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Guidelines for managing the endangered Growling Grass Frog in urbanising landscapes

Geoffrey Heard, Michael Scroggie and Nick Clemann

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Guidelines for managing the endangered Growling Grass Frog in urbanising landscapes

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Background: High-quality Growling Grass Frog habitat, Merri Creek, Victoria.

Inserts: Growling Grass Frogs (*Litoria raniformis*) from Donnybrook and Bundoora, Victoria.

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Summary

The Growling Grass Frog (*Litoria raniformis*) is a large, semi-aquatic tree-frog that is distributed widely across southern Australia, including eastern South Australia, Victoria, Tasmania, southern New South Wales and (formerly) the Australian Capital Territory. Despite being once widespread and abundant, *L. raniformis* is today recognised as threatened in all states in which it occurs, and considered nationally vulnerable to extinction.

While historical perturbations have been important factors in this decline (e.g., droughts and the introduction of exotic disease), habitat loss, fragmentation and degradation continue to be significant issues for the conservation of *L. raniformis*. Of particular concern in this regard are the many remnant populations of the species that occur in urbanising landscapes, such as on the fringe of Melbourne in southern Victoria. It is now clear that the ongoing decline of *L. raniformis* in these landscapes is driven by the outward spread of urban development, and that this decline will continue unless sound conservation programs are enacted.

These guidelines are designed to facilitate the development of such conservation programs. Based on nearly a decade of research on the population dynamics and habitat requirements of *L. raniformis* on the urban fringe of Melbourne, they describe wetland-level and landscape-level objectives for habitat management, and outline protocols for surveys aimed at determining wetland occupancy by the frog. The former is crucial for the planning of conservation reserves, because it can inform decisions about the prioritisation of wetlands to be protected, enable identification of enhancement opportunities at existing wetlands, and guide the design of dedicated artificial wetlands for the frog. Survey protocols are also of vital importance, because accurate assessments of wetland occupancy are needed to plan habitat management and monitor the success of such initiatives.

In addition to these core objectives, the guidelines also broach some secondary issues for the management of *L. raniformis* in urbanising landscapes. Techniques other than occupancy-based surveys are reviewed as a means of prioritising and monitoring habitat management actions, including surveys aimed at establishing the reproductive success of remnant populations, and mark-recapture and radio-telemetry for examining population size, survival and dispersal rates. Two experimental approaches to management are also considered: underpasses under roads to facilitate dispersal, and translocation of populations from wetlands ear-marked for destruction. The potential effectiveness of these approaches is reviewed, and advice provided on the context in which they should be applied.

These guidelines are aimed at the broad range of individuals and organisations involved in conservation of *L. raniformis* in urbanising landscapes, including both public land managers and conservation agency staff, individuals and organisations involved in urban development (developers, consultants, etc.), and private land holders who may wish to implement conservation initiatives for the species on their properties. In doing so, the guidelines aim to promote a scientifically validated and consistent approach to the species' management in these landscapes.

1 Introduction

1.1 The Growling Grass Frog

1.1.1 An ecological overview

The Growling Grass Frog (*Litoria raniformis*) is a member of the 'bell frog' species complex (Anura: Hylidae), a group of morphologically and ecologically similar frogs found across the temperate regions of southern Australia (Barker et al. 1995). The species inhabits lowland environments in the south-east of the continent, including (or formerly including) parts of the Australian Capital Territory, New South Wales, Victoria, South Australia and Tasmania (Barker et al. 1995).

Litoria raniformis is a largely aquatic species that occupies a variety of permanent and ephemeral wetlands, including slow-flowing sections of rivers and streams, lakes, swamps, billabongs and ponds (Pyke 2002). It is also known to inhabit artificial wetlands such as irrigation canals, rice growing bays, water-filled quarries, farm dams and water treatment ponds (Pyke 2002).

Broadly speaking, the morphology of *L. raniformis* is 'big' and its life-history is 'fast'. The species breeds aquatically, and may produce clutches of eggs numbering in the thousands (Germano and White 2008). Tadpoles grow quickly and can reach over 100 mm in length (Bell 1982). Although they may over-winter and emerge in the following spring (Pyke 2002), tadpoles typically metamorphose after only 2 or 3 months. Post-metamorphic growth rates are also rapid, and both sexes can reach sexual maturity within 4 months of metamorphosis (G. Heard unpubl. data). On average, males grow to about 73 mm (snout–vent length), and females to about 86 mm (G. Heard unpubl. data).

As with other members of the bell frog complex, *L. raniformis* is mainly active during the spring and summer, and may be active during both the day and the night (Pyke 2002). Seasonal patterns of mate-calling by males indicate that reproductive activity begins in September, and continues until January or February (Pyke 2002). Foraging appears the dominant activity in late summer and into autumn: individuals are often located in apparent 'ambush' positions at the waterline or in nearby terrestrial areas during this time of year (Heard et al. 2008a). Activity wanes as the temperature decreases. Torpor occurs during the coolest months of the year, at which time individuals may be located beneath cover such as rocks, logs and vegetation close to water (Pyke 2002).

The diet of *L. raniformis* has not been studied in detail, but available data suggest it is similar to that of the closely related Green and Golden Bell Frog (*L. aurea*). As noted above, *L. raniformis* displays an 'ambush' or 'sit-and-wait' foraging strategy to capture prey, both in the water when floating amongst aquatic vegetation, and when on land (Heard et al. 2008a). Like *L. aurea*, the species appears to be a generalist carnivore, consuming a variety of invertebrates and small vertebrates (Pyke and White 2001). A notable dietary trait of both species is the propensity to devour tadpoles and metamorphlings of their own species, as well as other frogs (Pyke and White 2001; Pyke 2002). This trait is perhaps a means of fuelling the rapid growth and great fecundity that these frogs display.

1.1.2 Conservation status

Once widespread and abundant, *L. raniformis* is now listed as 'endangered' on the IUCN Red List 2009, and 'vulnerable' under the Commonwealth *Environment Protection and Biodiversity Conservation Act 1999*. It is also listed as 'threatened' in each state in which it occurs (Clemann and Gillespie 2007): in South Australia under the *National Parks and Wildlife Act 1972*, in Victoria under the *Flora and Fauna Guarantee Act 1988*, in Tasmania under the *Threatened Species Protection Act 1995*, and in New South Wales under the *Threatened Species Conservation Act 1995*.

The conservation status of *L. raniformis* became of concern only relatively recently, following a marked contraction in its distribution during the latter half of last century (Mahony 1999). The decline appears to have been ubiquitous across the frog's range, and is likely to have been caused by a synergy of threatening processes. For example, populations of *L. raniformis* on the Southern Tablelands of New South Wales and the Australian Capital Territory disappeared completely sometime between 1978 and 1981 (Mahony 1999). These populations were subject to habitat destruction, fragmentation and alteration throughout the preceding century as agricultural practices expanded in the region, but were probably exterminated by the combination of a severe drought and the introduction of the exotic pathogen *Batrachochytrium dendrobatidis*, which spread across eastern Australia at roughly this time, and caused declines or extinctions of numerous Australian frog species (Mahony 1999).

1.1.3 Urbanisation as a key threatening process

Urbanisation is the spread of urban infrastructure (housing, industrial estates, roads, etc.), and is usually envisaged in terms of the conversion of land previously used for agriculture or dominated by natural environments (Forman 2008). It is driven by human population growth in cities, and regional towns and centres (Forman 2008).

As is the case for many endangered amphibians world-wide (Hamer and McDonnell 2008), urbanisation is thought to be an important threatening process for *L. raniformis*. There are two main reasons for this. Firstly, recent studies suggest that *L. raniformis* is highly sensitive to urbanisation. Working on the northern urban fringe of Melbourne, Robertson et al. (2002) documented the apparent extinction of numerous populations of the frog that were historically known from this region. Heard and Scroggie (2009) expanded on this work, resurveying 123 populations known from the same area prior to 2006. They found only 53 extant populations (43%). In both studies, most extant populations were located outside the urban boundary – a pattern that matches the broader contemporary distribution of *L. raniformis* around Melbourne (Figure 1). Secondly, many remnant populations of *L. raniformis* occur in areas that are being urbanised, or are likely to be urbanised in the near future. The many remnant populations on the urban fringe of Melbourne (Figure 1) are of particular concern in this regard, because almost all occur in areas designated for future urban growth.

1.1.4 Past research on the impacts of urbanisation

The threat that urban development poses to the conservation of many remnant populations of *L. raniformis* has motivated recent research on the impacts of urbanisation on the species, and the means by which those impacts may be mitigated. The guidelines presented here are based upon one of these projects, which entailed compiling and analysing monitoring data for the frog collected over six years at 167 wetlands in the Merri Creek corridor on Melbourne's northern outskirts (Heard and Scroggie 2009). The objective of that research was to develop an understanding of the population structure and dynamics of the species in urbanising landscapes.

The first key outcome from this work was the realisation that *L. raniformis* displays 'classical metapopulation dynamics' in the Melbourne region. A metapopulation is 'a set of discrete populations of a species that are connected by migration' (Hanski 1999). 'Classical metapopulation dynamics' refers to situation in which the constituent populations blink in and out of existence, given frequent population extinction and recolonisation (Hanski 1999). Thus, for a species to be described as displaying classical metapopulation dynamics, its populations must be spatially discrete yet exchange some migrants, and the occurrence of populations must change through time. This is precisely the scenario observed for *L. raniformis* in the Merri Creek corridor (Heard and Scroggie 2009). Mark–recapture studies demonstrated that individual wetlands (pools along streams, farm dams, quarries, etc.) support discrete populations of the frog, but that some individuals disperse between wetlands, and hence, between populations. Annual monitoring of wetland occupancy demonstrated frequent population extinction and recolonisation, and therefore, a temporally dynamic pattern of population occurrence.

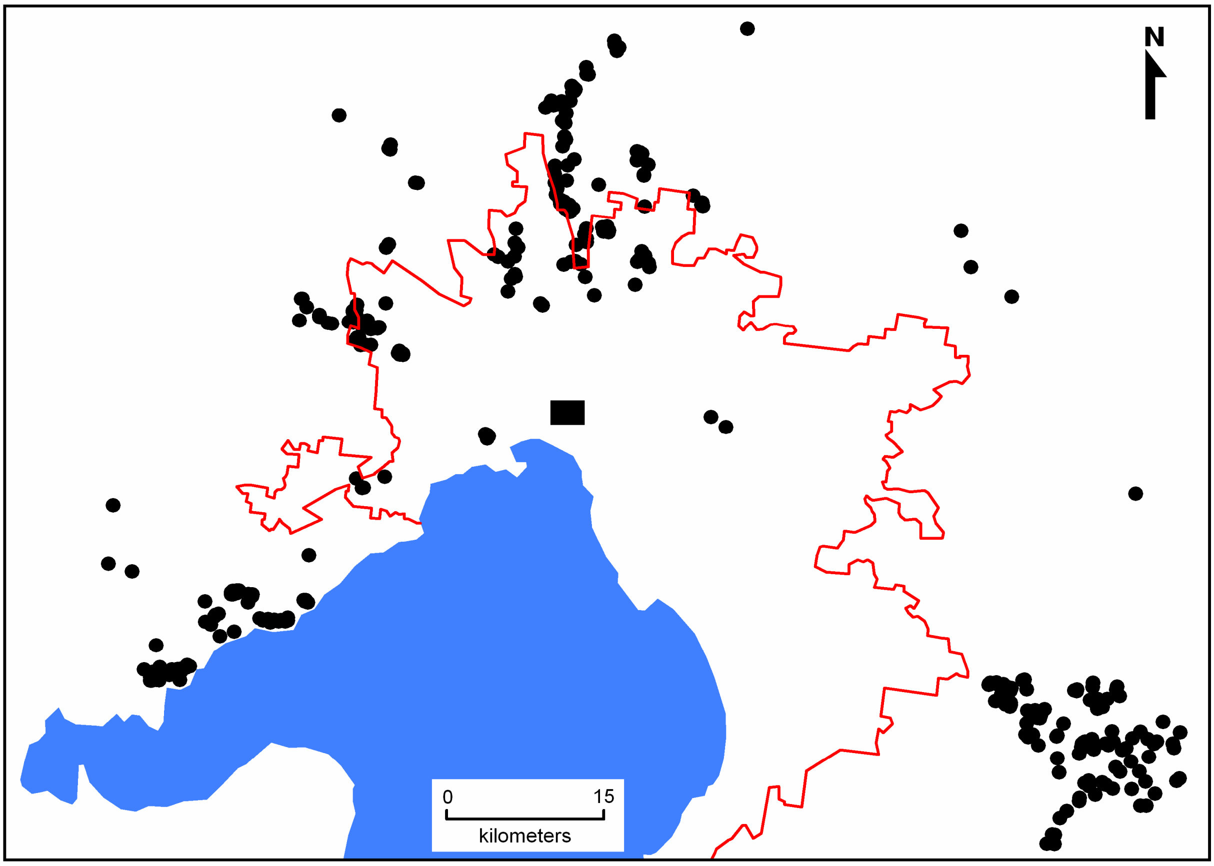

Figure 1. The contemporary distribution of *Litoria raniformis* around Melbourne. Known populations of the frog are represented by the black dots. The black rectangle represents the Central Business District of the city, and the red-line is the current extent of urban development. The blue shaded area is Port Phillip Bay. Only records from 2000 onwards are displayed. Source: Atlas of Victorian Wildlife, 2008.

The second key outcome of this research was the identification of the drivers of population extinction and recolonisation, and the quantification of these relationships. Metapopulation theory considers population extinction to be a function of habitat patch characteristics and the proximity of the population to its neighbours. The idea is that populations inhabiting large and/or high-quality habitat patches will have higher rates of recruitment and survival, and will therefore be more resilient to environmental or demographic stresses. Furthermore, populations that are close to other populations (i.e., those that display high connectivity) will be bolstered by immigration, so they can resist extinction even if they display low recruitment or survival rates. Likewise, the theory predicts that vacant patches close to extant populations will have a higher chance of recolonisation, because they are more likely to receive migrants. Heard and Scroggie (2009) used the extensive monitoring dataset they collated to test these relationships for *L. raniformis* in the Merri Creek corridor. As expected from the theory, they found the annual probability of extinction for populations of the frog to be strongly negatively related to wetland hydroperiod, aquatic vegetation cover and connectivity, and the probability of recolonisation of vacant wetlands to be strongly positively related to connectivity (Table 1, Figure 2). Thus, populations of *L. raniformis* that inhabit permanent wetlands with high aquatic vegetation cover and which are close to other populations of the frog not only have a substantially higher chance of persistence from year-to-year, but are also more likely to be recolonised in the event that extinction does occur.

Table 1. Definitions for the three wetland-level and landscape-level variables previously found to influence the extinction and recolonisation probabilities of *Litoria raniformis* in the Merri Creek corridor.

Variable	Definition
Hydroperiod (Wetland-level)	The likelihood that an individual wetland will remain inundated over the course of a single breeding season, on an ordinal scale from 0 to 3, where: 0 = fills only in years with above average rainfall ('intermittent') 1 = fills and dries out annually with average rainfall ('ephemeral') 2 = dries out only during years of below average rainfall ('semi-permanent') 3 = never dries out, regardless of rainfall ('permanent')
Aquatic vegetation cover (Wetland-level)	Mean percentage of the water surface area covered by the foliage of emergent, submergent and floating aquatic plants*, where each may range between 0–100%: = SUM(%emergent + %submergent + %floating)/3
Connectivity (Landscape-level)	Distance weighted number of populations within a 1000 m radius: = SUM($W_j + W_k + W_l + ... + W_x$) where W_j is the inverse of the straight-line distance (≤ 1 km) between the focal population and population *j*.

* Emergent vegetation is any aquatic macrophyte rooted below the water with foliage emerging above the water-surface, submergent vegetation is any aquatic macrophyte rooted below the surface with foliage below the water-surface, and floating vegetation is any rooted or unrooted aquatic macrophyte or alga with foliage floating on the water-surface.

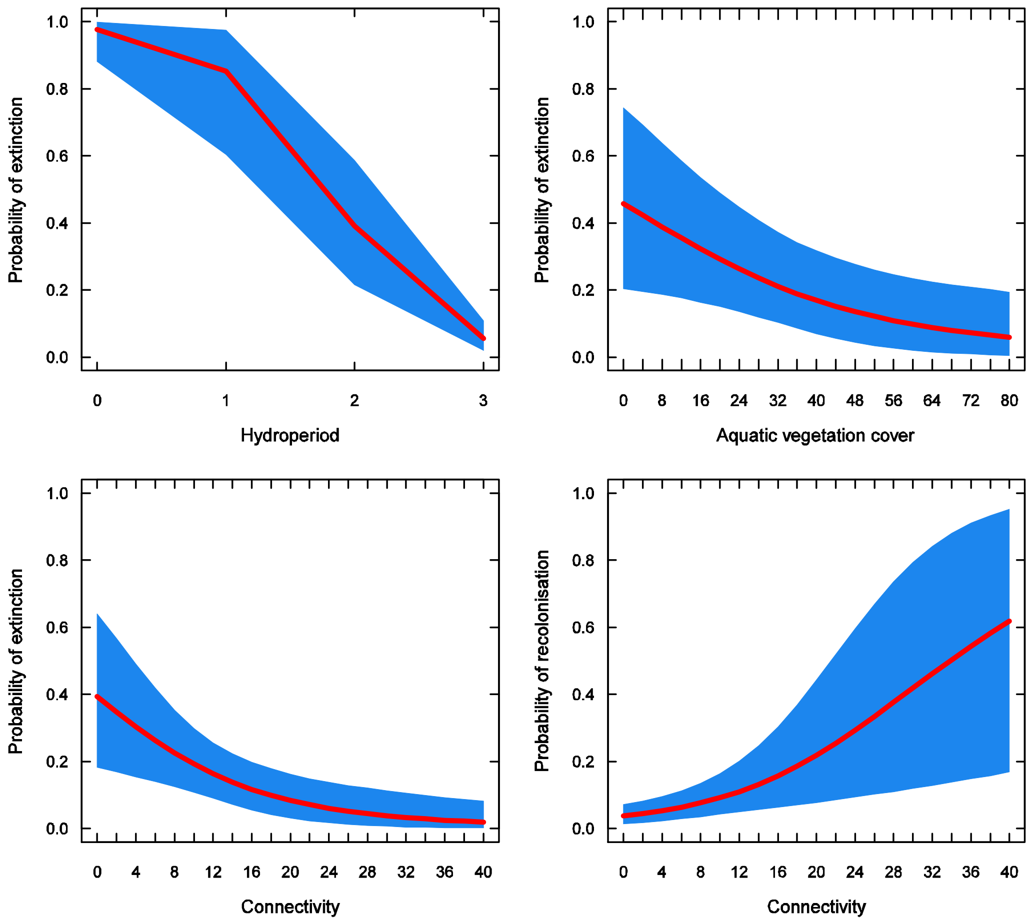

Figure 2. Relationships between the three wetland-level and landscape-scale variables previously found to influence the extinction and recolonisation probabilities of *Litoria raniformis* in the Merri Creek corridor. See Table 1 for definitions of the variables. The red lines represent the estimated mean relationships, and the blue shading the 95% credible intervals of the estimates. Relationships are shown with all other variables held at their mean. The upper limits for aquatic vegetation cover (80%) and connectivity (40) are based upon those observed by Heard and Scroggie (2009).

These findings provide clear direction for initiatives aimed at mitigating the impacts of urbanisation on *L. raniformis*. They demonstrate that (i) the likelihood of extinction for remnant populations can be minimised by maintaining (or enhancing) wetland hydroperiods and aquatic vegetation cover, and (ii) that the long-term persistence of populations requires they be part of a dense network of populations, within which migration and recolonisation can occur. However, the ramifications of this work extend well beyond these simple recommendations.

Firstly, in quantifying relationships between wetland-level and landscape-level variables and both the probability of population extinction and recolonisation for *L. raniformis*, this work provides a basis for developing very specific guidelines for managing the frog's habitat. There are several means of doing this, but a particularly straightforward approach is that offered by the notion of an 'equilibrium probability of occupancy' for populations within a metapopulation (Hanski 1994; MacKenzie et al. 2006; Ferraz et al. 2007; Martin et al. 2009). The equilibrium probability of occupancy can be interpreted as 'the chance that a population will persist at a given locality at any point in the future'. Its calculation begins with the premise that the probability of occupancy in one year is a function of the probability of occupancy in the preceding year, and the time-specific probabilities of extinction and recolonisation:

$$\psi_{t+1} = \psi_t \times (1 - \varepsilon_t) + (1 - \psi_t) \times \gamma_t, \qquad \text{(equation 1)}$$

where ψ_{t+1} is the probability of occupancy in the second year, ψ_t is the probability of occupancy in the first year, and ε_t and γ_t are the probabilities of extinction and recolonisation in the first year, respectively. The equilibrium probability of occupancy is derived from this equation by assuming that the probabilities of extinction and recolonisation are constant through time, in which case ψ^* can be substituted for ψ_{t+1} and ψ_t, giving:

$$\psi^* = \frac{\gamma}{\gamma + \varepsilon}. \qquad \text{(equation 2)}$$

It follows that if one knows how a species' extinction and recolonisation probabilities are affected by particular patch-level and landscape-level variables, it is possible to estimate relationships between these variables and the equilibrium probability of occupancy for that species (Ferraz et al. 2007). These relationships are very useful for developing guidelines for habitat management, because they allow the identification of the conditions required to attain a desired chance that a population will persist into the future. For example, from the relationships depicted in Figure 2 it is possible to estimate the equilibrium probability of occupancy by *L. raniformis* for various states of wetland hydroperiod, aquatic vegetation cover and connectivity. These relationships can in turn be used to identify the combination of conditions required to attain some target equilibrium probability of occupancy for the species, which equates to a target chance that a population of the frog will persist at any point in the future.

Secondly, the extensive monitoring dataset provides a basis for developing survey guidelines for *L. raniformis* in urbanising landscapes. Surveys aimed at establishing patterns of occurrence are fundamental to the regional management of the species for several reasons. Firstly, because *L. raniformis* is listed as threatened under both state and federal legislation, ascertaining whether remnant populations of the frog occur on land to be affected by urban development is a prerequisite of most developments in south-eastern Australia (DEWHA 2009). Secondly, because connectivity is a key variable controlling the extinction and recolonisation dynamics of *L. raniformis*, knowledge of the species' local distribution is a prerequisite for identifying appropriate habitat management strategies for the species (as above). Thirdly, monitoring local and regional changes in occurrence is vital for establishing the success or failure of habitat management, and, for this reason, is usually a prerequisite for the approval of urban developments that will affect the species (DEWHA 2009).

The data collated by Heard and Scroggie (2009) is useful for developing survey guidelines for *L. raniformis* because it allows the estimation of its probability of detection. The probability of detection is 'the chance that a species will be detected at a location at which it occurs during a single survey' (MacKenzie et al. 2006). This probability is rarely perfect, which means that multiple surveys are required to confidently determine whether a species occurs at a particular locality or not. For surveys aimed at establishing the distribution of wildlife, the key question is then 'how many surveys must be completed per locality to determine presence or absence with a desired level of confidence?'. As long as one has estimates of the probability of detection, and some idea about how confident one wants to be in the assessment of occupancy, this question can be answered using the following equation (Kéry 2002):

$$P = 1 - (1 - p)^n, \qquad \text{(equation 3)}$$

where P is the cumulative probability of detection, p is the probability of detection during a single survey, and n is the number of surveys conducted. The idea is to set a threshold for P and then calculate how many surveys are required to reach that threshold. For example, say that the probability of detection for a species during a single survey is 0.6, and that ones desires a cumulative probability of detection of 0.9. The number of surveys required to reach that threshold would be three, because P = 0.84 after two surveys and 0.93 after three surveys.

Of course, the detection probabilities of wildlife are rarely constant, being influenced by such things as time of year, weather conditions, skill of observers, etc. Analysis of the survey dataset collated by Heard and Scroggie (2009) revealed that the probability of detection of *L. raniformis* increases with increasing survey duration (measured in person-minutes), decreases with date (in days since October 1) [1,] and is much higher at night than during the day (Figure 3). These relationships can be easily accommodated when developing survey guidelines for wildlife using the approaches described above. The basic approach entails developing a series of plausible protocols with respect to the variables that influence the probability of detection, estimating the probability of detection under each protocol, and then estimating the number of surveys required in each case to attain the desired cumulative probability of detection. To illustrate with a simple example, consider a situation in which the probability of detection for a particular species only varies with survey duration, being 0.4 with a 30 minute survey, 0.6 with a 60 minute survey and 0.8 with a 90 minute survey. One could specify these different survey lengths as different survey protocols, and estimate the number of surveys required to attain a desired cumulative probability of detection when using each protocol. According to Equation 3, if P was set to 0.9 (as in the preceding example) the required number of surveys to reach this threshold would be seven, three and two, respectively.

[1] October 1 defines the start of the main active season for *L. raniformis* in southern Victoria (October–March).

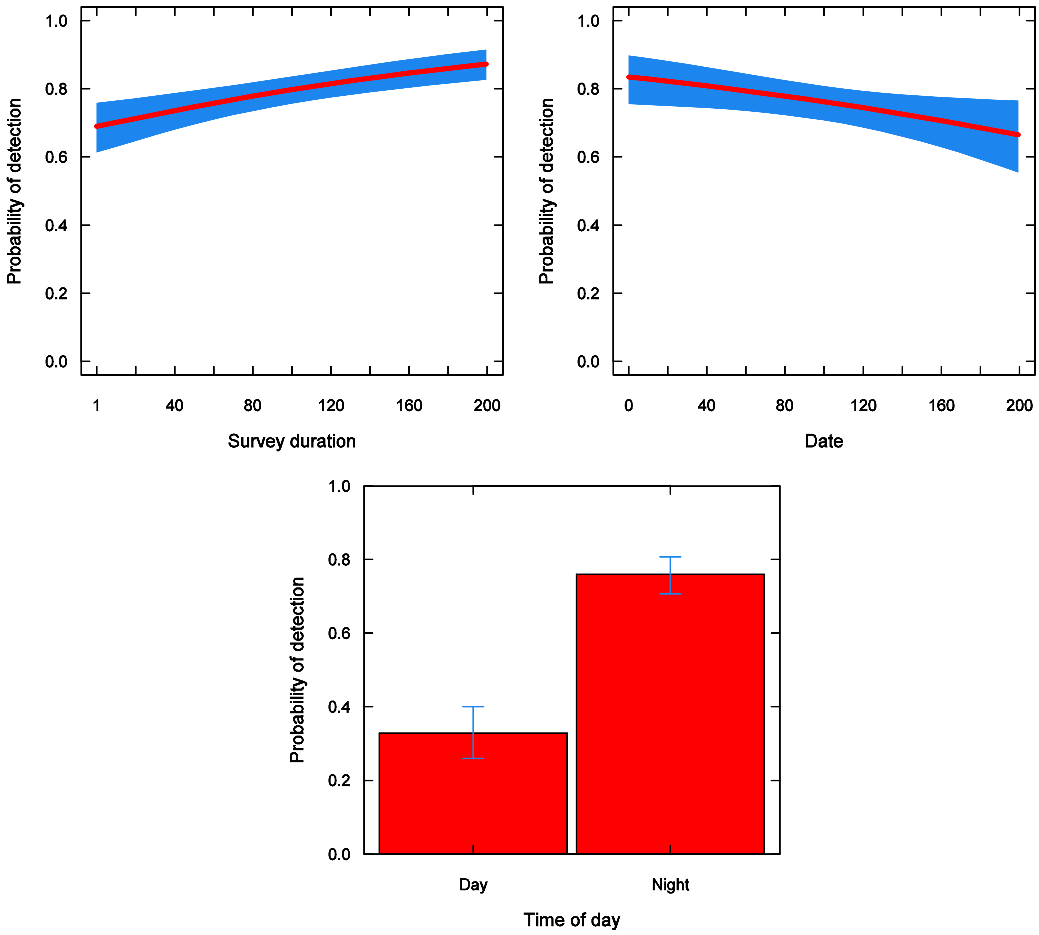

Figure 3. Relationships between the probability of detection of *Litoria raniformis* and survey duration (in person-minutes), date (in days since October 1) and time of day (day or night). In the two upper plots, the red line represents the estimated mean relationship, and the blue shading the 95% credible interval of the estimates. In the lower plot, bars represent the estimated mean, and error bars show the 95% credible intervals.

1.2 Purpose of the guidelines

The guidelines presented here are intended to facilitate management of *L. raniformis* in urbanising landscapes. They are aimed at public land managers and conservation agency staff, people and organisations involved in urban development (developers, consultants, etc.), and private land holders wishing to implement conservation initiatives for the species on their properties. The guidelines seek to promote a scientifically validated and consistent approach to the management of the species in these landscapes.

1.2.1 Objectives

The guidelines are primarily concerned with the description of protocols for habitat management and survey and monitoring, on the basis of the data and concepts introduced in the preceding section. Thus, the primary objectives of these guidelines are as follows:

- Present estimates of relationships between the equilibrium probability of occupancy for *L. raniformis* and wetland hydroperiod, aquatic vegetation cover and connectivity and, based upon varying thresholds of this probability, provide protocols for habitat management.
- Present estimates of the cumulative probability of detection for *L. raniformis* during each month of its active season and, based upon varying thresholds of this probability, provide protocols for survey and monitoring.

These two objectives cover the key issues relating to the conservation of *L. raniformis* in urbanising landscapes. Nevertheless, in both cases, some additional information may be useful or essential during the development of specific conservation plans for the species. For example, there is some evidence that the persistence of *L. raniformis* may be influenced by factors other than wetland hydroperiod, aquatic vegetation cover and connectivity (e.g., predatory fish, barriers to dispersal), and land managers may wish to incorporate measures to deal with those factors. Similarly, there may be circumstances in which it is desirable to measure or monitor variables other than occupancy. For example, managers may wish to know if particular habitat manipulations have altered reproductive success or population size. The secondary objectives of these guidelines are therefore as follows:

- Describe other wetland-level and landscape-scale variables that may influence the metapopulation dynamics of *L. raniformis* in urbanising landscapes, and outline additional priorities for habitat management.
- Describe alternative objectives for monitoring of *L. raniformis* in urbanising landscapes, and describe the effectiveness of techniques that may be applied to achieve those objectives.

Lastly, some recent plans for the conservation of *L. raniformis* in urbanising regions have involved experimental techniques to mitigate the effects of development, including the building of underpasses beneath roads, and translocating populations from wetlands earmarked for destruction. These techniques are considered experimental because there is not yet any evidence that they are effective. The final objective of these guidelines is therefore as follows:

- Review the appropriateness of several experimental management approaches that have recently been implemented during conservation initiatives for *L. raniformis* in urbanising landscapes.

1.2.2 Legislative and policy context

Litoria raniformis is listed as vulnerable to extinction under the Commonwealth *Environmental Protection and Biodiversity Conservation Act 1999*. It is also listed as threatened in each state in which it occurs. These various pieces of legislation require that approval be sought for any actions, including urban development, which may significantly impact upon them. These approvals typically require that proponents develop plans to mitigate such impacts.

The guidelines presented here are intended to facilitate the development of such plans, and to complement recently released 'significant impact guidelines' for *L. raniformis* produced by the Australian Government (DEWHA 2009). However, the guidelines presented here do not represent official policy of the Commonwealth government nor any State government with jurisdiction over the management of *L. raniformis*.

1.3 Legalities and limitations

1.3.1 Legalities

The implementation of these guidelines requires a consideration of the broader legalities of the actions they discuss. For example, most drainage lines (streams, creek, etc.) are protected by environmental overlays, so proposed changes to their hydrology or aquatic vegetation composition to enhance their quality for *L. raniformis* will require approval by the relevant authority. Similarly, with regard to surveys and monitoring, permits for these activities will need to be sought from the relevant State environment department. It is the responsibility of those implementing these guidelines to obtain the relevant approvals.

1.3.2 Limitations

The guidelines have several limitations, but two require particular consideration. Readers should be conscious of these limitations when applying these guidelines to inform conservation planning.

Potentially limited geographic applicability

The two primary objectives of these guidelines are to describe the habitat requirements of *L. raniformis*, and the means by which survey and monitoring of the species should be conducted. In both cases, the recommendations offered are based upon research conducted entirely within greater Melbourne (Heard and Scroggie 2009; see above). While the survey dataset collated during that project is extensive, its relevance for *L. raniformis* in other regions is unknown. This is particularly true for populations north and west of the Great Divide. Research on these populations indicates that their habitat requirements may differ from those of southern populations (Wassens 2005)[2]. Furthermore, the different climatic conditions they experience are likely to alter their seasonal patterns of activity and detectability. Caution is therefore recommended in applying the guidelines in regions far from Melbourne.

Uncertainty in parameter estimates

As described above, the primary objectives of these guidelines are to (i) present estimated relationships between the equilibrium probability of occupancy by *L. raniformis* and wetland hydroperiod, aquatic vegetation and connectivity, and to define protocols for management of the species' habitat on that basis, and (ii) to present month-by-month estimates of the cumulative probability of detection for *L. raniformis,* and to define protocols for the conduct of occupancy-based surveys for the species on that basis.

We have excluded uncertainty in these estimates from the guidelines for the sake of clarity. This is of little concern with regard to estimates of the cumulative probability of detection, because the uncertainty in the relationships that underlie them is relatively small (Figure 3), and the estimates subsequently displayed minimal error. However, given the substantial level of uncertainty in the estimated relationship between connectivity and the probability of recolonisation for *L. raniformis* (Figure 2), estimates of the equilibrium probability of occupancy for the species have a considerable uncertainty. As a result, the habitat management guidelines may fall short of achieving the desired chance in population persistence into the future.

[2] There is also some evidence that northern populations of *L. raniformis* may be genetically distinct from those south of the Great Divide (Vörös et al. 2008).

2 Habitat management

2.1 Equilibrium probability of occupancy

2.1.1 Recapping the principle

The equilibrium probability of occupancy (ψ^*) is a statistic that can be interpreted as the 'the chance that a population will persist at a given locality at any point in the future'. Thus, if the equilibrium probability of occupancy for a particular species at a particular locality is 0.8, there is an 80% chance that a population of that species will persist at that locality at any point in the future.

The equilibrium probability of occupancy is a useful statistic for developing guidelines for managing the habitat of a particular species. The basic idea is to set a threshold for this probability (i.e. a desired chance of persistence at any given locality) and, based upon its relationships with particular patch-level and landscape-level variables, to identify the set of conditions required to achieve that threshold. For example, if an equilibrium probability of occupancy of at least 0.9 for all remnant populations of a particular species is desirable, we might estimate that the equilibrium probability of occupancy reaches that threshold only if tree cover within a habitat patch is above 50%. A protocol for managing that species' habitat would then be to maintain tree cover above 50%.

2.1.2 Relationships with wetland hydroperiod, aquatic vegetation cover and connectivity

Estimating relationships between the equilibrium probability of occupancy and particular wetland-level and landscape-level variables first requires an understanding of relationships between those variables and the probabilities of extinction and recolonisation. This understanding is available for *L. raniformis* as a result of long-term monitoring across Melbourne's northern fringe (Heard and Scroggie 2009; see section 1.1.4). In that region, the probability of extinction was negatively related to wetland hydroperiod, aquatic vegetation cover and connectivity, and the probability of recolonisation was strongly positively related to connectivity (Figure 2).

The resulting relationships between these variables and the equilibrium probability of occupancy for *L. raniformis* are shown in Figure 4. For each of the four hydroperiod categories, the equilibrium probability of occupancy increases with aquatic vegetation cover and connectivity, given the negative relationships between these two variables and the probability of extinction, and the strong positive relationship between connectivity and the probability of recolonisation. Similarly, the very strong negative relationship between the probability of extinction and wetland hydroperiod is reflected in the fact that the equilibrium probability of occupancy for *L. raniformis* is fairly low over most values of aquatic vegetation cover and connectivity when the hydroperiod is short, but rises quickly and plateaus when the hydroperiod is long.

Thus, populations of *L. raniformis* inhabiting wetlands with short hydroperiods have a poor chance of persistence, even when those wetlands are of otherwise good quality and the neighbouring population density is high. Conversely, populations inhabiting semi-permanent wetlands have a moderate to high chance of persistence over roughly half the combinations of aquatic vegetation cover and connectivity, while populations occupying permanent wetlands with high aquatic vegetation cover and connectivity are effectively perennial.

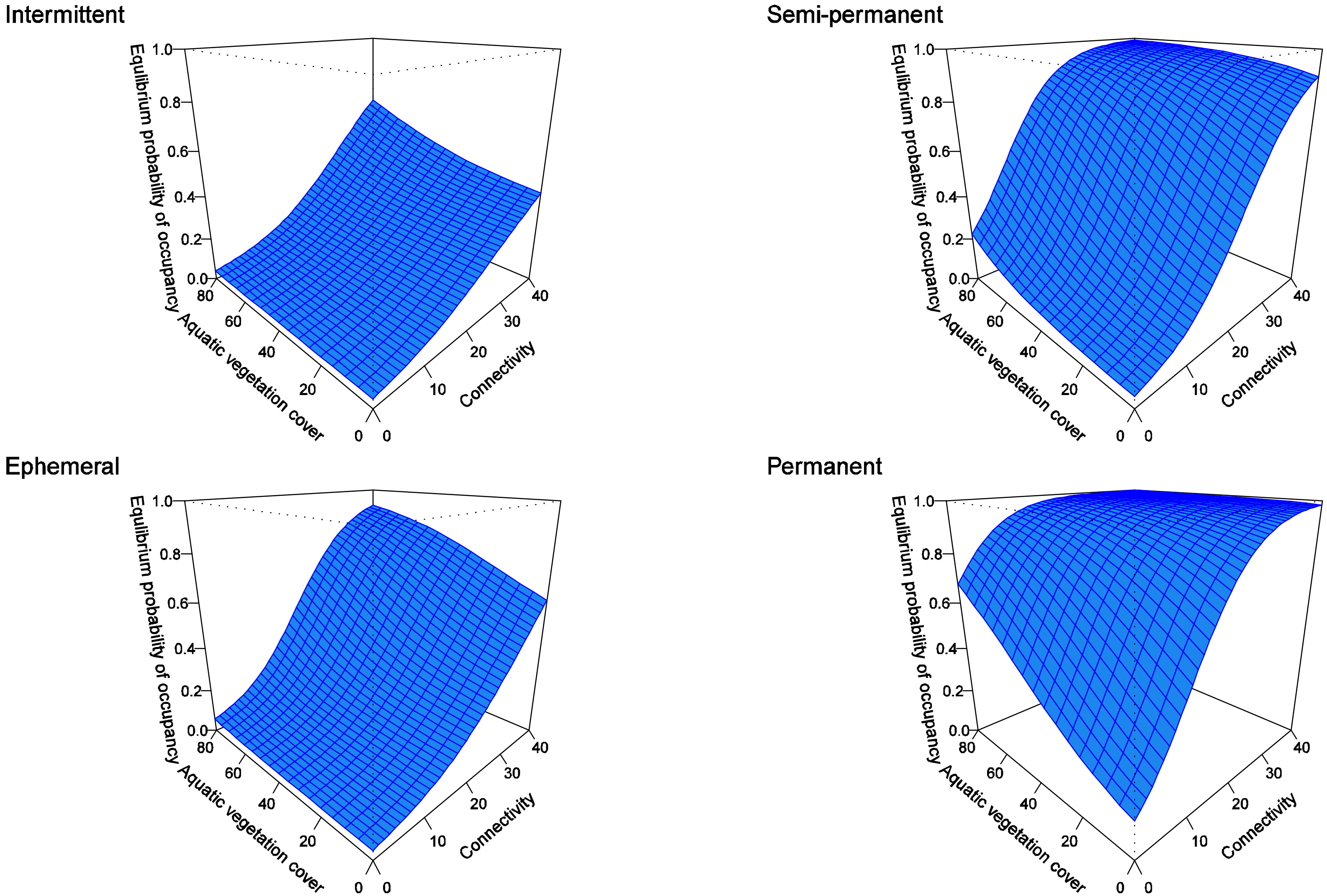

Figure 4. Relationships between the equilibrium probability of occupancy by *Litoria raniformis* and wetland hydroperiod, aquatic vegetation cover and connectivity. Relationships are categorised according to wetland hydroperiod. See Table 1 for definitions of the variables. The upper limits for aquatic vegetation cover (80 %) and connectivity (40) are based upon those observed by Heard and Scroggie (2009).

2.1.3 Thresholds

Because it is impossible to be certain that any population of wildlife will persist into the future, the key question in setting a threshold equilibrium probability of occupancy for *L. raniformis* is 'What chance of persistence is acceptable?' Questions such as this are essentially exercises in risk assessment and have no predefined answers. Instead, one must find some balance between the importance of risk minimisation and its practical, social or financial constraints (Burgman et al. 1993).

To provide managers with flexibility in this regard, the following guidelines encompass four thresholds of the equilibrium probability of occupancy: 0.8, 0.9, 0.95 and 0.99. Each of these thresholds is a credible balance between the objective of maintaining a very high chance that remnant populations of *L. raniformis* will persist into the future (given the threatened status of the species) and the practicalities of habitat management in urbanising landscapes.

Relationships between these thresholds and wetland hydroperiod, aquatic vegetation cover and connectivity are depicted in Figure 5. These relationships are a binary representation of those depicted in Figure 4; that is, they show the set of conditions in which the equilibrium probability of occupancy is estimated to be either at or above each threshold, or below it. From these relationships it is clear that wetlands that fill only intermittently confer an inadequate chance of persistence, regardless of their cover of aquatic vegetation or connectivity. Similarly, while the lower thresholds of the equilibrium probability of occupancy are attainable at ephemeral wetlands, this is only the case when both aquatic vegetation and connectivity are maximal. Semi-permanent wetlands confer an equilibrium probability of occupancy of 0.8 or more over roughly the upper third of combinations of aquatic vegetation cover and connectivity, reaching 0.95 if there is dense aquatic vegetation and numerous neighbouring populations of *L. raniformis*. In contrast, permanent wetlands confer this probability over almost a half of all combinations of aquatic vegetation cover and connectivity, and confer a probability of 0.99 over a reasonably wide set of conditions.

2.2 Guidelines for habitat management

2.2.1 General context and approach

Our interpretation of the equilibrium probability of occupancy in terms of population persistence means that the following guidelines have an explicit population-level focus. However, it should be clear from the preceding sections that these guidelines implicitly entail a metapopulation-level approach to management. The equilibrium probability of occupancy is a function of a species' extinction and recolonisation probabilities, which metapopulation theory tells us are (at least partly) a function of dispersal rates and hence connectivity. The very strong relationship between the equilibrium probability of occupancy and connectivity for *L. raniformis* is a reflection of the fact that dispersal rates exert a strong influence over both extinction and recolonisation probabilities for this species, and therefore that dispersal between populations is vital for their long-term persistence.

The result is that it is of little consequence whether one is interested in the preservation of just a single population or an entire metapopulation, because the requirement in both cases is to maintain a high chance of dispersal and population interaction by maintaining high connectivity.

Within this context, these guidelines will be applicable to a range of approaches to habitat management for *L. raniformis*, including habitat protection, habitat enhancement and habitat creation. For example, in the event that it is inevitable that populations will be lost during urban development, the guidelines could be used to ascertain the arrangement of populations (and hence, wetlands) that must be preserved to ensure that each retains the desired equilibrium probability of occupancy. In the same situation, the guidelines could be used to identify how many new populations must be created to offset the loss of existing ones, either by constructing new wetlands or by enhancing existing poor-quality wetlands. Similarly, if only one or two remnant populations of the frog persist at a locality, the guidelines could be used to plan enhancements to the occupied wetlands, and to ascertain the number and distribution of new wetlands that would be required to re-establish appropriate levels of connectivity.

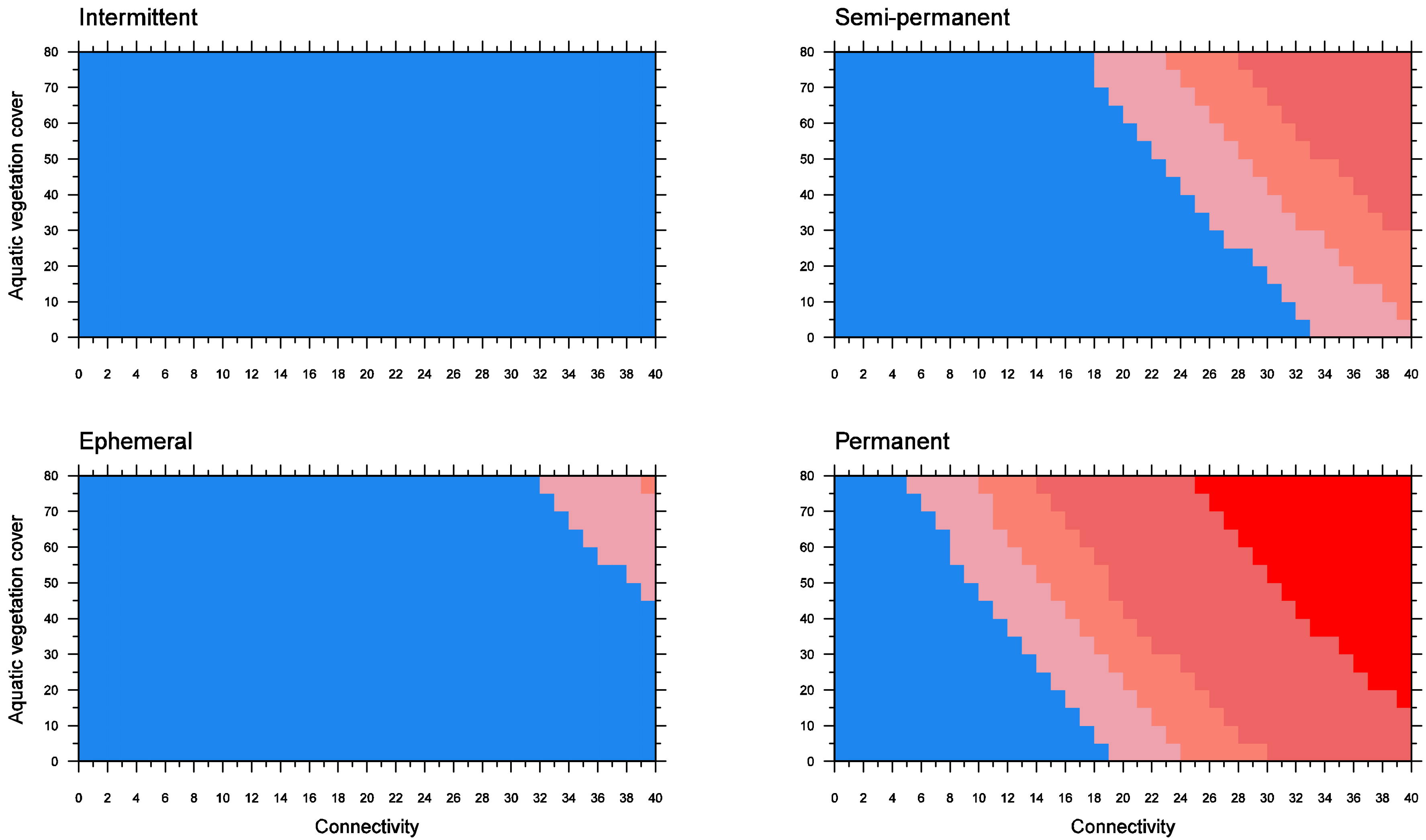

Figure 5. Relationships between the four thresholds of the equilibrium probability of occupancy by *Litoria raniformis* and wetland hydroperiod, aquatic vegetation cover and connectivity. In each plot, the blue shaded area represents the conditions in which the equilibrium probability of occupancy is < 0.8, and the red shaded areas the conditions in which the equilibrium probability of occupancy is ≥ 0.8, ≥ 0.9, ≥ 0.95 or ≥ 0.99 (from lightest to darkest). The upper limits for aquatic vegetation cover (80%) and connectivity (40) are based upon those observed by Heard and Scroggie (2009).

2.2.2 Wetland-level requirements

Wetland hydroperiod

An important goal of habitat management for *L. raniformis* in urbanising landscapes will be the maintenance and enhancement of wetland hydroperiods. This is clear with regard to wetlands that only fill intermittently, as the maximum equilibrium probability of occupancy under this regime is lower than the minimum threshold of this probability considered here. Furthermore, whilst ephemeral and semi-permanent wetlands can display a high equilibrium probability of occupancy (≥ 0.9), they do so only in the exceptional scenario in which aquatic vegetation cover and connectivity are at their maximum. Thus, in many cases, bolstering the hydroperiod of wetlands will be necessary to achieve the target equilibrium probability of occupancy.

The hydroperiod of a wetland is a function of various environmental characteristics, but is ultimately determined by the rate of inflow and outflow of water. Any manipulation of a wetland's hydroperiod is therefore ultimately concerned with increasing the former and decreasing the latter. Potential means of achieving this include the following:

- Increasing inflows:
    - diverting or creating drainage lines to capture overland flows
    - extracting water from nearby creeks or rivers, either by diversion structures or pumping
    - tapping underground water supplies
    - capturing stormwater from existing or future urban developments.
- Decreasing outflows:
    - enhancing wetland area and depth, and therefore storage capacity
    - installing weirs along streams
    - installing levees or increasing their height
    - capping or filling in existing drainage infrastructure.

Regardless of which of these options is pursued, it is vital to consider the possible impact of changes in hydrology on aquatic vegetation and other variables that may influence habitat quality for *L. raniformis* (e.g., water quality, predatory fish densities: see below). It is also vital to consider the impacts of these works on other nearby wetlands. It is obviously counter-productive to bolster the hydroperiod for one population of *L. raniformis* if this results in a decrease in the hydroperiod of other wetlands inhabited by the frog.

Aquatic vegetation cover

Table 2 provides estimates of the cover of aquatic vegetation needed to attain each of the four thresholds of the equilibrium probability of occupancy by *L. raniformis* considered here. These estimates are taken directly from Figure 5. They can be used to identify the cover of aquatic vegetation needed to attain each threshold, given a particular combination of hydroperiod and connectivity.

There is, however, one important caveat with regard to the values provided in Table 2. While moderate to high aquatic vegetation cover is required under most circumstances, there are some cases in which the cover can be very low (semi-permanent or permanent wetlands with very high connectivity). The implication is that under these circumstances resident populations of *L. raniformis* will be sustained largely by immigration from neighbouring populations, because aquatic vegetation cover is thought to be a fundamental determinant of reproductive success for *L. raniformis*. As a result, managers should employ some discretion and, wherever possible, pursue at least a moderate cover of aquatic vegetation in semi-permanent or permanent wetlands with high connectivity.

Another important point is that aquatic vegetation cover, as defined in Table 1, is <u>the mean cover of emergent, submergent and floating vegetation</u>. Thus, for aquatic vegetation cover to be high there must be a high cover of <u>each</u> of these vegetation types, not one on its own. Figure 6 shows an example of wetlands with increasingly favourable aquatic vegetation cover.

Table 2. Aquatic vegetation cover (%) required to attain a threshold equilibrium probability of occupancy by *Litoria raniformis* of 0.8, 0.9, 0.95 or 0.99. Values are provided for differing combinations of wetland hydroperiod and connectivity. A dash (–) represents combinations in which the equilibrium probability of occupancy is estimated to be less than 0.8. The upper limits for aquatic vegetation cover (80%) and connectivity (40) are based upon those observed by Heard and Scroggie (2009).

Connectivity	Ephemeral				Semi–permanent				Permanent			
	0.8	0.9	0.95	0.99	0.8	0.9	0.95	0.99	0.8	0.9	0.95	0.99
0	–	–	–	–	–	–	–	–	–	–	–	–
1	–	–	–	–	–	–	–	–	–	–	–	–
2	–	–	–	–	–	–	–	–	–	–	–	–
3	–	–	–	–	–	–	–	–	–	–	–	–
4	–	–	–	–	–	–	–	–	80	–	–	–
5	–	–	–	–	–	–	–	–	75+	–	–	–
6	–	–	–	–	–	–	–	–	70+	–	–	–
7	–	–	–	–	–	–	–	–	65+	–	–	–
8	–	–	–	–	–	–	–	–	55+	–	–	–
9	–	–	–	–	–	–	–	–	50+	80	–	–
10	–	–	–	–	–	–	–	–	45+	75+	–	–
11	–	–	–	–	–	–	–	–	40+	65+	–	–
12	–	–	–	–	–	–	–	–	35+	60+	–	–
13	–	–	–	–	–	–	–	–	30+	55+	–	–
14	–	–	–	–	–	–	–	–	25+	50+	75+	–
15	–	–	–	–	–	–	–	–	20+	45+	70+	–
16	–	–	–	–	–	–	–	–	15+	40+	65+	–
17	–	–	–	–	80	–	–	–	10+	35+	60+	–
18	–	–	–	–	70+	–	–	–	5+	30+	55+	–
19	–	–	–	–	65+	–	–	–	0+	25+	45+	–
20	–	–	–	–	60+	–	–	–	0+	20+	40+	–
21	–	–	–	–	55+	–	–	–	0+	15+	35+	–
22	–	–	–	–	50+	80	–	–	0+	10+	30+	–
23	–	–	–	–	45+	75+	–	–	0+	5+	30+	–
24	–	–	–	–	40+	70+	–	–	0+	0+	25+	80
25	–	–	–	–	35+	65+	–	–	0+	0+	20+	75+
26	–	–	–	–	30+	60+	–	–	0+	0+	15+	70+
27	–	–	–	–	25+	55+	80	–	0+	0+	10+	65+
28	–	–	–	–	25+	50+	75+	–	0+	0+	5+	60+
29	–	–	–	–	20+	45+	70+	–	0+	0+	5+	55+
30	–	–	–	–	15+	40+	65+	–	0+	0+	0+	50+
31	80	–	–	–	10+	35+	60+	–	0+	0+	0+	45+
32	75+	–	–	–	5+	30+	55+	–	0+	0+	0+	40+
33	70+	–	–	–	0+	30+	50+	–	0+	0+	0+	35+
34	65+	–	–	–	0+	25+	50+	–	0+	0+	0+	35+
35	60+	–	–	–	0+	20+	45+	–	0+	0+	0+	30+
36	55+	–	–	–	0+	15+	40+	–	0+	0+	0+	25+
37	55+	–	–	–	0+	15+	35+	–	0+	0+	0+	20+
38	50+	80+	–	–	0+	10+	30+	–	0+	0+	0+	20+
39	45+	75+	–	–	0+	5+	30+	–	0+	0+	0+	15+
40	40+	70+	–	–	0+	5+	25+	–	0+	0+	0+	10+

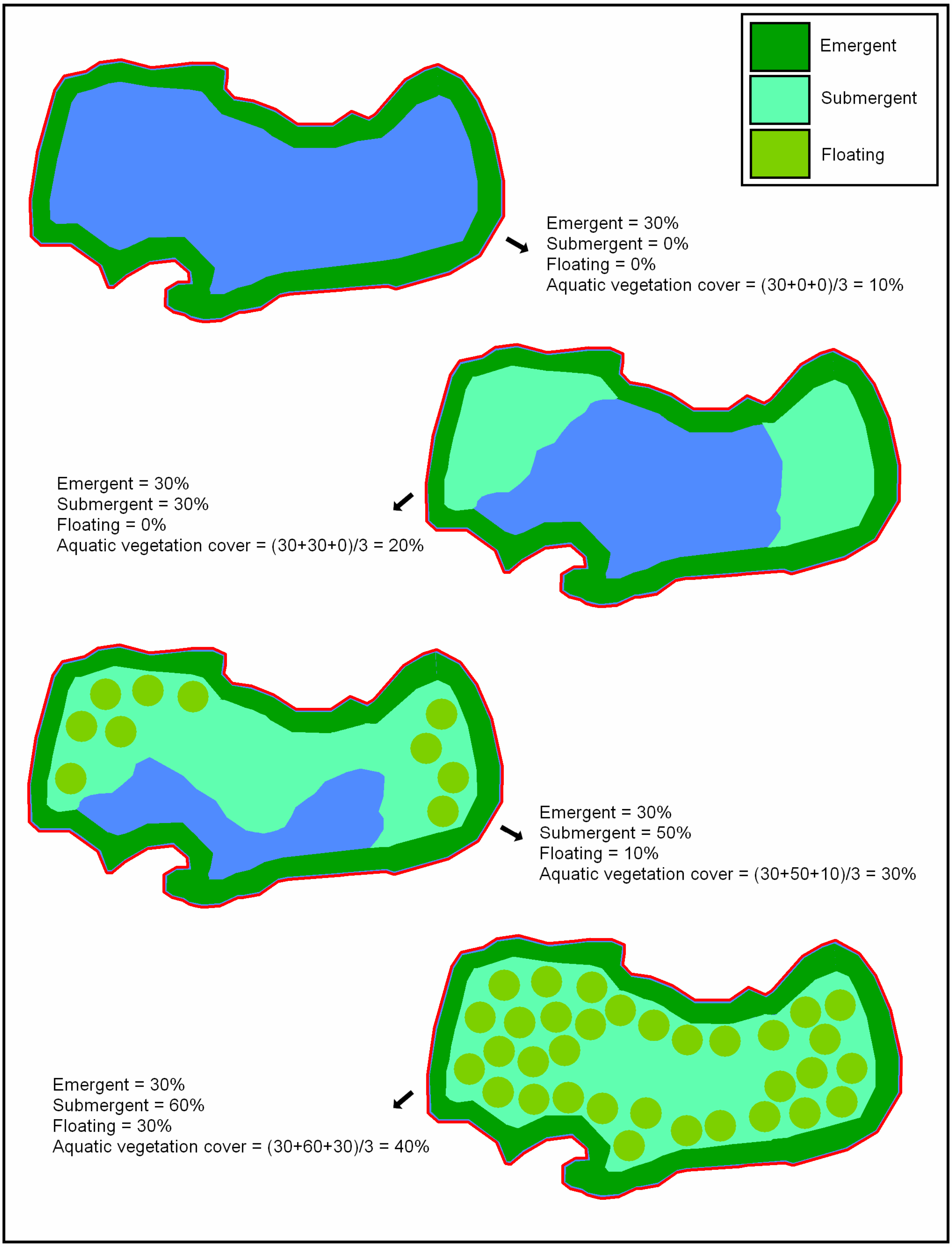

Figure 6. Diagram of wetlands with increasingly favourable aquatic vegetation cover for *Litoria raniformis*. Aquatic vegetation cover is the mean cover of emergent, submergent and floating vegetation, where each may range between 0–100% (see Table 1). Table 1 also provides definitions of each vegetation type. The Appendix provides a list of plant species that are characteristic of each vegetation type.

Within each of the three major types of aquatic vegetation, a wide variety of taxa appears suitable for *L. raniformis.* The Appendix provides a list of aquatic plants that are commonly observed within wetlands inhabited by the species on the northern fringe of Melbourne, and that have been observed to be utilised by the frog during diurnal and nocturnal activity (Heard et al. 2008a).

Because there are numerous constraints on the growth of aquatic vegetation, an aquatic botanist may need to be consulted on how to maintain or enhance aquatic vegetation cover. However, there are some generally effective and easily implemented means of facilitating aquatic vegetation growth in existing or newly created wetlands, including:[3]

- manipulating wetland depth to ensure areas > 1.5 m are permanently or consistently available (which constrains emergents, and facilitates proliferation of submergent and floating varieties)
- exclusion fencing to protect existing aquatic vegetation from grazing by livestock (but note the potential benefits of light grazing discussed below)
- exclusion netting to protect existing aquatic vegetation from grazing by waterfowl
- removing exotic riparian vegetation (Willow [*Salix* spp.], Poplar [*Populus* spp.], etc.) to increase solar radiation
- controlling invasive emergents such as Common Reed (*Phragmites australis*) and Spiny Rush (*Juncus acuta*)
- removing exotic fish, particularly Common Carp (*Cyprinus carpio*) and Goldfish (*Carassius auratus*), whose feeding strategy uproots aquatic vegetation (see also below)
- directly planting the desired taxa (see Appendix).

2.2.3 Landscape-level requirements

Table 3 provides estimates of the connectivity needed to attain the four thresholds of the equilibrium probability of occupancy by *L. raniformis*. These values are taken directly from Figure 5. They can be used to identify the density and proximity of neighbouring populations required to attain each threshold, taking into account variations in the hydroperiod and aquatic vegetation cover of the wetland. To help interpret these statistics, Figure 7 depicts neighbouring population arrangements that confer increasing levels of connectivity.

There are three ways to achieve appropriate levels of connectivity for individual populations of *L. raniformis* in urbanising landscapes: habitat protection, habitat enhancement and habitat creation. In most situations all three will be required to achieve an appropriate level of connectivity:

- *Habitat protection* is the preservation of existing populations of *L. raniformis* within a 1 km radius[4] of the focal population/s, by preserving and maintaining the wetlands in which they occur.
- *Habitat enhancement* is the improvement of existing wetlands close to the focal population/s so that they can be colonised by *L. raniformis* and support additional neighbouring populations. An example is the enhancement of farm dams, which are numerous across the range of the species but are frequently unoccupied because of short hydroperiods or poor aquatic vegetation cover. However, habitat enhancement need not be restricted to the improvement of artificial wetlands such as farm dams; it includes enhancing pools along streams, creeks and other drainage lines.
- *Habitat creation* is the construction of purpose-built wetlands for *L. raniformis* near the focal population/s, so that they be colonised and support additional populations. This option primarily entails the construction of wetlands to be filled by surface run-off, drainage diversions or pumping. Although the construction of pools along ephemeral streams or drainage lines may be considered habitat creation, it is really a form of habitat enhancement because it essentially involves enhancing the hydroperiod of an existing wetland.

[3] See also Romanowski (1998).
[4] 1 km is the spatial scale over which connectivity is calculated: see Table 1.

The specifics of wetland enhancement and creation for *L. raniformis* are likely to vary substantially from case to case because there will be different constraints and opportunities. Rather than attempting to cover these issues in detail here, managers are advised to consult the various texts now available on the subject of wetland management and construction when planning such activities (e.g., Romanowski 1998; Kent 2000; Mitsch and Gosselink 2000; Zentner 2000). Nevertheless, it is obviously vital that these wetlands display the same characteristics as those being pursued for the focal populations (as above), otherwise the new populations which they have been designed to support may be short-lived.

Table 3. Connectivity required to attain a threshold equilibrium probability of occupancy by *Litoria raniformis* of 0.8, 0.9, 0.95 or 0.99. Values are provided for differing combinations of wetland hydroperiod and aquatic vegetation cover. A dash (–) represents combinations in which the equilibrium probability of occupancy is estimated to be less than 0.8. The upper limits for aquatic vegetation cover (80%) and connectivity (40) are based upon those observed by Heard and Scroggie (2009).

Aquatic vegetation cover (%)	Ephemeral				Semi-permanent				Permanent			
	0.8	0.9	0.95	0.99	0.8	0.9	0.95	0.99	0.8	0.9	0.95	0.99
0	–	–	–	–	33+	–	–	–	19+	24+	30+	–
5	–	–	–	–	32+	39+	–	–	18+	23+	28+	–
10	–	–	–	–	31+	38+	–	–	17+	22+	27+	40
15	–	–	–	–	30+	36+	–	–	16+	21+	26+	39+
20	–	–	–	–	29+	35+	–	–	15+	20+	25+	37+
25	–	–	–	–	27+	34+	40	–	14+	19+	24+	36+
30	–	–	–	–	26+	32+	38+	–	13+	18+	22+	35+
35	–	–	–	–	35+	31+	37+	–	12+	17+	21+	33+
40	40	–	–	–	24+	30+	36+	–	11+	16+	20+	32+
45	39+	–	–	–	23+	29+	35+	–	10+	15+	19+	31+
50	38+	–	–	–	22+	28+	33+	–	9+	14+	19+	30+
55	36+	–	–	–	21+	27+	32+	–	8+	13+	18+	29+
60	35+	–	–	–	20+	26+	31+	–	8+	12+	17+	28+
65	34+	–	–	–	19+	25+	30+	–	7+	11+	16+	27+
70	33+	40	–	–	18+	24+	29+	–	6+	11+	15+	26+
75	32+	39+	–	–	18+	23+	28+	–	5+	10+	14+	25+
80	31+	38+	–	–	17+	22+	27+	–	4+	9+	14+	24+

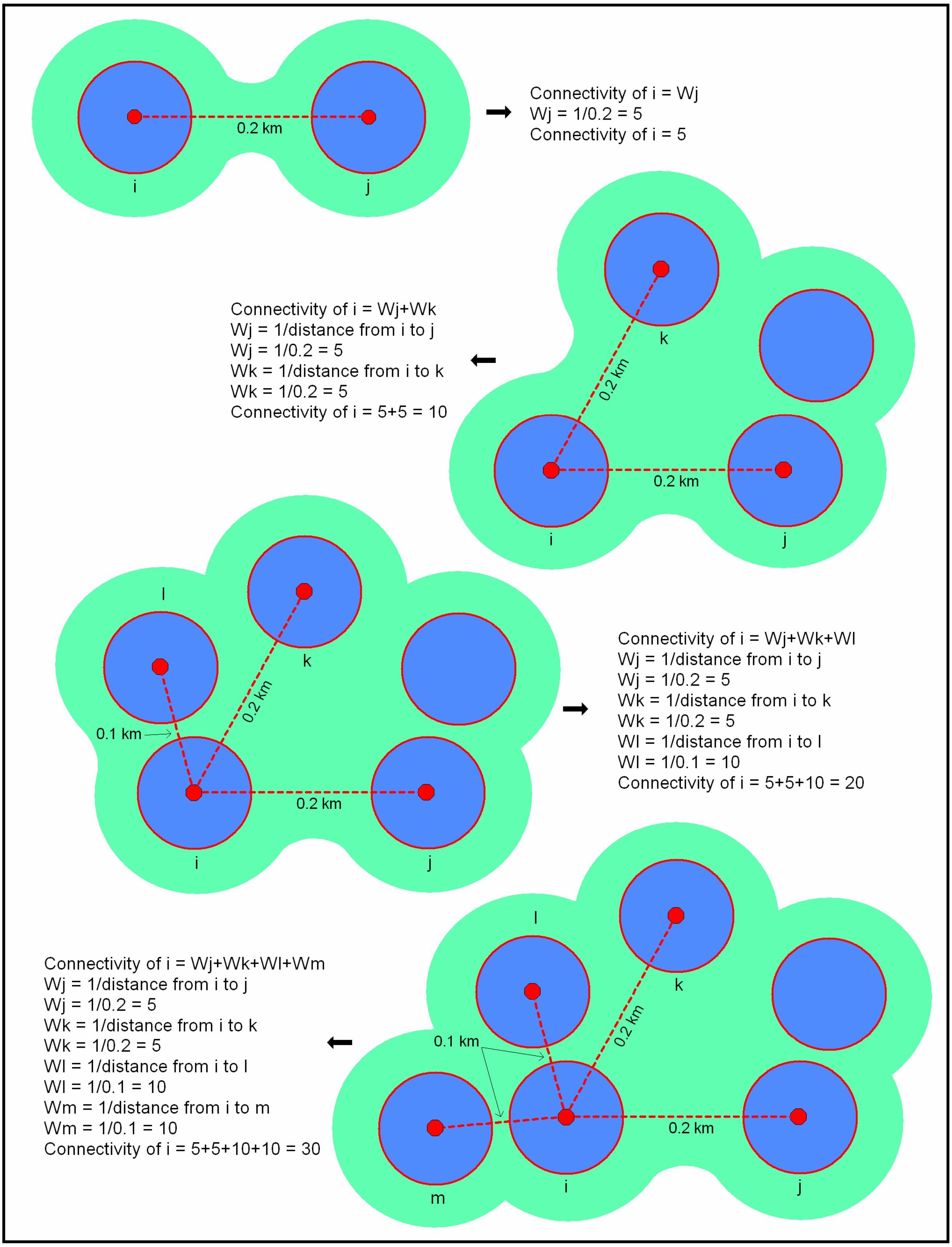

Figure 7. Diagram of neighbouring population arrangements of *Litoria raniformis* that confer increasing levels of connectivity. Blue circles represent wetlands, and those supporting a population of *L. raniformis* are signified by a red dot. The dashed lines represent the distances between the focal population in each scenario (population *i*), and each of its neighbouring populations (*j–m*). Inter-population distances are measured as the distance (km) between wetland centres. The calculation of connectivity for population *i* is provided for each arrangement, based upon the formula in Table 1.

2.3 Other considerations

2.3.1 Wetland-level

Wetland size

Because Heard and Scroggie (2009) found that larger wetlands are more likely to be occupied by *L. raniformis*, they examined the possible influence of wetland area on extinction probability for the species. They did not find a strong relationship between wetland area and extinction probability, and concluded that wetland hydroperiod and aquatic vegetation cover were more important wetland-level determinants of extinction probability for the species (section 2.2.3).

Nevertheless, there are good reasons to believe that larger wetlands may support larger, less extinction-prone populations of *L. raniformis*. The first is the correlation between the area and hydroperiod of off-stream wetlands. Larger, deeper wetlands are less likely to dry out from year to year, which means that resident populations of *L. raniformis* have an inherently lower chance of extinction. Another reason is that large, deep wetlands have a greater propensity to support the diverse aquatic vegetation that *L. raniformis* requires, because small, shallow wetlands have a tendency to be 'choked' by emergent vegetation. A third reason is that *L. raniformis* is a highly cannibalistic species (Pyke 2002), and males tend to be territorial during the reproductive season (G. Heard unpubl. data). As such, population growth rates may be limited by density.

The implication is that, in the case of habitat enhancement or creation, it would be prudent to maximise wetland size. In the study by Heard and Scroggie (2009), wetlands occupied by *L. raniformis* displayed a mean water surface area of 3837 square metres. It is recommended that habitat enhancement and creation initiatives seek to at least replicate this surface area.

Terrestrial buffer zones

It has been increasingly recognised in recent years that the terrestrial surrounds of wetlands are a crucial resource for many amphibians (e.g., Semlitsch and Bodie 2003). *Litoria raniformis* is a highly aquatic species but is active on land during the warmer months of the year, apparently foraging for prey (Heard et al. 2008a). The terrestrial zone of wetlands also provides cover such as rocks, logs and soil cracks, which are important shelter and overwintering sites (see below).

Although Heard and Scroggie (2009) did not find a positive relationship between extinction probability for *L. raniformis* and the extent of buffer zone degradation (measured as the cover of roads and buildings within a 150 m radius of a wetland), they did find a strong negative relationship between this variable and the probability of wetland occupancy by the species. They attributed the lack of a positive relationship between extinction probability and buffer zone degradation in their study to sample size constraints, rather than the lack any relationship *per se*.

The maintenance of significant terrestrial buffer zones around wetlands is an important aspect of habitat management for *L. raniformi*s in urbanising landscapes. With regard to the scale of terrestrial buffers, these guidelines defer to the recently released 'Significant Impact Guidelines' for *L. raniformis* (DEWHA 2009), in which buffer zones of at least 200 m from the water's edge are recommended.

Microhabitat requirements

Three recent studies have considered the microhabitat requirements of *L. raniformis*. Using radio-telemetry, Wilson (2003) and Wassens et al. (2008) gathered data on the types of shelter sites utilised by the species during the overwintering and active seasons, respectively. Overwintering sites recorded by Wilson (2003) included crevices beneath basalt boulders, crevices amongst rock-rubble, and dense vegetation at ground level, including emergent macrophytes and grasses. All were located close to the waterline. Wassens et al. (2008) observed a similar preference for shelter sites close to the water's edge. Most individuals sheltered among dense vegetation, but a few sheltered in soil cracks and the burrows of freshwater crayfish. Heard et al. (2008a) were interested in microhabitat preferences during nocturnal activity. They found aquatic vegetation to be overwhelmingly preferred when individuals were active aquatically, and patches of bare ground and basalt rocks to be preferred during terrestrial foraging.

Thus, it is apparent that a diversity of terrestrial structures are required by *L. raniformis*. Habitat management initiatives should strive to provide boulders and rock piles close to the water's edge, interspersed among a mosaic of vegetation and bare ground. This point is relevant to the question of controlling livestock access to wetlands inhabited by this species. High-intensity grazing tends to denude wetlands of the aquatic vegetation that *L. raniformis* require. Nevertheless, low-intensity grazing or pulse grazing may be beneficial, because it suppresses growth of grasses and woody weeds and helps maintain a mosaic of terrestrial vegetation and bare ground in the riparian zone.

Predatory fish

Predation by exotic fish has been implicated in the decline of numerous aquatic-breeding amphibians, including *L. raniformis* and its sister species *L. aurea* (Mahony 1999). While experimental work confirms that the larvae of *L. raniformis* are susceptible to predation by both exotic and native fish (Howard 2004), field studies have failed to demonstrate any exclusion of the species from wetlands inhabited by these predators (Robertson et al. 2002; Heard et al. 2004; Poole 2004; Hamer and Organ 2008). This conundrum was considered by Heard and Scroggie (2009), who also found that the presence of predatory fish had little effect on the probability of wetland occupancy by *L. raniformis*. Their interpretation was that although predatory fish are likely to suppress recruitment by this species, predation rates may be offset by high aquatic vegetation cover because of the protective cover it affords eggs and larvae. If this interpretation is correct, fish predation represents an additional mechanism underlying the relationship between extinction probability and aquatic vegetation cover for this species.

Recommendations for the management of fish predation are as follows:

- Exclude or eradicate predatory fish (particularly exotic species) where practicable.
- Enhance aquatic vegetation cover if the exclusion or eradication of predatory fish is problematic.

Water quality and flow rate

The influence of water quality and flow rate on habitat quality for *L. raniformis* has received only limited attention. Poole (2004) observed a negative relationship between flow rate and wetland occupancy by *L. raniformis*, which accords with the view of Heard and Scroggie (2009), who considered slow-flowing pools along streams to be breeding habitat for this species, and intervening fast-flowing sections to be occupied primarily during dispersal.

Wassens (2005), Hamer and Organ (2008), Smith et al. (2008) and Heard and Scroggie (2009) considered the relationship between wetland occupancy by *L. raniformis* and several basic water quality parameters. The results are conflicting. Wassens (2005) found that occupied wetlands had a relatively low pH, whereas Hamer and Organ (2008) reported the opposite relationship. Wassens (2005) and Smith et al. (2008) found a negative relationship between the probability of wetland occupancy and electrical conductivity (a surrogate measure of salinity), but Heard and Scroggie (2009) found no such relationship.

Recommendations with regard to flow rate are subsequently straightforward: still or very slowly flowing water is clearly required for reproduction. Efforts to control basic water chemistry appear unnecessary, apart from ensuring that electrical conductivity does not increase beyond about 10000 μS/cm (the approximate limit for the species reported by Smith et al. 2008).

An important footnote to this discussion is the fact that none of the above studies considered gross pollutants which commonly occur in urban storm water (e.g., heavy metals, surfactants, detergents). Nevertheless, the generally adverse effect of these substances on amphibians (Hamer and McDonnell 2008) suggested they will curtail habitat quality for *L. raniformis*. Urban storm water should consequently receive appropriate mechanical or biological treatment (or both) if it is to be diverted into wetlands occupied by *L. raniformis*, or if it is used as a water source to enhance or create habitat for the frog.

2.3.2 Landscape-level

Barriers to dispersal

Heard and Scroggie (2009) considered the possibility that urban infrastructure presents barriers to dispersal for *L. raniformis*. They hypothesised that wetlands with a high cover of roads and buildings in the surrounding landscape would have a lower probability of occupancy, because they would have a higher chance of extinction and a lower chance of recolonisation. Although they did observe a negative relationship between the cover of urban infrastructure in the surrounding landscape and wetland occupancy by *L. raniformis*, the relationship was weak.

Nevertheless, this finding should not be seen as evidence that these structures do not impede dispersal by the species, for two reasons:

- The occupancy data used by Heard and Scroggie (2009) had limited power to detect such relationships, because most of the wetlands surveyed were along stream corridors, and even if they did have a high urban cover in the surrounding landscape they were usually linked to other wetlands by open space.
- During a mark–recapture study also undertaken by Heard and Scroggie (2009), dispersal was observed only between wetlands connected by open space or drainage lines. Recent genetic analysis of tissue samples collected during that study confirms that adjacent populations connected by open space or drainage lines exchange migrants, whereas adjacent populations separated by urban infrastructure do not.

Thus, in addition to the distribution of occupied wetlands (as defined by connectivity), barriers to dispersal are also an important landscape-level consideration for habitat management. It is imperative that open space is maintained between wetlands to facilitate dispersal. Some infrastructure, including bike paths, minor roads and scattered buildings, are unlikely to be problematic in this regard, but the placement of major roads or networks of buildings between wetlands must be avoided.

3 Survey and monitoring

3.1 Cumulative probability of detection

3.1.1 Recapping the principle

The cumulative probability of detection (*P*) is the cumulative chance that a species will be detected at a location at which it occurs given multiple surveys at that location (see section 1.1.4). It is a simple function of the probability of detection (*p*) on a single survey (equation 3).

The cumulative probability of detection is a very helpful statistic for designing surveys for wildlife. The idea is to set a threshold for *P* (which equates to some desired chance of detecting the species at the locations at which it occurs), and then to calculate how many surveys are required at each site to reach that threshold. For example, say that one wants to have a 90% chance of detecting the species at each of the locations at which it occurs (i.e. $P = 0.9$), and knows that the probability of detection during a single survey is 0.6. According to equation 3 (section 1.1.4) the number of surveys required per site to reach this threshold is three, because $P = 0.84$ after two surveys and 0.93 after three surveys.

Furthermore, if one knows how the probability of detection is related to certain survey-level variables (e.g., weather conditions, time of year, effort) it is possible to specify a series of protocols with respect to these variables, and to estimate the number of surveys required per site under each protocol. To illustrate, say that the probability of detection for a particular species varies according to survey duration, being 0.4 with a 30 minute survey, 0.6 with a 60 minute survey and 0.8 with a 90 minute survey. One could set these time periods as different protocols for survey, and estimate the number of surveys required to attain a desired cumulative probability of detection under each protocol. According to equation 3, if *P* was set to 0.9 (as in the preceding example) the required number of surveys to reach this threshold would be seven, three and two, respectively.

3.1.2 Effects of survey duration, date and time of day

The extensive survey data collated by Heard and Scroggie (2009) demonstrates that the probability of detection of *L. raniformis* increases with increasing survey duration (measured as person-minutes), decreases as the active season progresses (defined as days since October 1), and is substantially lower during the day than at night (see Figure 3). Figure 8 displays estimates of the cumulative probability of detection of *L. raniformis* when surveys are conducted in each of the six months of the primary active season of the frog (October–March), using one of six protocols for survey duration and time of day: surveys lasting for 30, 60 or 90 person-minutes, and conducted during either the day or night.

Estimates of the cumulative probability of detection do not vary substantially between months, as there is only a weak relationship between 'date' and the probability of detection. Similarly, the cumulative probability of detection does not varying markedly between the three protocols with respect to survey duration. The primary variation is between surveys conducted during the day and night: the much higher probability of detection during nocturnal surveys ensures that the cumulative probability of detection is vastly superior when all surveys are conducted at night rather than during the day.

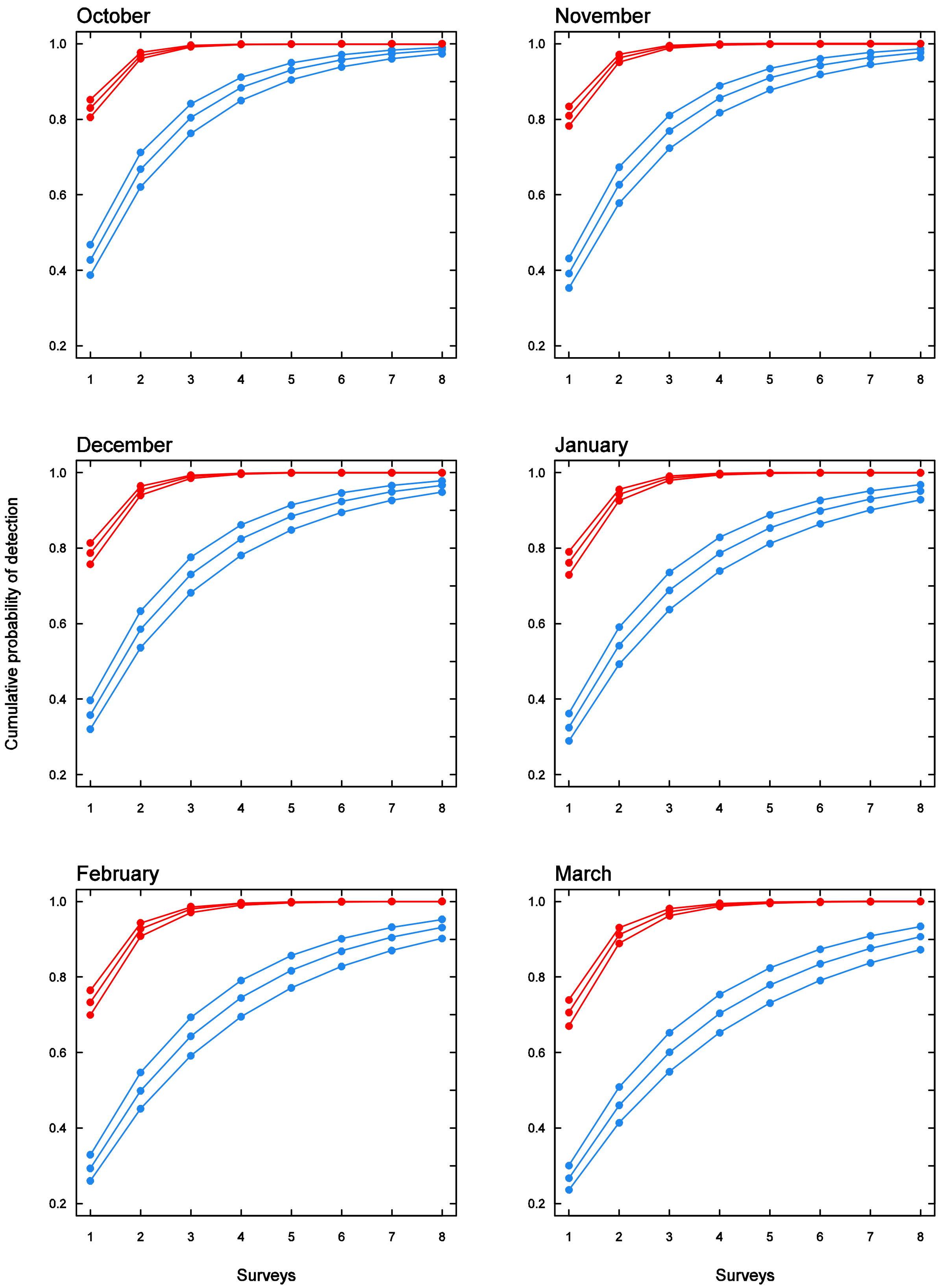

Figure 8. Estimates of the cumulative probability of detection of *Litoria raniformis* in each month of the primary active season, for surveys conducted during the day (blue lines) or night (red lines), with varying survey effort (person-minutes). Lower lines represent surveys lasting for 30 person-minutes, middle lines are surveys lasting for 60 person-minutes, and upper lines are surveys lasting for 90 person-minutes.

3.1.3 Thresholds

In the interests of developing straightforward survey guidelines for *L. raniformis*, it was preferable to set a standard protocol for survey duration and time of day, and to estimate the number of surveys required month-by-month under that protocol. The protocol chosen involves surveys conducted at night for a period of 60 person-minutes. The choice between day or night surveys was simple, since night surveys give substantially higher cumulative probabilities of detection compared to day surveys. Survey duration of 60 person-minutes was selected because it provided a balance between minimising survey duration and minimising the number of repeat visits.

As with the selection of a threshold equilibrium probability of occupancy, selecting a threshold cumulative probability of detection is an exercise in risk assessment. The risk in this case is the non-detection of population, leading to incorrect inferences about wetland occupancy or changes in occupancy through time. Two thresholds of the cumulative probability of detection were used in developing these guidelines: 0.95 and 0.99. Both were considered manageable targets for programs aimed at establishing and monitoring wetland occupancy by *L. raniformis* in urbanising landscapes.

Estimates of the cumulative probability of detection under the chosen protocol (surveys at night extending for 60 person-minutes) are depicted in Figure 9. The results are clear:

- For a threshold of 0.95, at least <u>two</u> surveys are required when surveys are conducted in October–December, whereas <u>three</u> are required in January–March.
- For a threshold of 0.99, the required number of surveys rises to <u>three</u> in October–November and <u>four</u> in December–March.

An important addendum here is the fact that these estimates implicitly assume that surveys are conducted under particular weather conditions. The survey data collated by Heard and Scroggie (2009) were collected almost exclusively under conditions thought to be suitable for activity by *L. raniformis*; that is, night-time air temperatures above 12°C with little or no wind. Thus, in addition to a protocol of only conducting surveys at night for 60 person-minutes, the above estimates are based upon the conduct of surveys when night-time air temperature exceed 12°C and there is little or no wind.

3.2 Guidelines for survey and monitoring

3.2.1 Basic survey techniques

Surveys to detect *L. raniformis* should follow the general procedures for the conduct of visual encounter surveys for amphibians (Crump and Scott 1994). They should commence by quiet observation from the water-line for a period of 10 minutes, listening for calling males. Call playback may be applied during this period to stimulate calling by males that are present but not calling spontaneously. Weather data (see below) can be obtained during this period.

The site should then be systematically searched for active *L. raniformis* with the aid of spotlights. Frogs may be found floating among submerged and floating vegetation, perched in emergent vegetation or bank-side grasses, perched on bank-side rocks or logs, or active on the bank in areas with little ground-level vegetation (see Heard et al. 2008a).

All such areas should be visually inspected while traversing the length of the site (for pools along streams) or its circumference (for off-stream wetlands). Individual frogs are likely to be seen directly, or initially by their eye-shine. Waders are useful for accessing deeper water to confirm the identity of frogs observed. Binoculars are also useful for this purpose.

Active searching by lifting bank-side rocks, logs or surface debris (e.g., sheets of tin or other refuse) should also be conducted wherever practical. Not all individuals will be active each night, and it is not unusual for detections to arise solely from observations of individuals sheltering beneath surface cover. These searches, and searches for frogs on the ground, should be conducted within about 20 m of the waterline.

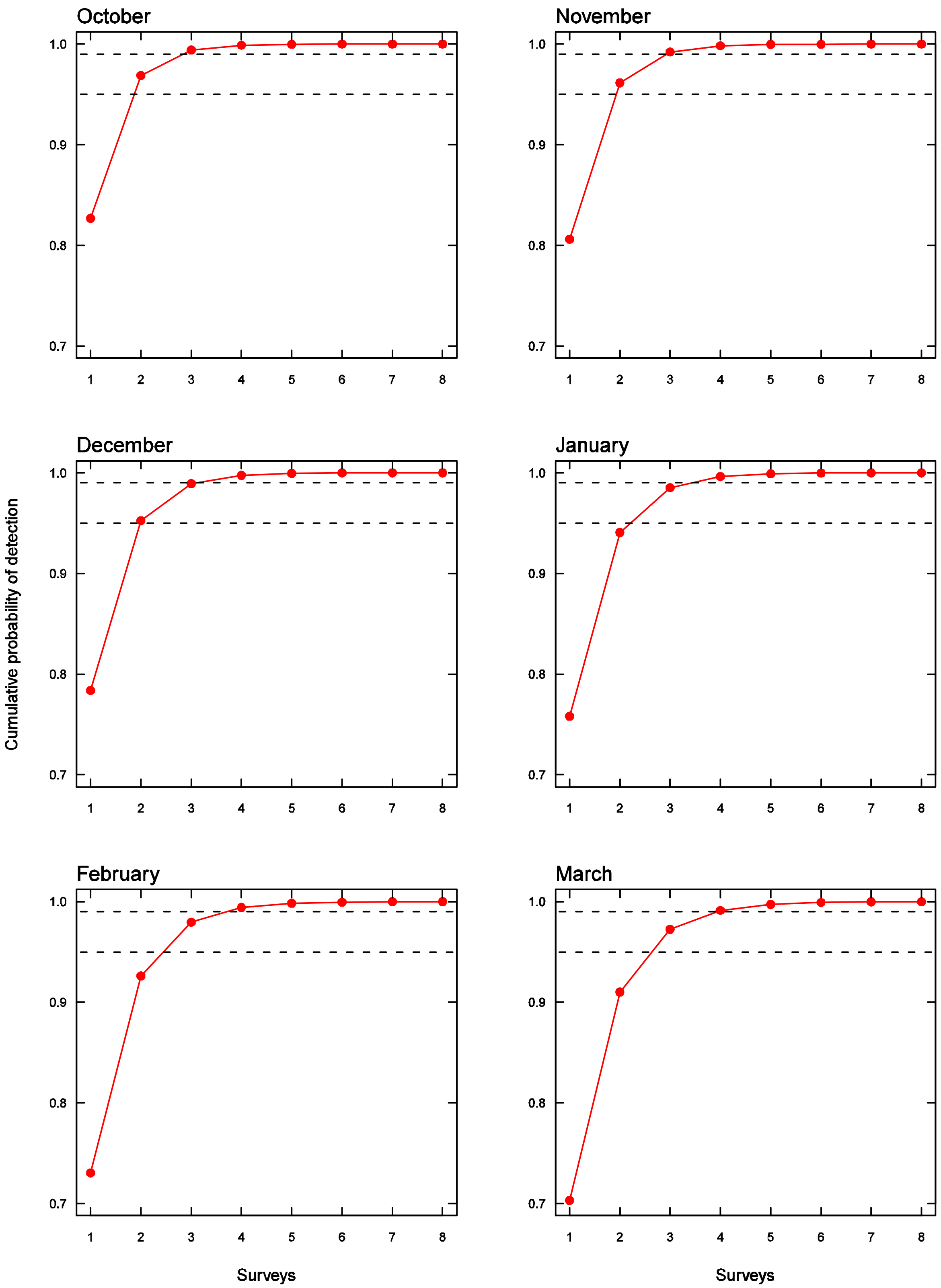

Figure 9. Estimates of the cumulative probability of detection of *Litoria raniformis* during each month of the primary active season, when surveys are conducted at night and extend for 60 person-minutes. The dashed horizontal lines represent the two thresholds of the cumulative probability of detection: 0.95 and 0.99.

Standardised data sheets should be developed and used for recording all data during each survey. Essential data to record are:

- Site location, coordinates (latitude and longitude, or map and grid reference) and a site reference number (unique site code)
- Date of survey and survey number (e.g., 1 of 3)
- Start and finish times
- Personnel involved
- Weather conditions (at least air temperature and wind strength, but also water temperature, relative humidity and rain intensity)
- Detection or non-detection of *L. raniformis* during call survey
- Detection or non-detection of *L. raniformis* during visual census.

3.2.2 Surveys to determine occupancy during a single season

There are two primary scenarios in which surveys aimed at establishing the occupancy of wetlands by *L. raniformis* at a single point in time would be conducted in urbanising landscapes. The first is as part of an environmental impact assessment for a specific development. The second is during some wider project to ascertain the distribution and status of *L. raniformis* across a particular region, such as within a particular catchment or urban growth corridor.

In the first case, site selection is essentially determined by the extent of the development, within which all potential habitat for the species (pools along streams, off-stream wetlands) should be surveyed. However, it is imperative to also survey *all* wetlands within a 1 km radius of the target wetlands, recalling that judgements about the conservation of any remnant populations will depend on knowledge of the species' distribution over this spatial scale (section 2.2.3). In the second case, it is highly desirable to survey *all* wetlands within the proposed study area. The reason also stems from the spatial dependency between populations of *L. raniformis*: without a complete knowledge of the occupancy pattern of the species, connectivity for remnant populations may be underestimated, and misleading inferences derived about the likelihood of population persistence using the approaches advocated above.

Surveys must to be conducted within the primary active season (October–March), following the protocols described above. The timing of surveys within the season is quite flexible, but for simplicity it is advisable to begin and complete all the required surveys at each site within the block of months in which that requirement does not change. When pursuing a threshold cumulative probability of detection 0.95, these block are October–December (two surveys required), and January–March (three surveys required). If pursuing a threshold of 0.99, these blocks are October–November (three surveys required) and December–March (four surveys required).

A targeted survey for an environmental impact assessment will only require qualitative analysis of the data in most circumstances. However, for regional surveys estimation of the probability of occupancy (ψ) and its derivatives (e.g., estimates of the number of occupied sites) is desirable, because such estimates can be used to quantify the regional status of the species (is it rare, moderately abundant or abundant?), and may also be used as a baseline against which future changes in occupancy can be assessed (see below). The repeat sampling scheme advocated here allows new and sophisticated analytical techniques to be applied to these problems (MacKenzie et al. 2006; Royle and Dorazio 2008). Assistance should be sought from a biometrician to implement these techniques.

3.2.3 Surveys to monitor changes in occupancy through time

There are two primary scenarios in which surveys aimed at monitoring temporal changes in wetland occupancy by *L. raniformis* are conducted in urbanising landscapes. The first is as part of a monitoring program designed to assess the impact of a particular development. The aim in this scenario is normally to assess whether specific populations persist through time, from which judgements can be made about the success or failure of specific habitat management initiatives. The second entails monitoring changes in wetland occupancy over entire catchments, growth corridors, etc., to gauge changes in the regional status of the frog. These surveys may be conducted simply to understand the regional status of the species, or to assess the effectiveness of a regional habitat management strategy.

When monitoring to assess the impact of particular developments, monitoring should not only encompass the target populations but also all wetlands at the site (including those which have been enhanced or created as part of a management plan) and within the surrounding landscape (i.e,. within a 1 km radius of the target populations). The maintenance of connectivity at or above levels specified in the preceding sections will obviously be an important additional criterion for judging the success or failure of particular habitat management initiatives for *L. raniformis*, meaning that information on the occupancy statuses of surrounding wetlands will be required.

Regional monitoring programs often involve selecting a random subset of monitoring sites from a larger pool of sites, with the assumption that changes in occupancy at those locations will be representative of the wider trend. However, for *L. raniformis* it is desirable to monitor all potentially inhabitable sites (i.e. all pools along streams and all off-stream wetlands) or, where wetlands are strongly clustered, to select several clusters for monitoring and monitor all sites in each cluster. The reason again stems from the strong dependence of occupancy of any individual wetland on the occupancy statuses of its neighbours. Without knowing these statuses in each year, connectivity for each individual wetland cannot be calculated, which in turn confounds attempts to estimate the probability of occupancy and its derivatives. As discussed below, it is these estimates upon which inferences about regional trends should be based.

In each year, the timing, conduct and number of surveys per site should follow the protocols described above. With regard to regional-scale monitoring programs, it may not be possible to survey all sites within the block of months in which survey effort remains the same. In this case, randomisation could be used to split sites between these blocks. All surveys at each site would then be completed within the block of months assigned to that site.

A crucial factor in any monitoring program is its duration. The duration will vary from case-to-case, depending on a variety factors such as the objectives of monitoring, funding and logistical constraints, and perhaps preliminary results. It is therefore difficult to provide any generally applicable rule with regard to the duration of a monitoring program for *L. raniformis*. However, the duration should, at the very least, encompass the expected period over which the effects of management will become apparent. Researchers are encouraged to think carefully about these timelines, taking into account that there may be considerable lags between the implementation of a management action and its effect on the populations under study.

Monitoring data collected during development-specific programs are unlikely to be suitable for quantitative analysis. However, in regional monitoring programs it is imperative that the data are used to produce robust estimates of changes in occupancy through time. There are two primary ways this may be achieved. First, the single-season approach described above may be used to produce annual estimates of the probability of occupancy (or the number of occupied sites), from which an assessment of a trend can be made. The second (and much more flexible) approach is to use multi-season occupancy models (MacKenzie et al. 2006; Royle and Dorazio 2008). These models can be used to produce annual estimates of the probability of occupancy (or the number of occupied sites) and also to estimate annual probabilities of extinction and recolonisation. The latter may be very useful for diagnosing the reasons behind a particular trend in occupancy and manipulating habitat management to suit. Regardless of the modelling approach taken, assistance from a biometrician should be sought when implementing these techniques.

3.3 Other objectives and approaches

Wetland occupancy is the obvious variable to measure and monitor for *L. raniformis* in urbanising landscapes, because (i) knowledge of occupancy underpins the application of the above guidelines for habitat management, (ii) occupancy is easily assessed, and (iii) robust statistical techniques are now available for monitoring changes in occupancy over time. Nevertheless, they may be circumstances in which variables other than occupancy may be of interest to managers of this species. For example, among a set of populations, one may want to establish which are breeding populations and which are not, perhaps as a means of prioritising which populations should be the focus of works to improve habitat quality. One may be interested in variation in population size and survival rates for similar reasons, or one may want to know something of dispersal rates and pathways to understand how a particular development (such as a new road) would affect population interactions. The following sections briefly consider such objectives, and the approaches that may be applied to achieve them.

3.3.1 Establishing and monitoring reproductive status

Litoria raniformis breeds aquatically, with a cycle that typically includes the commencement of calling by males in October, egg-laying in October–November, larval (tadpole) development from that period until January or February, and metamorphosis shortly thereafter. While the presence of calling males may seem indicative of a reproductive population of *L. raniformis*, it is in fact a poor measure of reproductive success in this species, because the factors that control the survival of adults differ from those that control the survival of eggs and larvae. Establishing and monitoring the reproductive status of populations of *L. raniformis* requires techniques to detect larvae or metamorphlings.

Surveys to detect larvae

Heard et al. (2006) discussed the effectiveness of funnel-trapping for detecting larval *L. raniformis*, based upon a dataset collected on the northern outskirts of Melbourne – the same study area as that used by Heard and Scroggie (2009). They used commercially available, collapsible funnel-traps made of nylon netting (45 cm long × 25 cm high × 25 cm wide, with a 15 cm long funnel at each end, aperture 5 cm), that were suspended with the aid of floats (to allow tadpoles to breath air) and fitted with fluorescent glow stick lures. Their survey protocol entailed setting five traps at random points along the bank after dark, retrieving them the following morning, and identifying all tadpoles trapped using the keys of Anstis (2002). This protocol did not vary between streams and off-stream wetlands, and did not take variation in wetland size into account.

The results of a re-analysis of the data of Heard et al. (2006) are presented in Figure 10. The implications of this re-analysis are twofold. First, under the protocol employed by Heard et al. (2006), detection probabilities for larval *L. raniformis* are relatively low, which leads to generally poor cumulative probabilities of detection for this life stage. Second, the re-analysis demonstrates a strong decline in the probability of detection of larvae as the active season progresses, which results in a strong decline in the cumulative probability of detection after December (which is not surprising, considering that most larvae metamorphose during January).

Of the four months for which data were available, the re-analysis indicates that December and January are the only months in which a cumulative probability of detection of 0.95 or 0.99 can be attained using the techniques of Heard et al. (2006). From Figure 10 it can be seen that:

- At least <u>three</u> surveys are required per site in December to attain a cumulative probability of detection of 0.95, or <u>six</u> surveys for a cumulative probability of detection of 0.99.
- In January, <u>seven</u> surveys are required per site to attain a cumulative probability of detection of 0.95, or <u>10</u> surveys for a cumulative probability of detection of 0.99.

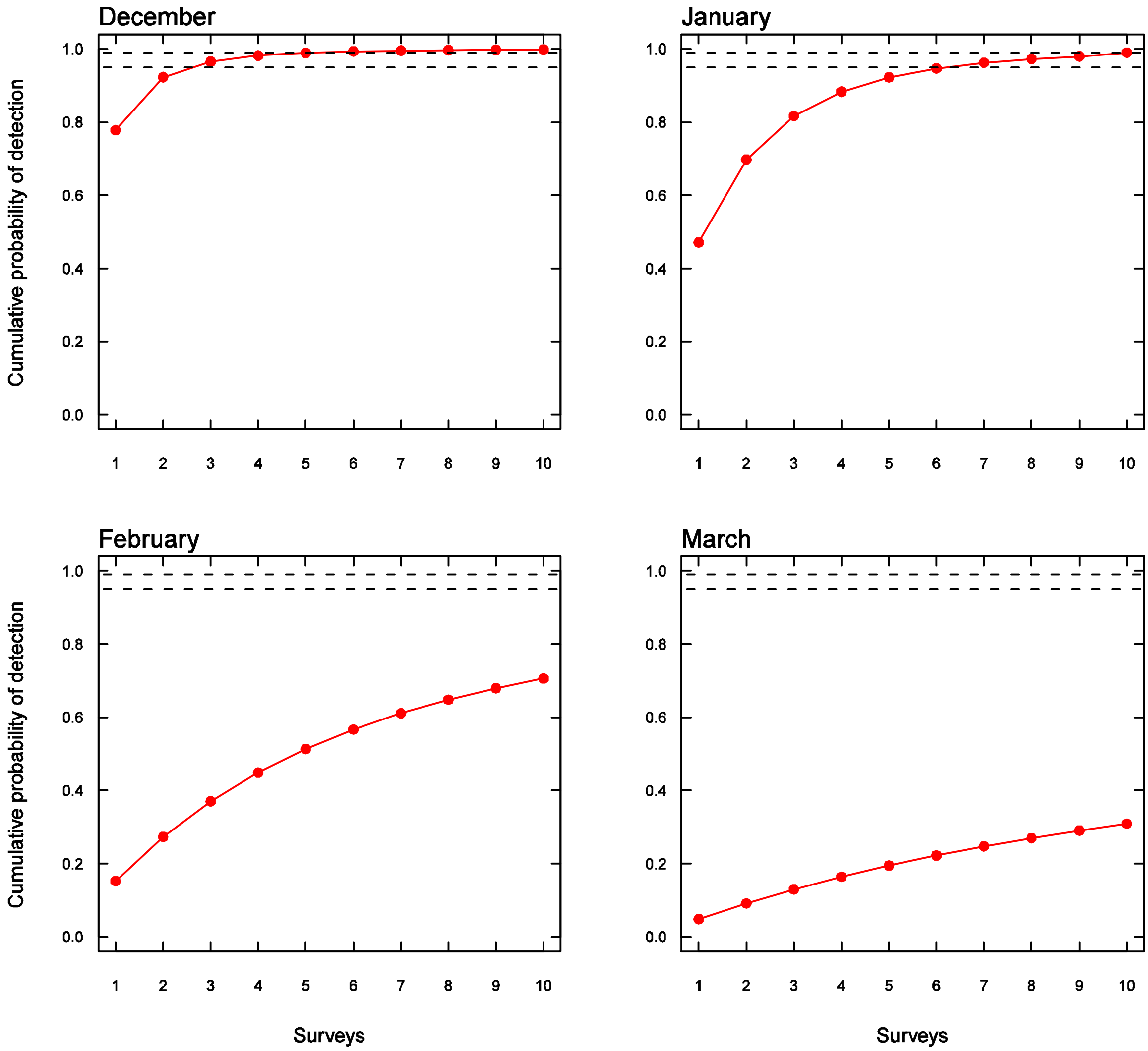

Figure 10. Estimates of the cumulative probability of detection of larval *Litoria raniformis* between December and March using funnel trapping. These estimates relate specifically to the case in which five traps, each fitted with fluorescent glow stick lures, are set shortly after dark and retrieved the following morning. The dashed horizontal lines represent the two thresholds of the cumulative probability of detection: 0.95 and 0.99.

Surveys to detect metamorphlings

Like most aquatic breeding amphibians, larval *L. raniformis* undergo a process of metamorphosis in which they develop limbs, lose their tails and emerge from the water to begin their amphibious life-stage. These newly emerged frogs are called 'metamorphlings'. Whilst data are limited, larval *L. raniformis* appear to take roughly two–three months to reach metamorphosis.

For the purpose of these guidelines, metamorphlings are defined as those still displaying tail-stubs (which they do not fully reabsorb until several days after emergence), or those which have lost their tail-stubs but display a body length ≤ 40 mm. This body length limit is based upon a sample of 104 newly emerged *L. raniformis* measured by Heard et al. (2008b).

Adherence to this definition is important for surveys aimed at establishing or monitoring the reproductive status of populations of *L. raniformis* on the basis of the presence or absence of metamorphlings. The reason is that some frogs disperse to nearby wetlands after metamorphosis, which may lead to incorrect inferences about the occurrence of successful reproduction at those wetlands. However, given that these frogs grow very rapidly post metamorphosis and usually don't disperse for several weeks, one can be fairly confident that juvenile *L. raniformis* with a body length of ≤ 40 mm metamorphosed at the wetland at which they are observed.

Survey techniques for metamorphlings are exactly the same as those for adults: visual encounter surveys conducted after dark with the aid of spotlights. They may be conducted jointly with surveys for adults at times of year when metamorphlings are likely to be present.

Metamorphling detection data available from a subset of the surveys reported by Heard and Scroggie (2009) are helpful in this regard. Not surprisingly, estimates of the cumulative probability of detection derived from these data display the opposite temporal trend to those for tadpoles (Figure 11). Based upon a threshold cumulative probability of detection of 0.95 or 0.99, these estimates show that:

- Surveys for metamorphlings are not feasible until January, when six surveys are required per site to attain the threshold cumulative probability of detection of 0.95, and 10 surveys are required to attain a threshold of 0.99.
- In February, five surveys are required to attain a cumulative probability of detection of 0.95, and seven surveys are required to attain a cumulative probability of detection of 0.99.
- In March, four surveys are required to attain a cumulative probability of detection of 0.95, and five surveys are required to attain a cumulative probability of detection of 0.99.

3.3.2 Mark–recapture and radio-telemetry

Count-based methods such as area-constrained searches have long been considered effective for gathering information about amphibian density and population size. However, it is now widely accepted that these techniques will produce severely biased estimates of population size in many situations, because they assume that detection probabilities are perfect (i.e., all individuals present will be detected) or constant in space and time (Williams et al. 2002). This assumption is particularly dubious for amphibians because of their small size, cryptic nature and erratic patterns of activity (Mazerolle et al. 2007).

Attempts to quantify population size in amphibians subsequently require the application of mark–recapture techniques. 'Mark–recapture' broadly describes techniques in which individual animals are permanently marked, and capture (detection) histories for those individuals obtained during repeated site surveys (Williams et al. 2002). Specific statistical techniques can then be applied to these data to derive estimates of population size that account for imperfect or heterogeneous detection probabilities, or both. The data can also be used to derive estimates of survival and recruitment rates, and, where surveys are completed at multiple sites, to infer rates of dispersal between sites.

For *L. raniformis*, the only practical means of mark–recapture is to repeatedly survey wetlands using the techniques described above, capture by hand all frogs observed (directly, or with the aid of a net), and permanently mark them by subcutaneously implanting either a passive integrated transponder (PIT) tag in adults or a visible implant alphanumeric (VIA) tag in metamorphlings and juveniles (Hamer and Organ 2008; Heard et al. 2008b; Heard and Scroggie 2009). The basic procedure entails tagging frogs upon their first capture (either on-site immediately following capture or in the laboratory the following day), releasing them shortly thereafter, and then recording subsequent captures during the remainder of the monitoring period. The intensity and length of a mark–recapture study depends on the objective of the study. Where it is to measure population size, surveys are best conducted frequently for a short period (e.g., Hamer and Organ 2008). However, where the objective is to measure survival, recruitment or dispersal rates, a longer monitoring regime is required (Heard and Scroggie 2009).

An alternative to mark–recapture for quantifying the dispersal rates and behaviours of *L. raniformis* is radio-telemetry. Radio-telemetry entails fitting individual animals with radio transmitters, and then tracking their movements with the aid of a directional radio receiver (Kenward 2000). It is applicable only to adult *L. raniformis* because current transmitters are too large to be attached to juveniles. The basic approach entails capturing frogs by hand, fitting them with a small transmitter (1–2 g) attached to a purpose-built waist belt, releasing them at their point of capture and then relocating them at some defined interval (e.g., daily) for the life of the transmitters battery (currently about 8–10 weeks; see Wassens et al. 2008). Dispersal rates and behaviours for the population are inferred from those displayed by the tracked individuals.

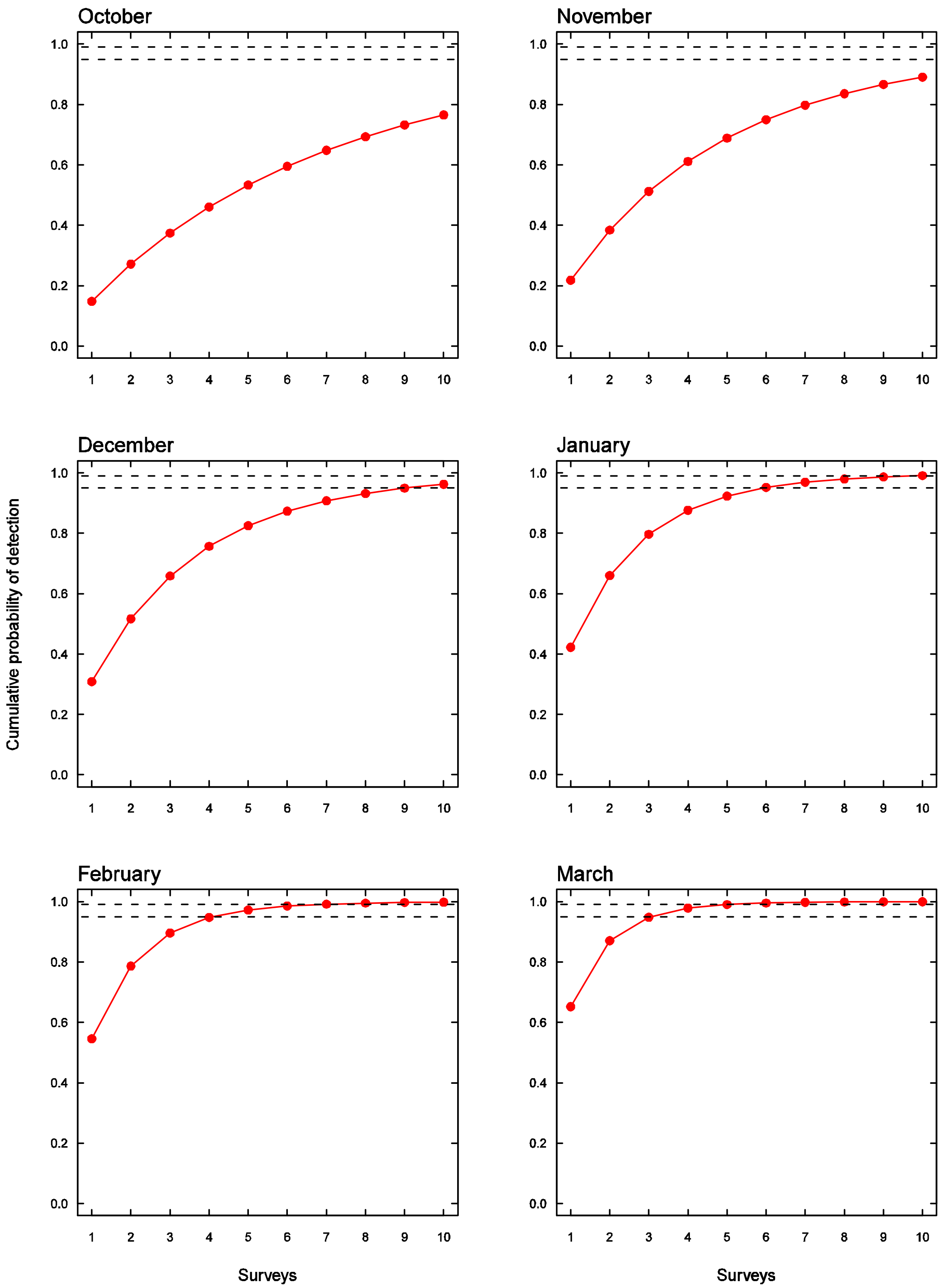

Figure 11. Estimates of the cumulative probability of detection of metamorphling *Litoria raniformis* during each month of the primary active season, when surveys are conducted at night. The dashed horizontal lines represent the two thresholds of the cumulative probability of detection: 0.95 and 0.99.

Mark–recapture and radio-telemetry are potentially very useful for gaining an understanding of the management needs of *L. raniformis* in urbanising landscapes, but their application is also problematic. Both are very expensive and require specialised equipment, they are extremely time-intensive, require dedicated permits and (usually) approval by an animal ethics committee; and most importantly, specialised knowledge is required to implement them properly. For these reasons, and because of their relevance and ease of implementation, the occupancy-based approaches describe above are better options for research on the management needs and responses of *L. raniformis* in urbanising landscapes. Mark–recapture and radio-telemetry is recommended only when occupancy-based approaches are demonstrably inadequate for answering management questions.

3.4 Minimising the risk of disease transmission

No matter which of the above approaches is taken, an important consideration is the possibility that researchers may spread pathogens between remnant populations of *L. raniformis*, or between individuals within a population. The spread of Chytrid Fungus (*Batrachochytrium dendrobatidis*) is of particular concern in this regard, because it has been implicated in the decline of *L. raniformis* and numerous other Australian frog species (Berger et al. 1998; Mahony 1999). It is imperative that survey and monitoring programs for *L. raniformis* adhere to standard hygiene protocols for the conduct of field research on amphibians, such as those produced by the New South Wales National Parks and Wildlife Service (NPWS 2001).

4 Experimental management approaches

4.1 Experimental approaches and conservation

Conservation biology has been described as a crisis discipline whose practitioners are often required to make decisions based as much on intuition as knowledge (Soulé 1985). While this is true to some degree, this perception of conservation biology can be dangerous because it could be used to justify the implementation of actions for which there is no reliable evidence of their efficacy. The application of such experimental approaches to management could produce a conservation 'house of cards' in which the future of a population, or even an entire species, rests upon untested assumptions.

4.1.1 Underpasses and translocation

Two examples of such experimental approaches to management for *L. raniformis* are the construction of underpasses under roads, and the translocation of populations from wetlands designated for destruction. Both have been applied with some frequency around Melbourne in recent years.

Underpasses entail the construction of tunnels under roads that are close to remnant populations of *L. raniformis.* The intention is that these tunnels would be used by frogs during dispersal, thereby helping to maintain the ability of populations to interact. The tunnels are usually concrete pipes or culverts, with drift fences of wire mesh designed to funnel dispersing frogs towards the pipes. Their use is based upon the apparent success of underpasses to maintain the migration routes of some European and North American amphibians (e.g., Langton 1989). However, there is no evidence that such underpasses will be used by *L. raniformis* during dispersal, nor that they are effective in maintaining historical (i.e. pre-road) rates of dispersal between populations. Despite their use in recent years there has been no intensive research on their effectiveness for *L. raniformis*, or any other Australian frogs.

Translocation entails the capture and movement of *L. raniformis* from wetlands that are to be destroyed to make way for urban development, to either existing nearby wetlands or newly created ones. The aim of translocation is to ensure that there is either no net reduction in the abundance of *L. raniformis*, or no net reduction in the number of populations present. The first point to make is that the former aim – to ensure there is no net reduction in abundance – is a poor one for *L. raniformis*, because the end result is still a reduction in connectivity for each of the remaining populations, and subsequently, a reduction in their chances of persistence in the long-term. The second point to make is that, just like underpasses, there is currently no evidence that populations of *L. raniformis* can be successfully relocated. In fact, the few attempts to judge the success of translocations for this species and the closely-related Green and Golden Bell Frog (*L. aurea*) indicate that they have failed (Smith and Clemann 2008; White and Pyke 2008). Indeed, there has been considerable debate over recent years about the general effectiveness of translocation as a conservation strategy for amphibians (Burke 1991; Dodd and Seigel 1991; Reinert 1991; Seigel and Dodd Jr 2002; Trenham and Marsh 2002). There are also inherent risks involved in translocation at both the population- and individual-level. The possible spread of disease is an example of the former, and heightened predation-risk an example the latter.

4.1.2 Application

Because of the experimental nature of underpasses and translocations, they should not at present be regarded as primary measures to mitigate or offset the impacts of urban development on populations of *L. raniformis* (see also DEWHA 2009). Priority should always be given to managing habitat according to the guidelines provided here, which are based upon many years of field research on the population dynamics of the species.

This does not mean that underpasses and translocations have no place in the management of *L. raniformis* in urbanising landscapes, but rather that any application of these techniques should be seen as a management experiment, secondary to the primary goal of managing habitat appropriately, and whose outcome is not crucial to the conservation of the target population or populations. With the application of sound experimental designs and adequate levels of data collection, these experiments will be useful for determining the reliability of these approaches as management techniques for *L. raniformis* in the future.

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Appendix

Aquatic plants which are a suitable for cultivation to enhance habitat quality for *Litoria raniformis* in urbanising landscapes. These plants are commonly found in wetlands occupied by *L. raniformis* around Melbourne, and are utilised by the frog during both diurnal and nocturnal activity.

Vegetation type	Species
Emergent	Water Ribbon (*Triglochin procerum*)* Cumbungi (*Typha orientalis*) Tall Spikerush (*Eleocharis sphacelata*) River Clubrush (*Schoenoplectus validus*) Water Plaintain (*Alisma plantago-aquatica*) Mud Dock (*Rumex bidens*) Common Spikerush (*Eleocharis acuta*)
Submergent	Curly Pondweed (*Potamogeton crispus*) Blunt Pondweed (*Potamogeton ochreatus*) Fennel Pondweed (*Potamogeton pectinatus*) Hornwort (*Ceratophyllum demersum*) Eelgrass (*Vallisneria gigantean*)
Floating	Water Ribbon (*Triglochin procerum*)* Floating Pondweed (*Potamogeton tricarinatus*) Swamp Lily (*Ottelia ovafolia*) Ferny Azolla (*Azolla pinnata*) Duckweed (*Lemna* spp.)

*Water Ribbon occurs in both emergent and floating forms.

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