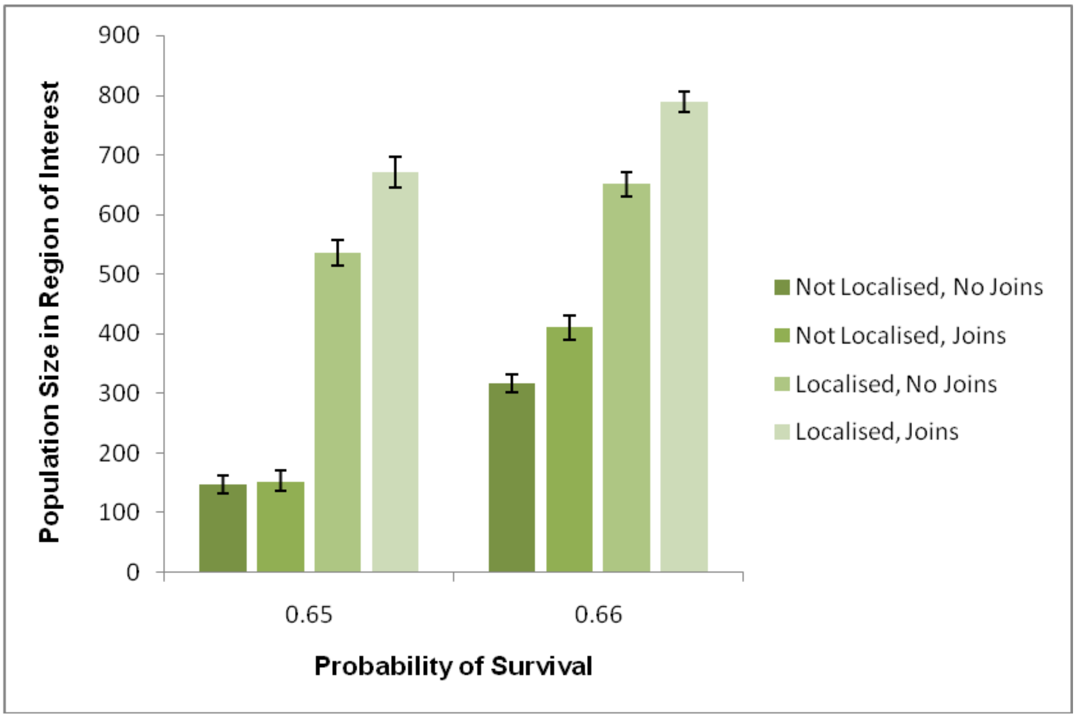


Figure 15. The effect of having low survival of Squirrel Gliders localised to a 100 metre buffer around the residential zone on the density of Squirrel Gliders within the region of interest. In the model runs for the "Not Localised" category, broad-scale survival was set to the survival values on the horizontal axis; whilst in the "Localised" category broad-scale survival was set to 0.7 and the residential zone parameter was set to the values on the horizontal axis.

Comparison of Best Estimate Model Run with Field Data

The density of Squirrel Gliders estimated from our field surveys were in general agreement with the model's best estimate results for the sites in, or close to, the region of interest (sites 1 to 4). The model predicted densities of 0.91 and 0.72 for the sites at Mitchell Park and Mr Brown's respectively (close to the Hume Freeway), a higher density of 1.11 at the Old Sydney Road site, and an even higher density of 1.29 at the Bells TSR site. The field data followed a similar trend with densities of 0.67 and 0.78 at Mitchell Park and Mr Brown's respectively, a higher density at Old Sydney Road of 0.89, and an even higher density of 1.22 at Bells TSR. The comparisons for all the sites are listed in Table 7.

Table 7. Comparison of the density of Squirrel Gliders estimated from field surveys and from the model using best estimate for the input variables. The location of sites 5, 6 and 7 occurred in regions with a carrying capacity of zero according to the region map. Instead of comparing these sites with the model outputs corresponding to these locations (which would necessarily be zero), model predictions for the closest map points with a non-zero carrying capacity were used instead. The numbers in the last three columns are Squirrel Glider densities (i.e. individuals per ha) and "A" represents an absence and "P" a presence.

Site Number	Site Name	Region Map Habitat Type	Field Result	Best Estimate	Closest Point Estimate
1	Bells TSR	High quality	1.22	1.29	-
2	Old Sydney Rd	Disperse-only	0.89	1.11	-
3	Mitchell Park	High quality	0.67	0.91	-
4	Mr. Brown’s	High quality	0.78	0.72	-
5	Urana Rd	High quality	1.00	-	0.66 (556 m away)
6	Thurgoona Dve	Low quality	A	-	0.00 (5 m away)
7	Olympic Way	High quality	A	0.29	-
8	Central Reserve Rd	Non-habitat	P	-	0.43 (736 m away)
9	Bungambrawatha Creek	High quality	A	0.00	-
10	Centaur Rd	Disperse-only	A	0.66	-
11	Peasall St	High quality	A	1.10	-
12	Nail Can Hill Reserve	High quality	P	0.83	-
13	Bowna Reserve	High quality	P	0.21	-

DISCUSSION

Best Estimate

When the model was run using our best estimates for the parameter values, the population in the region of interest stabilised at around 1021 individuals by the hundredth year (Fig. 7). Since there is some uncertainty associated with the parameter values, the input maps and the model assumptions, this result should be interpreted with some caution. However, it does suggest that the population in this region may be viable into the future.

It is interesting to note that the population in the region of interest did not stabilise at carrying capacity, but rather at 68.57 percent of carrying capacity. This is in contrast to typical results for less spatially explicit models, which usually stabilise at carrying capacity when they reach equilibrium. However, it follows in line with metapopulation models, which can reach equilibrium with a portion of empty patches (Hanski 1999). In a sense, the social groups used in the model of this study act in an analogous way to the patches within metapopulation models.

Endpoint Comparisons

The ordering of the parameters given in Table 5, from highest to lowest relative effect on population size can be used to help prioritise future field work efforts. Efforts could focus on getting accurate estimates of those parameters with the highest impacts. In particular, since our uncertainty in the true value of the broad-scale survival parameter had a very high relative effect on final population size, future field work could focus on improving our estimate of this parameter.

It was found that for each interval the midpoint evaluation lay in between the endpoint evaluations. Whilst it is hard to make conclusive statements, since only three points for each interval were evaluated, this result does suggest that the model may be behaving montonically; that is, the value of its output will either increase or decrease with an increase in a particular parameter value.

Of the parameters varied in the endpoint comparisons, management has the potential to influence the value of the survival and K_density parameters and perhaps to some degree the size of litters. The order of impact, from high to low, given in Table 5 can help prioritise management efforts. It suggests that management should mainly focus on keeping broad-scale survival at a high level and then allocate effort in decreasing order to keeping the values of the birth number, high K_density, road zone survival and low K_density parameters high. However, the current parameter values (as measured in the field) and the ease with which a parameter can be changed should also be taken into account. For example, if broad-scale survival is at a high level but high K_density is only at a medium level, then priority should be given to increasing the high K_density parameter as the returns will be greater. On the other hand, it may be better to allocate effort to broad-scale survival over the birth number, even if broad-scale survival is already at a high level and the birth number is only at a medium level. This is because it may be easier to manage for broad-scale survival and a slight increase in this parameter is better than a negligible increase in the birth number.

As noted in the methods section, the exploratory intervals were primarily set to reflect the uncertainty around the state of knowledge and the intervals were mostly wide enough to give an indication of the most important parameters for management to focus on. However, this was not the case for the width of the road zone survival interval. The potential to influence the population by changing this parameter would be greater than that reflected in its impact value[8] in Table 5. Furthermore, since it may be easier to shift the value of the road zone survival parameter through management than the other parameters, it may be worth prioritising effort for this task.

As the model evaluations for the endpoint comparisons only calculate population size in the region of interest, it is not surprising that the low K_density parameter had a low impact. This is because there is no low quality habitat in the region of interest and therefore changing this parameter could only ever indirectly affect population size in the region through dispersal of individuals from adjacent areas with lower quality habitat. If maintaining high population densities to the east of the Hume FWY in the region of interest is of primary concern, then this result suggests that increasing the K_density of the low quality habitat to the west of the Hume FWY should only be given a low management priority.

There are some straight forward ways management could have a positive impact on the parameter values. The high K_density parameter could be maintained at a high level by ensuring that large hollow bearing trees are conserved. When there are few large hollow bearing trees in a region, K_density could be increased artificially through the installation of nest boxes. Moreover, increasing the number of trees in area through revegetation might also increase K_density. Broad-scale survival could be maintained at a high level by controlling predation by cats and reducing the negative impacts of barbed-wire fences. Monitoring could be used to detect when survival rates are getting low. Ensuring that there is a regular food source available could be beneficial in more than one way. It might increase the number of individuals that could persist in an area - and therefore the K_density. This may increase the probability of survival for an individual - and therefore the broad-scale survival; and furthermore it could possibly increase the birth number. One way to ensure food is available is to maintain an understorey (including Acacia species) in the Squirrel Glider habitat. This would support the persistence of invertebrates, sap, nectar and pollen) which compose part of the Squirrel Glider's diet.

Varying Broad-Scale Survival

Further runs of the model were undertaken to gain a clearer understanding of the effect of varying broad-scale survival on the density of gliders. The population curves of Figure 9 show that over a broad-scale survival interval of 0.65 to 0.7 the population trajectories will change from having a relatively steep decline to having a non-zero equilibrium. These results suggest there may only be a short interval in Squirrel Glider survival rates that marks the phase between extinction and viability.

Perhaps more revealing than the population curves (Fig. 9) are the density maps of Figure 10. These maps show lower Squirrel Glider densities in the centre of the region of interest than in the east.

[8] Note that with full road mitigation accounted for in the exploratory interval of the road zone survival parameter, the relative ordering of the parameters in Table 5 would still probably stay the same.

This is likely to be due to the combined effect of low survival values near the Hume FWY and the low number of connections between the central and eastern portions of the region of interest as defined on the region map (Fig. 2(a)). In the region map, Old Sydney Rd causes a break in the habitat. Irrespective of whether or not this is an actual barrier in real life, the results suggest that such a barrier could have a negative effect in the study area, especially in combination with the Hume FWY. In the model, the Hume FWY induces a population sink[9] and, most noticeably in the scenarios with low to medium broad-scale survival, this seems to cause the population to be drained from the central part of the region of interest. This process is illustrated in the time series of density maps in Figure 11.

The results discussed in this section demonstrate the impact that a change in the rate of broad-scale survival can have on population size. A small shift in survival can cause what was a viable population to become one vulnerable to extinction. As mentioned in the end point comparisons section, management could help keep survival rates high by controlling cat predation and monitoring could be used to keep a check on survival rates and changes in population size. Apart from the impact of broad-scale survival, the results also suggest that the Hume FWY may be having a negative impact. This motivated the exploration of two different landscape alteration scenarios, and we thought that landscape alterations might be able to help mitigate the negative impact of the Hume FWY.

Landscape Alterations

We were not exactly sure whether creating joins would have a positive effect on population size. For while joins could increase viability by facilitating movement, we had already seen in the base case scenario (Fig. 11) that the Hume FWY could be draining animals from the central portion of the region of interest and therefore enabling more pathways to the Hume FWY could cause even more of the study area to be drained of individuals. The results, however, do show an increase in population size when joins are added even in the presence of road mortality (Fig. 12). This suggests that the sink effect of a major road can be counteracted to some extent by having a sufficient number of connections in the landscape away from the road. In the absence of road mortality, Figure 12 again shows that having joins in the landscape will increase population size. This reiterates the importance of maintaining connections in the study area. An aerial photograph of the region indicates that some of the connections and corridors proposed for the Thurgoona and Albury Ranges areas still need to be established (Fig. 16). Revegetation should be undertaken to make these connections as the results of the modelling suggest that the more connections there are, the more robust the population will be against low densities and ultimately extinction.

[9] A population sink is a part of the population where the birth rate is less than the death rate and where the number of emigrants is less than the number of immigrants. A population source is where the birth rate is greater than the death rate and where the number emigrants is greater than the number of immigrants (Pulliam 1988).

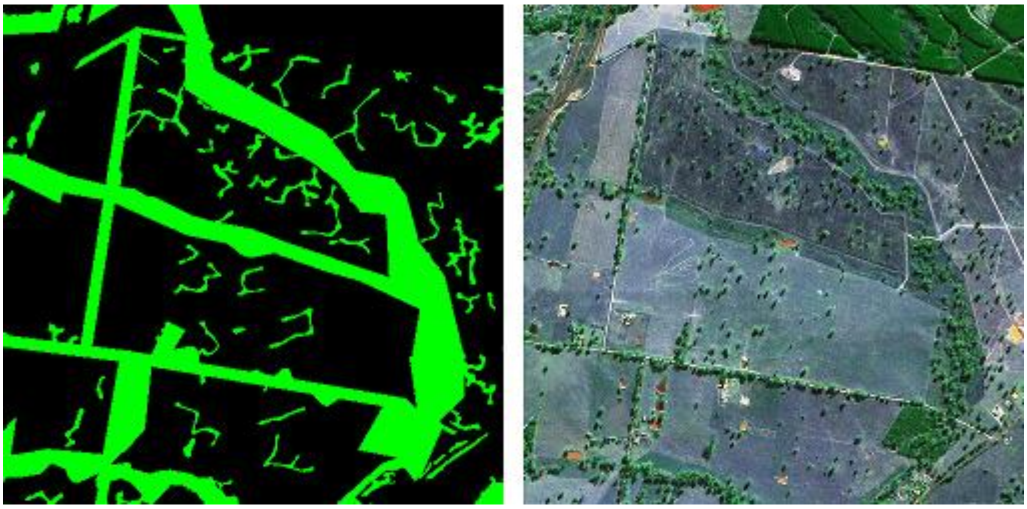


Figure 16. Some of the discrepancy between the proposed vegetation of the region map (left panel) and the current vegetation in an aerial photograph (right panel) is shown for a part of the study area.

The results in Figure 13 show that using only cuts increases population size more than only joins. However, creating cuts in the landscape is a fairly drastic measure to take, especially when the field data supporting a reduced survival rate adjacent to the highway is based on a single study (McCall *et al*. in review). If the survival rate close to the Hume FWY is not lower (or only slightly lower)than the broad-scale survival rate, then adding cuts would unnecessarily fragment the landscape and would likely result in a reduction in the viability of the population in the region of interest. At this stage, the results of the field study supporting a low survival rate adjacent to the Hume FWY are not conclusive and further research needs to be conducted to clarify those results (McCall *et al*. in review). Thus while both cuts and joins could be used to help mitigate the negative impact of road mortality, creating joins is probably the more sensible option at this time as it will have a positive impact irrespective of the level of road mortality.

The relative effects of joins and cuts on the density of gliders can also be seen in the density maps of Figure 14. In Figure 14(b), when the Hume FWY is isolated with cuts, the central portion has a higher density than in Figure 14(a). This result confirms the sink effect from the Hume FWY is lowering the density in the central portion of Figure 14(a). In Figure 14(c) the sink effect is partly counteracted with joins and without the use of cuts. Figure 14(d) shows the effect of both joins and cuts and that using both alterations gives the biggest increase in population density. As mentioned above, since there is some uncertainty surrounding the actual level of road mortality, adding joins is probably a more preferable landscape alteration than creating cuts. When broad-scale survival is 0.68, joins alone causes an increase in population size of 16.11 percent. This is quite considerable given that only a few joins were added in the simulations.

Apart from landscape alterations, rope bridges and glider poles could also be used to mitigate the negative impact of the Hume FWY. If the installation of such structures decreases the chance of Squirrel Gliders colliding with vehicles, then using rope bridges and glider poles could reduce the sink effect and therefore decrease the chance of population extinction. However, we are still unsure about the effectiveness of such structures at reducing road mortality (particularly glider poles), and therefore facilitating Squirrel Glider movement by maintaining a well connected path is probably the most preferable management strategy.

Localisation

It is currently unclear how the residential zone (Fig. 2(b)) may be affecting Squirrel Glider survival in the study area. If the main negative impact of the residential zone results from cat predation, then the effects could be widespread and broad-scale survival could be affected. On the other hand, the effects could be localised. The results in Figure 15 show that if survival values of 0.65 or 0.66 are localised to within 100 metres of the residential zone, then the population size will be greater than if broad-scale survival is 0.65 or 0.66 respectively. Thus, the results show that the population can sustain localised areas of relatively low survival as long as they are not too widespread. When low survival is localised, areas of higher survival can act as sources and individuals dispersing from these areas can increase the number of individuals in the areas with lower densities. Field studies, such as mark-recapture studies, could be used to determine survival rates in different parts of the study area. This would help establish if there are any urban pressures acting on the population. These results could then inform future modelling and management efforts.

Comparison of Best Estimate Model Run with Field Data

If a model's inputs are accurate and if its structure and assumptions are realistic, then some agreement between its output and empirical data might be expected. However, determining whether a model is in agreement with empirical data is often a difficult and subjective process. For example, the Squirrel Glider population acts as a stochastic system and when starting from an initial state many different population trajectories are possible, making it unlikely that the real population would follow the trajectory of the model output exactly. Therefore, a subjective decision has to be made about whether any discrepancy between an observed result and the model output is due to the inherent stochasticity of the system or due to inaccuracies in the model's inputs, structure or assumptions.

We believe the results of the field survey are in general agreement with the best estimate model output. Given that medium to high densities occur throughout the study area in both cases, and a pattern in the density was observed in both cases: the densities at Mr Brown's and Mitchell Park were less than the density at Old Sydney Rd, which was less than the density at Bells TSCR (Table 7). However, we are only looking at one snap shot in time and whilst the best estimate results represent equilibrium densities and therefore won't change much through time, we cannot be sure that the real system is in equilibrium. Nevertheless, the spatial pattern observed was present in many of our model outputs that were not in equilibrium. This suggests that the pattern may be caused by the geometry of the landscape, and the fact that it was observed in the field gives support to our model's structure and assumptions - particularly to our method of modelling spatial structure.

Even with agreement between a model and empirical results, it is hard to determine from a single comparison how accurate the model's structure and assumptions might be. Confidence in models grows when their outputs agree with empirical data for different sets of parameter values over many comparisons. In an ecological setting, ongoing field work can be used to test models under different field conditions. If discrepancies arise, then models can be recalibrated or readjusted based on new information. Improving models in this way is part of the process of "adaptive management" (Beissinger and Westphal 1998).

Not all the field site observations matched with our model predictions (Table 7). The discrepancies may just be a result of the inherent stochasticity of the system or they could result from inaccuracies in the model inputs, structure or assumptions (see Appendix D). However, whilst not being conclusive, the general agreement observed between the field survey and the best estimate run noted is consistent with having accurate model assumptions and structure.

Further Applied Research and Management Directions

Further research (Actions 1 – 4 below) that would improve the accuracy of the model predictions and help to prioritise management effort (Actions 5 – 6 below) should focus on:

1. The habitat maps we used for this model were based on a GIS layer of polygons provided by the AWDC that describes the current and proposed woodland habitat for the Albury Ranges and Thurgoona Conservation strategies. Hence, it includes areas that contain woodland, as well as areas that are currently tree-less but are proposed to be planted into the future. All models are only as good as the data which goes into them, and we propose that when the better and more accurate vegetation data is available for the study area that a revision of the model possibly be considered.

   We decided to use a 5 m gap size as the threshold for dispersal connectivity because (i) we knew that there was a discrepancy between what tree cover was mapped and what existed and we didn't have the resources to ground-truth the habitat maps; (ii) gap size in our model relates specifically to the ability of gliders to cross gaps during dispersal, which is largely unknown and (iii) by using 5 m we have ensured that our population size estimates were cautious and not an overestimate. Ultimately, an underestimate of movement capability will result in an underestimate of population density, ensuring that management actions will err on the side of caution, reducing the risk of extinction.

2. Quantifying differences in the rate of survival and causes of mortality, and the spatial scale of these impacts in road zones and residential areas compared to larger areas of woodland habitat. This data would improve the accuracy of the model at predicting areas of low survival, and hence the projected population size. It could also inform management actions of the causes of mortality that most limit Squirrel Glider populations.

   An important sub-component of this research area would be to further clarify and quantify the size of potential sinks and hence their importance at influencing populations size. For example, we used estimates of survival rate adjacent to the highway of 0.3 compared to 0.7 away from the highway. This difference is based on a single study from NE Victoria and is not sufficient evidence to commence the disconnection of sink areas. Therefore, an important study is to quantify survival adjacent to the highway by undertaking a mark-recapture study over a number of years.

3. Quantification of the relative importance of various factors influencing habitat quality for the Squirrel Glider (e.g. presence and composition of understorey, number and distribution of large hollow bearing trees). Interestingly, an analysis of factors influencing the occurrence and density of Squirrel Gliders within northern Victoria and southern NSW has not been completed. This information could guide future management actions and improve the accuracy of predicted Squirrel Glider habitat both within Thurgoona and across the broader landscape.

4. Determine what constitutes a barrier to movement during dispersal by gliders. For example, what land-uses, infrastructure or characteristics of these, form barriers to the nightly or dispersal movements of Squirrel Gliders? This would provide information on how to ensure connectivity is maintained and restored where required, and also how to limit movement into sink areas, if research suggests that this approach would be beneficial. Approaches to address this current data deficiency include genetic tests, radiotracking and perhaps experimental releases of gliders into novel habitats.

5. Ground-truth and analyse the actual size of gaps in existing woodland and fill all gaps > 30 – 40 m. At present, we have modelled that any gap > 5 m is a barrier to the movement of gliders, which clearly is an underestimate of their actual gliding capability. We used this size because the "woodland habitat" layer we used for the model was the extant and proposed conservation networks proposed in the Albury Ranges and Thurgoona Conservation strategies. Therefore, an analysis of woodland habitat missing from the landscape (compared to what was mapped) is a high priority to identify missing links in the landscape. These missing links can then be prioritised according to the amount of habitat reconnected, land tenure, location, cost etc.

   Other ground truthing includes an assessment of the density of large and hollow-bearing trees across the study area. Hollows are a critical resource for gliders, with individuals and groups using many different hollows and different trees, swapping on average every 3 to 5 days (van der Ree unpub. data). Woodland structure is also likely important for gliders, including the presence and diversity of understorey shrubs that provide food resources (especially Acacia species that provide sap, nectar, pollen and invertebrates).

6. On-ground works are a priority for glider conservation, particularly actions that aim to minimise anthropogenic causes of mortality and increase survival rates. This includes the removal of all barbed-wire fences, preventing or reducing the density of cats (both feral and domestic) in areas with the potential to support gliders, and reducing mortality rates from vehicles (depending on the results of studies to clarify survival adjacent to roads, this may include the provision of rope bridges and glider poles to enable safe road crossings, maintaining tall trees adjacent to roads to minimise glide distance and maximise glide height, and keeping the width of the road to a minimum through glider habitat).

   Other on-ground works include the provision of nest boxes in areas with few hollow-bearing trees and extensive tree and shrub planting. Tree planting should focus on filling in any gaps in the tree canopy that exceed 30 m, as this threshold is a conservative estimate of their average daily glide distance. They can glide further (potentially up to approx 70 m, depending on the launch height), but 30 m represents a good rule of thumb.

The research directions and management actions are difficult to cost because of the range of delivery models available. Many of the further research questions are being tackled as part of other ongoing projects, and the Thurgoona area could simply be an additional study area, thus reducing the cost to the Albury Conservation Company. Other models include competitive tenders, collaborative research partnerships, tertiary student projects and the use of volunteers.

In decreasing order of priority, we recommend the following management actions:

1. Ensure the habitat network is created as planned, including the addition of understorey and nest boxes in new habitat plantings. This also means that incremental clearing and loss of woodland habitat, particularly where it forms a link in the conservation network, is minimised, as all links are valuable.

2. Identify all gaps in the canopy greater than 30 m and fill them

3. Assess the density and distribution of hollow-bearing trees and add nest boxes in areas where they are lacking

4. Map the occurrence and density of shrubs, especially Acacia species, and restore where lacking

In decreasing order of priority, we suggest the following research and monitoring actions:

1. Assess rate and causes of mortality adjacent to major roads and residential areas and if significantly lower than the broad-scale survival of approx 0.7, act to increase survival appropriately.

2. Establish a long-term monitoring program of selected key populations within Thurgoona and Albury Ranges (numerous sites have already been monitored for a number of years as part of various development proposals, and in particular the Hume Fwy upgrade, which should continue). This monitoring should include an assessment of broad-scale survival rates, as the model output was sensitive to this variable.

3. Quantify the size of gaps that limit movement of gliders, especially during dispersal.

4. Contribute to a broader study of the habitat requirements and preferences of Squirrel Gliders on the SW slopes of NSW and Nthn Plains region of Victoria.

5. Re-run the models using improved and more accurate vegetation and habitat maps.

Many of the research priorities have the potential to be answered as part of the same project. For example, a well-designed long-term monitoring program would include an assessment of population density and distribution, reproductive output, survival rates, and causes of mortality etc. Furthermore, the study could include sites that have been part of previous surveys, such as current surveys as part of the Hume freeway duplication, pre-development/EIS studies etc.

CONCLUSION

The best estimate scenario suggests that that the Squirrel Glider population in the study area is viable and likely to persist for at least a hundred years. However, we should interpret this result cautiously as there is uncertainty in many of the model inputs and hence also in the model outputs. The endpoint comparisons showed that varying the input parameters within intervals reflecting our uncertainty in their true values could change the output population size greatly. This was most noticeable for the broad-scale survival parameter, which had the most impact. The results also showed that there might only be a small interval between viability and extinction when broad-scale survival is varied. Management could help keep broad-scale survival at a high level by controlling cat predation. It was also found that the carrying capacity of the high quality habitat (i.e. the high K_density parameter) could have a large impact on population size. By conserving large hollow bearing trees and by installing nest boxes, the carrying capacity could be kept high. Maintaining an understorey in the habitat would also be beneficial as this would support supplement the Squirrel Glider's diet. This could increase the carrying capacity, survival rate and possibly even birth rate of the Squirrel Glider.

Creating and maintaining connections in the habitat to facilitate Squirrel Glider movement could help keep the population size high. This was seen in the landscape alterations section. Furthermore, it was observed that creating joins in the vegetation could help offset the negative impact of the Hume FWY. The population adjacent to the Hume FWY could be acting as a population sink and draining animals from the study area. We confirmed this when we isolated the Hume FWY with cuts in the landscape and this lifted the density in the central portion of the region of interest. However, we assumed the existence of low survival rates adjacent to the Hume FWY and that these rates were driving the sink effect. At this stage the research supporting these rates needs to be clarified (McCall *et al*. in review) and therefore we cannot be certain whether the sink effect exists or not. Thus whilst using cuts could also be used to help mitigate a possible sink effect, to avoid unnecessarily fragmenting the landscape, it might be more appropriate to prioritise efforts towards maintaining connections in the habitat, especially in areas away from the highway initially, as this would be beneficial either way.

Caution should be taken when interpreting the results of a model due to the uncertainty surrounding the model's inputs, structure and assumptions. For example, whilst our model suggests that the population might be viable if broad-scale survival is 0.7 (i.e. in the best estimate scenario), this should only be used as a rough guide. Modelling should work hand in hand with monitoring and field studies, which can be used to detect population decline. Furthermore, monitoring and field studies can improve an understanding of the system; new information from field data can help better inform management and can also feed back into future modelling efforts. (For a short discussion of monitoring see Appendix E.) In the Thurgoona and Albury Ranges study area, field studies could help establish what the survival rates are in different parts of the landscape. This would be useful to know, as we found in the localisation scenario that the population could possibly support regions of low survival as long as they are not too widespread.

In conclusion, our best estimate scenario suggests that the Squirrel Glider population is likely to persist in the study area and that it is therefore viable. However, due to the uncertainty surrounding our model inputs and hence outputs, this result should be treated with some caution. Monitoring and field studies could be used to detect population decline and to better understand the system. Furthermore, management could aim to make the population robust against extinction by taking measures such as controlling predation by cats, conserving large hollow bearing trees, keeping an understorey, and creating and maintaining connections in the habitat.

APPENDIX A

Core Model Components

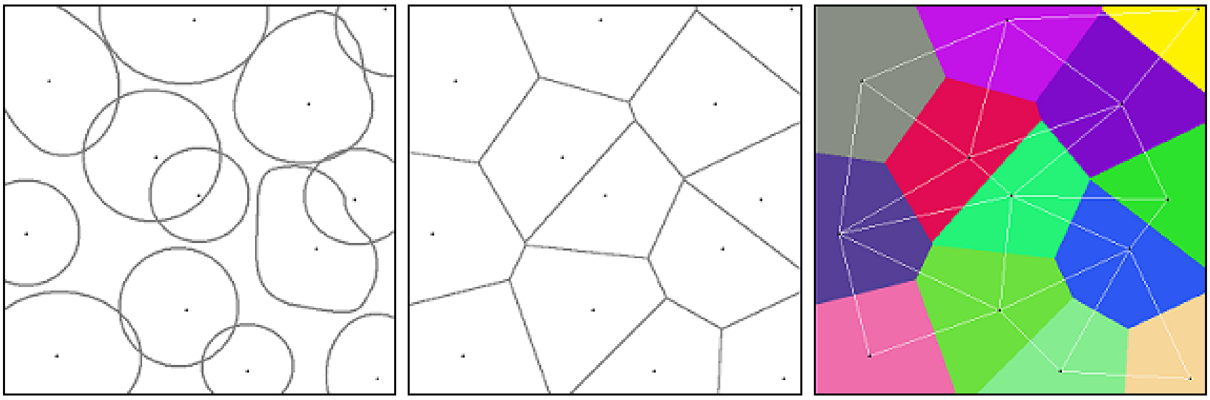
Spatial Module

The spatial module is used to distil the landscape into a network of seeds and links with each seed having a survival value and a ceiling value associated with it. The survival value at a seed sets the probability of survival for the individuals associated with that seed in the population dynamics module. The ceiling value at a seed sets the maximum number of animals that can be supported by the seed's social group in the population dynamics module. There are three underlying assumptions about social groups and the behaviour of dispersing individuals that are made by the spatial module:

1. Each social group of a population has a centre referred to as a seed.
2. After travelling some distance dispersing squirrel gliders will attempt to join the social group they are closest to - where the "closeness" of a group is the distance to its seed.
3. Individuals disperse on a network that links neighbouring seeds with shortest paths.

A dispersing glider that is closest to a particular seed than to any other will have a position in a region of other points also closer to that particular seed than to any other. This region is known as the Voronoi region associated with that seed. This means that each social group has a Voronoi region associated with it and the social group that a disperser will attempt to join will be the same as the one in whose Voronoi region it lays. Thus, the first two assumptions induce a theoretical structure on a patch which is made up of Voronoi regions and is related to how dispersing individuals perceive the arrangement of social groups. This structure also gives meaning to the notion of "neighbouring" seeds in the third assumption; neighbouring seeds are those whose Voronoi regions are adjacent. Note that a collection of Voronoi regions is known as a Voronoi diagram.

In Figure A.1(a) some hypothetical home ranges of some social groups are shown in a square habitat patch. The seeds of each social group are represented by black dots and the loop around each seed represents the edge of the social group's home range. In Figure A.1(b) the Voronoi diagram is shown for the seeds of the patch. In this diagram the home ranges of social groups have been abstracted and now appear as polygons; these are the Voronoi regions. In Figure A.1(c) the Voronoi regions have been coloured and adjacent Voronoi regions have been linked with shortest paths which, in this context, are straight lines.



(a) (b) (c)

Figure A.1. Some hypothetical Squirrel Glider home ranges are shown in (a); in (b) they are represented abstractly by a Voronoi diagram; in (c) the Voronoi regions have been coloured and the seeds of neighbouring regions have been linked with straight lines to form a hypothetical dispersal network.

The patch of habitat depicted in Figure A.1 has a simple square geometry. More complex patches may have regions of non-habitat (e.g. cleared farmland) embedded in them. If one assumes that such regions cannot be traversed by the Squirrel Glider (which is true for cleared agricultural land, and likely to be true for residential and industrial areas), then shortest paths between pairs of points may no longer be straight lines; some shortest paths will have to make detours around regions of non-habitat. A Squirrel Glider's closeness to a particular seed is then taken to be the length of a habitat-confined shortest path between itself and the seed. In this setting, the same three assumptions given above can be used again to induce a theoretical structure on a habitat patch and to establish a dispersal network. The structure in this more general context is known as the shortest path Voronoi diagram. Stewart and van der Ree (2009) introduce a Voronoi diagram that approximates this referred to as the q-grid Voronoi diagram; this Voronoi diagram also has a related network that can be used to represent dispersal paths. The spatial module generates a network formed from a q-grid Voronoi diagram. An example of a q-grid Voronoi diagram and its related network on the Thurgoona and Albury Ranges landscape is shown in Figure A.2; the image has been cropped to an arbitrary region of the landscape.

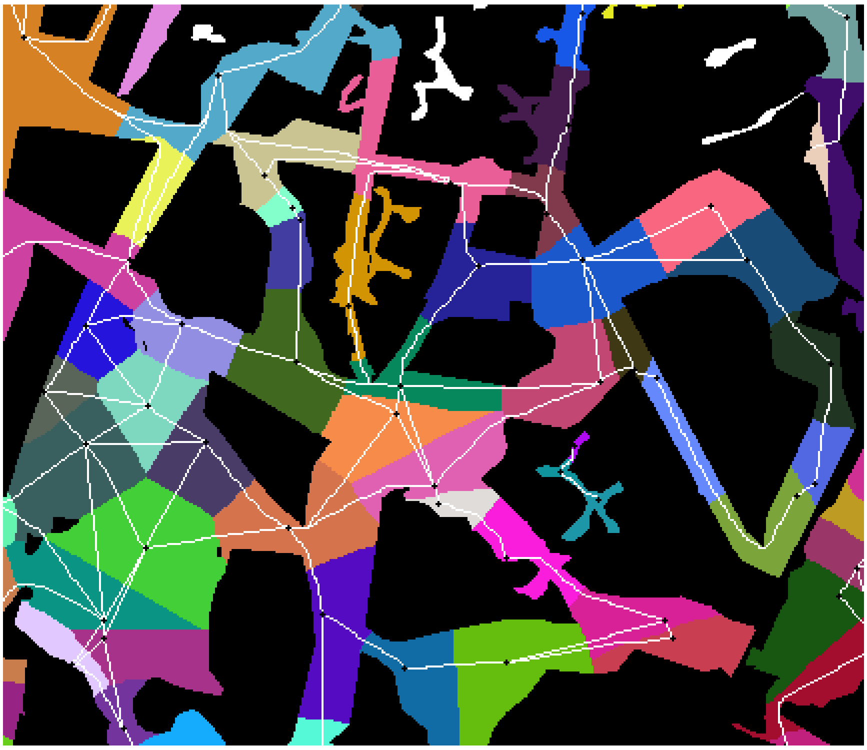


Figure A.2. A q-grid Voronoi Diagram for a part of the study area. The black areas represent regions of non-habitat (i.e. cleared agricultural land or residential land). Seeds are represented by small black dots, shortest paths are drawn as white lines, and the coloured areas represent the Voronoi regions.

The spatial arrangement of seeds for the network is determined by the region map, the K_density parameters and the social parameter inputs. Firstly, the number of seeds to be added is calculated, and this is equal to the carrying capacity divided by the social parameter. The carrying capacity is equal to the area of the low quality habitat region multiplied by the low K_density parameter plus the area of the high quality habitat region multiplied by the high K_density parameter. The seeds are then given random positions in the landscape. This random assignation does not follow a uniform distribution; instead it accounts for weights on the habitat types. Regions are weighted according to their contribution to the carrying capacity. Therefore, seeds are never assigned to the non-habitat and dispersal-only habitat regions as these do not contribute to the carrying capacity.

Once the seed positions have been determined, the spatial module generates a q-grid Voronoi diagram and a network that links neighbouring seeds. The ceiling value at each seed is then calculated as a weighted sum which accounts for the habitat types that its Voronoi region overlaps with. Voronoi regions may overlap with dispersal-only habitat, low or high quality habitat, but not with non-habitat. A K_density of zero is assigned to the dispersal only habitat while low and high K_densities are assigned to the low and high quality habitat regions respectively (as described in the "Model Parameters and Inputs" section). By multiplying the area of each region of overlap with its corresponding K_density and by summing over all the regions of overlap, a ceiling value is calculated for the seed. The survival at each seed is also determined by a weighted sum, but this time, in addition to habitat types, survival zones are also taken into account. The positions of the survival zones in the landscape are determined by the "other-land-use map" (Figure 2(b)) and the survival values associated with the zones are determined by the survival parameters (as described in the "Model Parameters and Inputs" section).

Population Dynamics Module

The population dynamics module outputs mean population densities at yearly intervals averaged over 100 population dynamic projections. The way each projection is generated will be described here. Each projection starts with an initial state with the number of animals in each social group equal to half the social group's ceiling value (rounded to the nearest whole number) and with the Squirrel Gliders' sex determined at random (50% chance of being either male or female). Social group sizes are then determined at yearly intervals following birth, death and dispersal events. These events have random components and therefore, to simulate the necessary random variables, a random number generator is used.

In a birth event, a female will choose a non-related male at random from her social group to mate with. A male and female are considered non-related if they are not full siblings and if neither is a parent of the other. In the initial state all males and females are set to be non-related. Once a female chooses a mate she will produce either 1 or 2 offspring with the chance of each alternative determined by the birth number (see "Model Parameters and Inputs" section for details). The sex of an offspring is determined at random (50% chance of being either male or female). If no mate is available for a female (i.e. there are no males, or all males are related to her), then she will produce no offspring.

Following the birth events, the death events take place. Every animal has a chance of dying. The probability that an individual will die is determined by the survival value allocated to its social group; specifically, it is 1 - (social group's allocated survival value).

After this, the surviving juveniles (i.e. surviving animals born that year) then disperse. Dispersing individuals start from their natal social group's seed on the network created by the spatial module. They then choose a link at random from those connected to the seed and disperse along that link. When a new seed is reached a new link is chosen at random and the dispersal continues along that link. This process of moving along links between seeds continues and dispersal ends when an individual has covered the dispersal distance set for it. At this point (which may occur when the disperser is between two seeds) the disperser attempts to join the social group it is closest to (i.e. the one in whose Voronoi region it lays).

Each social group is only permitted to hold a number of animals set by its ceiling value. If a social group is at its ceiling capacity then no dispersers can join it and those attempting to do so are made to die. If on the other hand a social group is below its ceiling capacity then it can take in a maximum number of dispersers equal to the number that will raise its social group size to its ceiling capacity. If there are more dispersers than this maximum number attempting to join, then dispersers are admitted one by one and at random until the social group size reaches its ceiling capacity and the surplus individuals are made to die. If the number of dispersers attempting to join is less than or equal to the maximum number able to join, then all the dispersers are admitted.

Variation in environmental conditions may contribute to fluctuations in Squirrel Glider population size. For example, periods of drought may affect the flowering of trees that are important in the Squirrel Glider's diet and this in turn could affect birth and death rates and hence population size (Sharpe 2004). Environmental variation is incorporated into the population dynamics module by randomly varying the birth number and mortality probability in the birth and death processes. A random variable from a normal distribution is simulated each year. Its distribution is set with a mean of 0 and standard deviation equal to the environmental stochasticity parameter divided by 100. Variation is added to the birth process by creating a temporary birth number each year that is used in the birth process for that year. The temporary number is equal to the original birth number *plus* the normal random variable multiplied by the original birth number. Variation is added to the death process in a similar way, but this time a temporary mortality probability is set equal to the original mortality probability *minus* the normal random variable multiplied by the original mortality probability. Truncation is used to keep the temporary birth number between 1 and 2 and the temporary mortality probability between 0 and 1. Note that the same random variable is used for both birth and death processes in the same year. This ensures that the effect of the environment on births is correlated with the effect on deaths. Therefore in a good year, for example, the birth rate will increase and the death rate will decrease. This will result from a positive value of the random variable simulated for that year. The environmental stochasticity parameter can be thought of as controlling the strength of the environmental variation.

APPENDIX B

How the Endpoint Comparisons Relate to Sensitivity Analysis

The endpoint comparisons can be thought of as a basic type of sensitivity analysis. A sensitivity analysis is "the study of how uncertainty in the output of a model … can be apportioned to different sources of uncertainty in the model input" (Saltelli *et al*. 2004). It involves running the model over different combinations of input parameter values. Usually each input parameter is restricted to an interval of interest and in our endpoint comparisons we refer to these as exploratory intervals (e.g. 0.2 to 0.4 for road zone survival is an exploratory interval). The results of a sensitivity analysis can be used and interpreted in different ways: they can give an overall indication of the model's behaviour; they can help determine the "robustness" of a model outcome by quantifying the surrounding uncertainty; they can help prioritise the efforts of future empirical studies to focus on getting accurate estimates of those parameters with the largest impact; and in some cases they can directly inform management decisions by highlighting which parameters would, if changed through management, have the most influence of the system being modelled. However, the extent to which a sensitivity analysis can be used and interpreted depends largely on the width of the exploratory intervals used.

In some cases, different widths of exploratory intervals may be appropriate for different uses of a sensitivity analysis. For example, if the "robustness" of a model is being investigated and the uncertainty in the model's output is being quantified, then each interval should reflect the uncertainty that "… derives from a lack of knowledge about the appropriate [parameter] value to use …" (Helton *et al*. 2006). This means that if there is a lot of confidence from experts for a particular parameter, then a thin exploratory interval should be used. On the other hand, if the analysis is being used to directly inform management, then wider exploratory intervals may be more appropriate. This is because a management decision could be used to "shift" a parameter value and the exploratory interval should be wide enough to account for such a shift. Similarly, when investigating model behaviour wider intervals may be the most useful; however, when prioritising efforts for empirical studies, intervals should be the same as those used for addressing robustness, that is, they should represent the "state of knowledge" surrounding each parameter value (Helton *et al*. 2006).

Some times when exploratory intervals are set for the purpose of investigating model robustness and for prioritising future empirical study efforts, the intervals may be wide enough so that the results of the sensitivity analysis can also be used to inform management decisions and give insight into model behaviour. In fact, in some cases it may be possible to choose appropriate intervals, by design, that satisfy the many needs for a sensitivity analysis. This may mean setting wide intervals for parameters whose value are well known in order to adequately explore model behaviour and the potential for management strategies. Choosing intervals to serve multiple purposes can be important when model runs are computationally expensive and it is only feasible to complete one set of runs for a sensitivity analysis.

One aspect of model behaviour that the endpoint comparisons can give some insight into is monotonicity. A model is monotonic over a particular parameter interval if the model output (which we assume here to be a single scalar value) either increases or decreases with an increase in the parameter value. This means that the value of the model output cannot move up and down over the interval of a particular parameter and the maximum and minimum values of the output must be reached at the endpoints. In the endpoint comparisons if the model has a value that is either greater than both the endpoint evaluations or less than both the endpoint evaluations when it is evaluated at the midpoint of a parameter interval, then the model is not monotonic. If on the other hand the midpoint evaluation lies in between the endpoint evaluations, then it could be that the model is monotonic. In this later case, the results will not conclusively support monotonicity since the model is only ever evaluated at three points on each parameters interval and we cannot be sure how the model will behave in between these points; however, such results would be consistent with monotonicty.

Aside from monotonicity, other aspects of model behaviour cannot be investigated using endpoint comparisons. In particular, interactions between parameters cannot be detected. Nevertheless, population models are likely to be simple systems with few interactions and therefore the endpoint comparisons offer a simple and effective way to examine sensitivity. One instance, however, where interactions could occur, might be between different landscape configurations and survival values. For example, population density may only reach a high level when survival is high and a particular landscape configuration is present. Whilst we did look at the effect of creating joins in the landscape, we did not do this in the context of a sensitivity analysis and we did not look for interactions. Current sensitivity analysis techniques are primarily designed to deal with single scalar value outputs and examining the effect of using different landscape configurations on population density is best done with more complex outputs such as density maps. Sensitivity analysis techniques have not been established to deal with such outputs. It would be worthy of further study to develop techniques for this purpose, however it is beyond the scope of this study.

APPENDIX C

Field Survey

Trapping was used to determine the presence or absence of Squirrel Gliders at some sites and their density at other sites. A total of 13 sites were selected from across the precincts defined in the Thurgoona and Albury Ranges Threatened Species Strategies (Davidson, Datson et al. 2004; Davidson, Datson et al. 2005). Trapping was conducted in November and December 2007.

Trapping to assess density of gliders was conducted at five sites (Table 7) that were known to have resident Squirrel Glider populations (this information was obtained from the results of previous trapping within the region). Three of the sites, Bell's TSR, Old Sydney Rd, and Mitchell Park were from the Thurgoona area, and two from the Albury Ranges region, Urana Rd, and Mr Brown's. At each site, nine traps were set in a (3x3) grid formation (traps approximately 100 m apart) where possible with all density sites covering approximately the same area to allow for direct comparison between sites. Due to the often irregular shape of remnant habitat patches, a 3 x 3 grid was not possible at all sites. In these instances, every effort was made to keep the total site area consistent. All capture rates were converted to a per ha estimate, assuming a 50 m buffer (i.e. half the distance between traps) around the perimeter of the grid. All density sites were trapped for between 7 and 9 nights.

Trapping to determine the likely presence or absence of Squirrel Gliders was conducted at presence-absence sites (Table 7). Areas that had not previously been trapped for Squirrel Gliders as well as areas that were of particular interest (conservation or development) were selected to increase knowledge of Squirrel Glider distribution. Seven presence-absence sites were selected from within the Albury Ranges Region (Nail Can Hill Reserve, Olympic Way, Bungambrawatha Creek, Central Reserve Road, Centaur Road and Pearsall Street), and one site at Thurgoona Drive, was selected from the Thurgoona Region. An additional site was set up at Bowna Reserve near Lake Hume, outside the precincts defined by the Thurgoona Threatened Species Conservation Strategy.

At each presence-absence site 5 – 6 traps were set in large or hollow-bearing trees at intervals of 20 – 50 m. The distance between traps and the number of nights varied among sites because the aim was to detect gliders, and not to measure abundance or density. Previous studies have shown that if no Squirrel Gliders are captured after four nights, it is unlikely that they are present in the habitat patch at that point in time. On this basis, all sites were set for four to five nights, and were taken down as soon as a Squirrel Glider was detected. The only exception was at Bungambrawatha Creek, where a trap was vandalised on the first night. The site was taken down immediately due to concern for the welfare of any trapped animals.

At all sites, wire cage traps (20cm x 20cm x 50cm) were set 4 – 6 m high on the trunk of tree, and baited with a mixture of honey, peanut butter and rolled oats. Diluted honey water was sprayed on the trunk above the trap to act as a further attractant. Processing of captured animals involved recording weight, sex and reproductive condition and tooth wear of the upper incisor to determine approximate age. All captured Squirrel Gliders were given unique ear tattoos and microchips for individual identification. Small (2mm diameter) tissue samples were also taken from ear margins for use in genetic analysis as part of further studies. All captured animals were released at point of capture immediately after processing.

Trapping results

From 28th November to 7th December 2007, 44 Squirrel Gliders were captured 85 times over 499 trap nights (Table C.1). Other species captured were 10 Common Brushtail Possums, captured 13 times, as well as a single Sugar Glider and Yellow-footed Antechinus, both captured only once.

Squirrel Gliders were detected at all density sites, with a total of 41 Squirrel Gliders captured over 369 trap nights. The highest number was found at Bell's TSR (11 individuals) and the lowest at Mitchell Park (7 individuals). Common Brushtail Possums were less common (9 individuals in total), and were only detected at Urana Road, Mitchell Park and Old Sydney Road. One Yellow-footed Antechinus was also detected at Urana Road.

Three Squirrel Gliders were trapped once each at three of the presence-absence survey sites during 130 trap nights. The three presence sites were Nail Can Hill Reserve, Bowna Reserve and Central Reserve Road. A single Common Brushtail Possum was also detected at Bowna Reserve, and one Sugar Glider was captured at Central Reserve Road.

In total, 29 female and 15 male Squirrel Gliders were captured during this survey, including two juvenile males. All of the females captured were reproductively active with 11 carrying pouch young, 14 lactating (meaning they had young left in the nest) and four had recently bred. This level of activity is consistent with the normal breeding season for Squirrel Gliders.

Table C.1. Trapping effort and trapping results for density and presence-absence sites. A number followed by the symbol "M" indicates the number of males found at the site; similarly "F" indicates females and "U" animals with unknown sex. The symbol "-" indicates no captures and "*" indicates a site where not all traps were open all nights.

Site	# Traps (# nights)	Total trap nights	Squirrel Gliders	Common Brushtail Possums	Sugar Glider	Yellow-footed Antechinus
Density sites						
Bell's TSR	9 (9)	81	3 M, 8 F	-	-	-
Old Sydney Rd	9 (8)	72	3 M, 5 F	1 M, 1 U	-	-
Mitchell Park	9 (8)	72	2 M, 4 F	1 M, 1 F, 1 U	-	-
Mr. Browns	9 (7)	63	3 M, 4 F	-	-	-
Urana	9 (9)	81	4 M, 5 F	2 M, 2 F	-	1 M
Presence-absence sites						
Thurgoona Dve	5 (4)	20	-	-	-	-
Olympic Way	5 (5)	25	-	-	-	-
Central Reserve Rd	5 (2)	10	1 F	-	1 M	-
Bungambrawatha Ck	5 (1)	5	-	-	-	-
Centaur Rd	5 (4)	20	-	-	-	-
Pearsall St	5 (4)	20	-	-	-	-
Nail Can Hill Reserve	5 (2)	10	1 F	-	-	-
Bowna Reserve	6 (4)	20*	1 F	1 F	-	-
Total		499	44	8	1	1

APPENDIX D

Possible Sources of Error in the Modelling Process

Many sources of error could account for differences between the observed field results and the model predictions (Table 7). Some of the reasons are: the simplifying assumptions and the model's structure may be incorrect; model parameter values may be wrong; field results may not be exact; if nature is regarded as a stochastic system, then it might not (and is likely not to) be following its mean trajectory; and finally the wrong point in the models trajectory might be being used for the comparison - this might happen, for example, if one compares a model's equilibrium outputs with a natural population that is not in equilibrium. If the model, its parameter values and the field data are all reasonable, and if the natural population is in equilibrium, then one might expect errors between observed and predicted results to be primarily a result of the inherent stochasticity of the system.

Although not explicitly investigated in this report, it is likely that having regions connected in the wrong way on the input map could significantly affect the predicted population size. An indication of this was given in the results section. It was shown that making joins in the landscape could increase population size; hence, if it is assumed on the input map that areas are connected when in reality they are not, then population viability may be overestimated. This source of error might explain the discrepancies between the predicted densities and observed densities at the Centaur Rd and Pearsall St sites (Table 7). Figure D.1 shows the input region map and an aerial photo of these sites (which have been circled). As can be seen the links assumed in the input map appear to be tenuous in the photo and may not actually be links. On the other hand, the input map might assume that the landscape is more fragmented than it actually is. At Bowna Reserve (Fig. D.2), the model predicted a low density (0.21 animals per ha, Table 7) at year 100 for the best estimate scenario. Such a low density might go undetected in the field, however the field data observed a "presence" at this site. The model is treating the region around this site as many isolated patches when they may be operating as one large patch. Hence the model density may be underestimating the true density at this site.

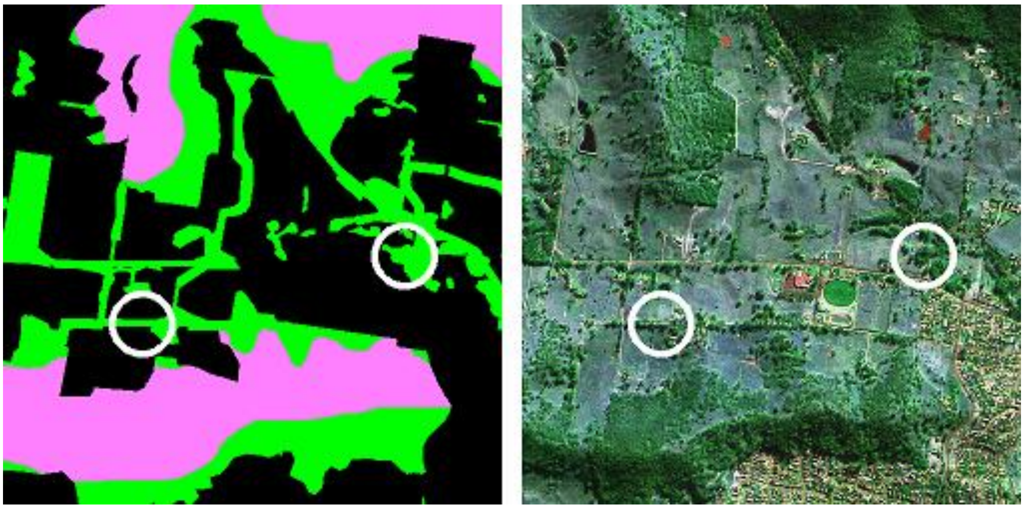


Figure D.1. Input region map and corresponding aerial photograph for Centaur Rd (left circle in each diagram) and Pearsall Rd (right circle in each diagram) sites.

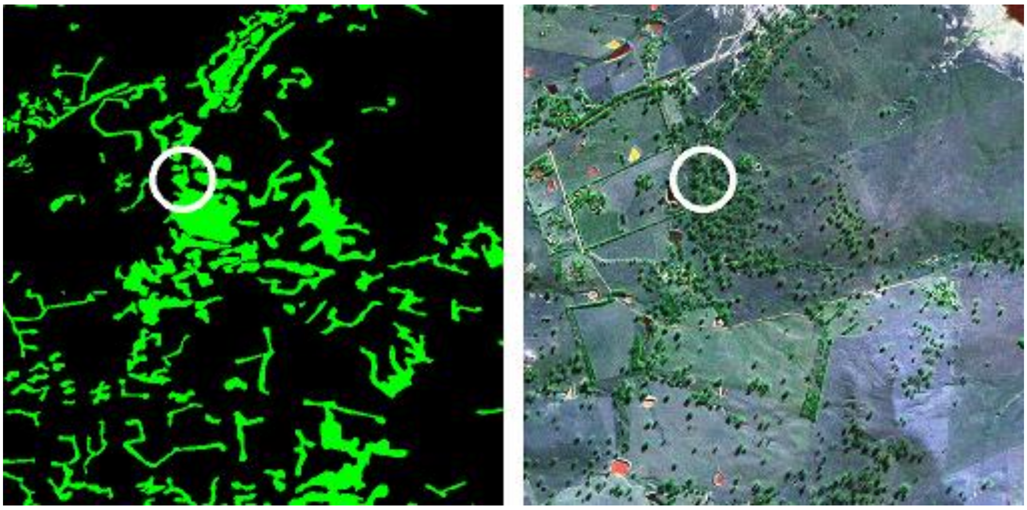


Figure D.2. Region input map and corresponding aerial photograph for the Bowna Reserve site. The site location has been circled.

Another source of error could arise from input maps if the density regions are incorrectly defined. For example, areas we classified as dispersal-only habitat may actually support a high density of gliders. This may be the case at Urana Rd, where the field surveys found a moderately high density of 1.0 gliders per ha (Fig. D.3). Similarly, this may also be the case for the site at Central Reserve Road where a "presence" was recorded (Fig. D.4), however this may just be an individual detected at the site while dispersing. Additional repeat surveys would be required to clarify the status of the population at these types of sites.

Figure D.3. Region input map and corresponding aerial photograph for the Urana Rd site. The site location has been circled.

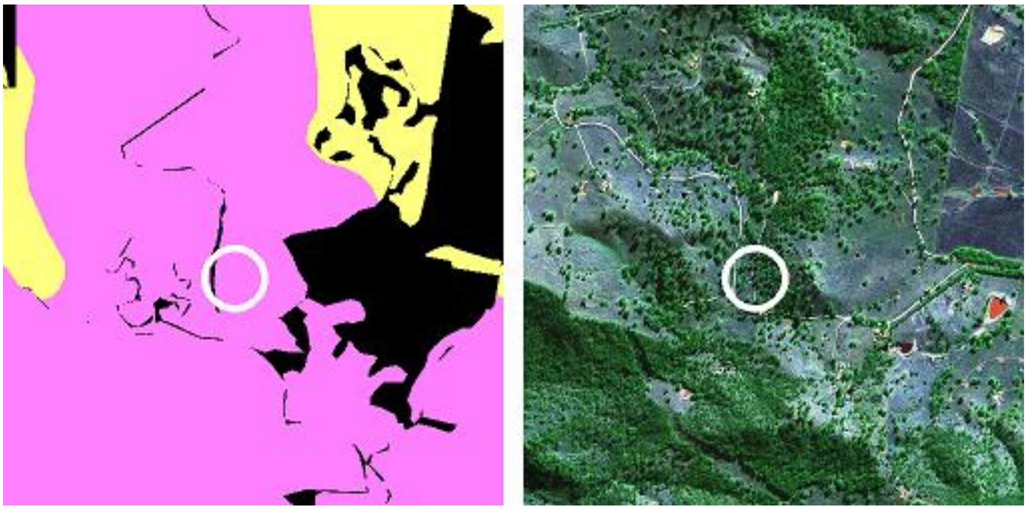


Figure D.4. Input region map and corresponding aerial photograph for the Central Reserve Rd site. The site location has been circled.

APPENDIX E

Monitoring

A long-term monitoring strategy would be beneficial to track changes in population density and distribution, as well as detect declines in the population which can be reversed before it gets too advanced. The aim of the monitoring would be to follow trends in population size, as well as quantify key parameters, such as birth and death rates. The monitoring program should census the Squirrel Glider population at a range of sites. The actual sites selected will depend upon the specific objectives of the monitoring program. For example, if the objective was to monitor population size, then once per year at approximately 15 sites would probably suffice. Rates of survival could also be estimated simultaneously with population density. Reproductive output can be assessed during one survey per year, provided it takes place during winter/spring. If population distribution is required (i.e. presence/absence), then somewhere in the order of 50 sites would need to be surveyed once every 1 to 2 years. The birth and death rates and population density, could not be estimated using this approach, because as soon as Squirrel Gliders are detected at a site, the survey would cease.

Long-term monitoring programs need to be carefully designed to ensure that they meet the initial objectives. Many long-term monitoring programs fail to deliver the required outcomes because of poor design. The specifics of the long-term monitoring at Thurgoona can only be designed when the level of funding available for the monitoring is known. However, the following principles apply: (1) specific goals/questions need to be identified; (2) the study design is such that there is a high probability that the questions can be reliably and confidently answered; (3) data is analysed and reported regularly; (4) long-term funding and support is secured.

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