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Population viability assessment and sensitivity analysis as a management tool for the peri-urban environment

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Abstract Anthropogenic disturbance occurring within urban ecosystems is often extreme and highly variable. A quantifiable measure of their effect on the persistence of urban wildlife populations would contribute to conservation efforts. This study suggests that population viability assessment, a commonly utilized modeling tool for creating management strategies for rare and threatened wildlife populations, is also appropriate in an urban context. It can be used to create proactive management strategies that quantify the impacts of anthropogenic disturbances and rank a range of management options within an active adaptive framework. To show this, population viability assessment and sensitivity analyses were run to forecast the population trends of a seemingly robust but isolated swamp wallaby (*Wallabia bicolor*) population living in peri-urban Sydney, Australia; a population exposed to anthropogenic disturbances from towns, hobby farms and roads. Modeling suggested this population was in a slow decline and that predictions were highly dependent upon stochastic events and the precision of reproduction rates. However, a number of management options are identified that will dramatically reduce the risk of total population decline, with complementary options utilized in tandem the most effective.

Keywords Peri-urban · PVA · Management · Wallabia bicolor · Sensitivity analysis

Introduction

Aldo Leopold played a pivotal role in formulating the wilderness concept in the United States (Miller and Hobbs, 2002). Yet during the last decades of his life he focused on small farmsteads in human-dominated landscapes and "... the oldest task in human history; to live

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on a piece of land without spoiling it" (Leopold, 1991). He recognized that large protected areas were not sufficient for conservation. Since then there has been much debate regarding the relationship between land-use and biodiversity (Pimm and Raven, 2004), with the realization that there are too few extensive wild lands to enable the protection of global biodiversity (Grumbine, 1990; McNeely et al., 1994; Newmark, 1995), and that nature will struggle to cope with the expected land-use changes necessary to support a burgeoning human population, hungry for resources (Imhoff et al., 2005; Foley et al., 2005). This is the key conservation issue of our time (Crane and Kinzig, 2005); giving rise to a much broader focus in conservation planning that encompasses protected areas, smaller reserves and unprotected lands (Margules and Pressey, 2000; Miller and Hobbs, 2002). Understanding the threats to wildlife in these often urbanized reserves is crucial to biodiversity conservation as a whole.

It is well known that biodiversity hotspots typically coincide with higher-than-average human population densities and growth rates (Myers, 1988, 1990; Mittermeier et al., 2000). In the USA, urbanization has been identified as a primary cause, singly or in association with other factors, for declines in more than half the species listed as threatened or endangered under the U.S. Endangered Species Act (Czech et al., 2000; Miller and Hobbs, 2002). Similarly, Australia is one of the most urbanized nations in the Asia Pacific region with a population that is 85% urban (UNPD, 2001), mostly along the temperate eastern sea-board. Australia's vast interior is sparsely populated but highly disturbed due to habitat loss caused by exotic livestock (Fisher et al., 2003) and plundering by introduced predators (Dickman, 1994). Yet Australia's urbanized regions also have severely reduced wildlife populations, with many species now considered endangered (Flannery, 2004). Like in many expanding nations, the condition of urbanized landscapes in Australia have deteriorated in recent years, with calls for the taking of strong measures to conserve diversity (Williams et al., 2001).

A study of wildlife sightings around Melbourne, Victoria indicates that the mammal community has been reduced by a third since the time of European settlement in 1856 (van der Ree, 2004). A number of Australia's macropodid species such as the rock wallaby (*Petrogale spp.*) (Flannery, 2004) and the parma wallaby (*Macropus parma*) (Maxwell et al., 1996) occurred in the urban environment and are now endangered or vulnerable. Other more common species such as the swamp wallaby (*Wallabia bicolor*), the red-necked wallaby (*M. rufogriseus*), common wallaroo (*M. robustus*), eastern grey kangaroo (*M. giganteus*) and western grey kangaroo (*M. fuliginosus*) also inhabited what are now urban centers, and as a consequence are relegated to peri-urban environments where they are also showing signs of decline (CSIRO, 1996).

The effects of urbanization also provide a context for answering ecological questions of general importance and applicability, as well as questions that are specific and unique to urbanization (McDonnell and Pickett, 1990). For example, urbanization can provide large-scale experiments to test the effects of habitat fragmentation on ecological communities (McDonnell, 1997). Moreover, the ability to form or alter social attitudes to the environment enhances the value of the urban ecological resource (Rotherham, 1994, 1999). Urban areas are accessible to an increasing portion of the population, in particular decision makers who influence economics and politics at regional and national levels. The exposure of people to wildlife and natural systems can also lead to action to safeguard the environment as public awareness leads to grassroots action, both of which are recognized as effective precursors to environmental management (Jones and Rotherham, 1998; Rotherham, 1999).

In NSW, Australia, the NSW National Parks and Wildlife Service (NSW NPWS) is charged with the mission of 'working with people and communities to protect and conserve natural and cultural heritage in the New South Wales landscape' (NSW NPWS, 2006). Accordingly, NSW NPWS operates in both remote areas and peri-urban parks, striving to maintain the integrity of localized ecosystems. The task is complex as urban ecosystems can include a multiplicity of anthropogenic disturbances that are both ongoing (such as road caused fatalities, disruption by pets, and altered resources) and stochastic (such as human caused fires, sporadic pollution, and habitat loss). Fortunately, many of these disturbances follow localized trends that can be assigned probability values of occurrence. However, the competing interests of local communities, available resources and government directives may impact the decision making process.

Ideally, managers should prioritize management options whilst striving to ultimately preserve the integrity of the reserve under their care. However, the integration of biological assessment of the impact of disturbances and competing interests into a comprehensive management plan is difficult at best. At present the process of prioritizing wildlife management strategies in peri-urban environments is qualitative, typically based on a best guess of experienced reserve managers. Given the ready availability of statistical tools for assessing the impact of conservation strategies, testing the sensitivity of models to a suite of variables, and the quantifiable nature of most disturbances, an important step in addressing the effects of urbanization should be to integrate these into an adaptive management framework that also enables prioritization of management strategies.

An appropriate tool for achieving this is Population Viability Assessment (PVA). PVA is currently used in conservation biology to assess minimum viable populations (Shaffer, 1981; Ayala et al., 1973) and assists in the management of both threatened (Burgman et al., 1993; Possingham et al., 1993; Brook et al., 2000) and abundant populations (McCarthy et al., 1996). It is a reliable and quantitative tool that can be used to identify environmental factors that impact negatively on a specific conservation target, predict the outcome of management strategies, and account for both anthropogenic and environmental stochastic events (Lacy, 1993). We argue that it should be utilized in the urban context as well to enable managers to be proactive rather than reactive in mitigating future threats to the persistence of urban wildlife populations (Banks, 2004).

In this paper we model the impacts of urbanization on an indicator species, the swamp wallaby, and highlight the usefulness of PVA as a management tool in urban areas. We model a well-studied population suspected to be declining to determine which factors contribute most population decline. Management options targeting population decline and predictive uncertainty are modeled and ranked in order of effectiveness.

Methods

Study site and species

The population of swamp wallabies chosen to model exists within Muogamarra Nature Reserve (33°37′35″ S 151°09′20″ E), a forested habitat located 50-km north of the city centre of Sydney, New South Wales, Australia (Fig. 1). Fauna records of the reserve indicate that once present koalas (*Phascolarctos cinereus*), common wombats (*Vombatus ursinus*), and a number of macropodids, such as the common wallaroo (*Macropus robustus*) and the red-necked wallaby (*Macropus rufogriseus*), have been lost (Fallding et al., 1994). The swamp wallaby (*Wallabia bicolor*) is the only medium sized species to have persisted. The swamp wallaby, a mostly solitary marsupial except for reproductive associations, is the only member of the monophyletic clade *Wallabia*, and has vastly different dentition, genetic, reproductive and behavioral characteristics from other wallabies (Merchant, 2002). While the range of the swamp wallaby is thought to be extensive, from Cape York to southern Victoria, its abundance within this range is currently unknown. Although often regarded as

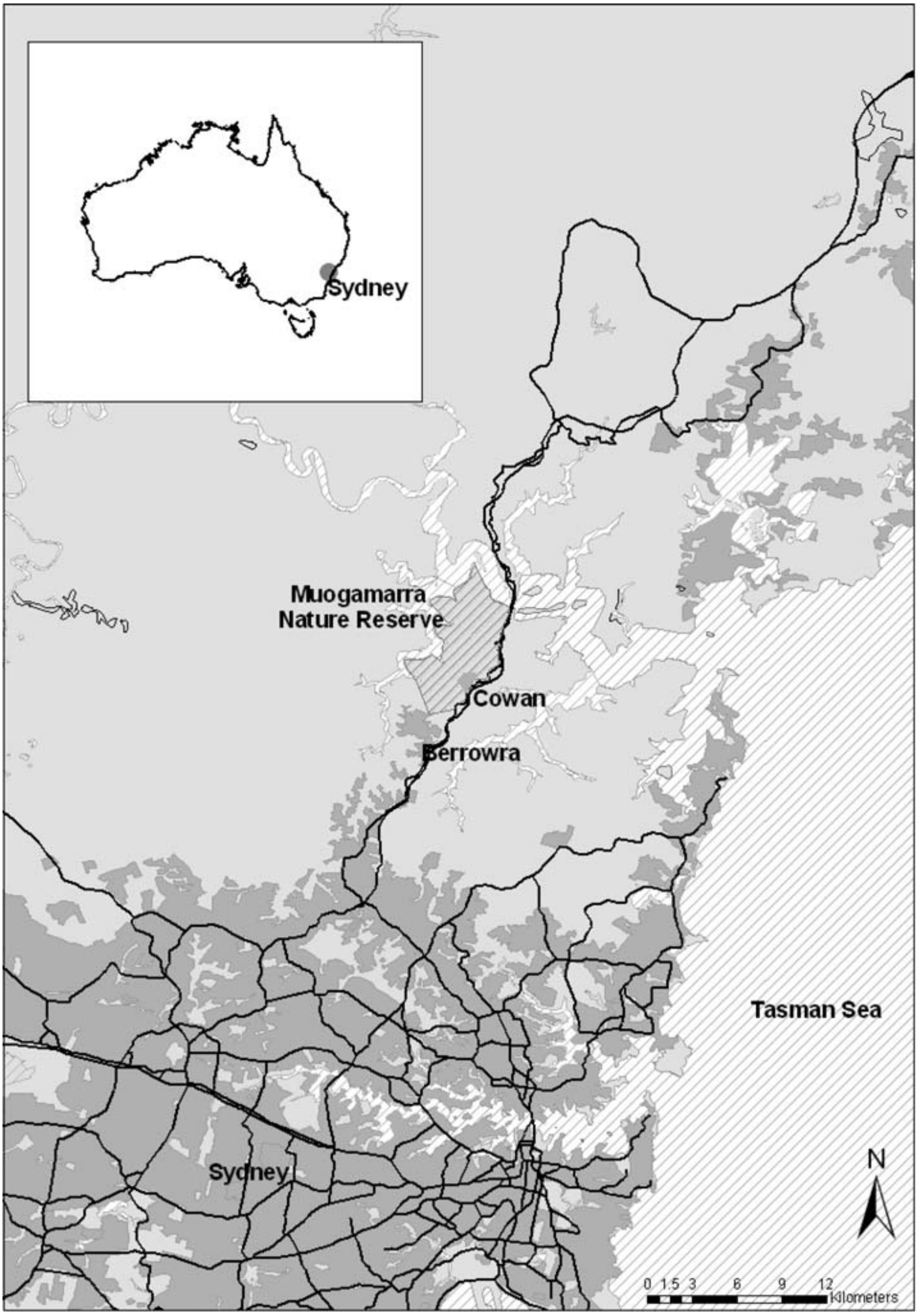

Fig. 1 Muogamarra Nature Reserve (MNR) situated on Sydney's urban fringe. Dark grey areas are developed urban areas. Light grey areas are undeveloped. Stippled areas are water ways

being resilient to disturbance, the preferred habitat of the swamp wallaby typically coincides with land occupied by humans, placing increasing pressure on the sustainability of local populations (Ramp and Ben-Ami, in press).

Muogamarra Nature reserve is part of the Sydney sandstone landscape and is comprised of contiguous gullies of dry sclerophyll forest (Benson and Howell, 1994). Anthropogenic boundaries include a township, several rural properties, two sealed roads and a train-line to the east, sealed and unsealed roads and a large town to the south. Natural boundaries include the Hawkesbury River to the north and Berrowra Creek to the west. A yearly mean precipitation of 1000 mm and many watering holes and ephemeral creeks ensure that water is available year round (Benson and Howell, 1994).

Given the proximity of Muogamarra Nature Reserve to townships and its location within steep terrain surrounded by water, the reserve is not only isolated but is subjected to a variety of anthropogenic disturbances such as habitat loss, road-based fatalities, predation by introduced animals (dogs and foxes) and fire (Ben-Ami, 2005). Current management strategies address fire and fox control in a holistic approach to preserving ecosystem integrity, but do not specifically address impacts on the last medium-sized marsupial within the reserve or provide a criterion for assessing the priority of management strategies (NSW NPWS, 1998). As a consequence, the reserve represents an ideal test case for assessing the usefulness of the proposed approach.

The basic model

A generic PVA model constructed in Vortex 8.0 (Lacey, 1993) was used to assess the long-term survival of the population. See Lacey (1993) for the model details. The assessment of urban impacts on the swamp wallaby population in Muogamarra Nature Reserve consisted of several steps: a basic model using best estimates of parameters; sensitivity testing of the model to ‘variable’ parameters; and the prioritizing of theoretical management actions to target ‘sensitive parameters’. The best estimates of parameter values were used for the basic model (Table 1), modeled over 100 years and replicated 1,000 times. In cases where information was lacking, parameters were estimated from qualitative assessments, anecdotal information and an understanding of the biology of the swamp wallaby.

Life history details of the swamp wallaby were qualified via an extensive body of research (Edwards and Ealey, 1975; Robertshaw and Harden, 1986; Hollis et al., 1986; Troy and Coulson, 1993; West, 1992; Meek and Triggs, 1998; Osawa, 1989; Kirkpatrick, 1970; Floyd, 1980; Lunney and O’Connell, 1988; Harrington, 1976), as well as a long-term study within Muogamarra Nature Reserve itself (Ben-Ami, 2005). Swamp wallabies are mostly solitary (Edwards and Ealey, 1975; Kirkpatrick, 1970), although sub-populations within the reserve typically exist within single gullies, albeit with some inter-gully mixing (Ben-Ami, 2005). The size of core areas of individuals home ranges (Anderson, 1982) vary from 4 to 19 ha with little overlap among individuals (Ben-Ami, 2005). Mature wallabies possess the larger home ranges although differences are not gender biased (Ben-Ami, 2005). Males tend to aggregate on resource deficient hotspots along gully slopes while females show preference for the more resource rich gully bottoms (Ben-Ami, 2005). Although they are not aggressively territorial (Crebbin, 1982) they exhibit strong site fidelity (Edwards and Ealey, 1975; Troy and Coulson, 1993). Their browse consists of a diverse range of native vegetation (Hollis et al., 1986; Osawa, 1990; Harrington, 1976), yet individuals often venture onto the periphery of human development along ridges and plateaus to browse introduced plants (Ben-Ami, 2005; Watson, 1993; Osawa, 1989).

Table 1 Best estimate values of parameters used in Vortex simulations

Parameter	Value	References
Type of mating system	Polygamous	
Age of first reproduction (females)	1 year	Strahan (2002)
Age of first reproduction (males)	1 year	Strahan (2002)
Age after which adults do not reproduce	10 years	Estimate
Sex ratio at birth (proportion males)	0.50	Estimate
% adults in the breeding pool	70	Estimate
Annual Fecundity		
Max. litter size/year	2	Strahan (2002)
% females breeding annually	95 ± 5	Paplinska (2005)
% females with one birth	25	Estimate
% females with two births	75	Estimate
Annual Mortality		
% females (0<=age<=1)	25 ± 9	Robertshaw and Harden (1986)
% Adult females (1<=age<=10)	20 ± 3	Robertshaw and Harden (1986), Ben-Ami (2005)
% males (0<=age<=1)	25 ± 9	Robertshaw and Harden (1986)
% adult males (1<=age<=10)	20 ± 3	Robertshaw and Harden (1986), Ben-Ami (2005)
Inbreeding depression	None	
Catastrophes throughout model period		
Number of catastrophes modeled (fire)	2	
Type of catastrophes	Wildfire	
% Probability of the 1st wildfire	95	NSW NPWS (1999)
Effect on reproduction	0.70	Ben-Ami (2005)
Effect on survival	0.85	Garvey (2005)
% Probability of the 2nd wildfire	50	NSW NPWS (1999)
Effect on reproduction	0.70	Ben-Ami (2005)
Effect on survival	0.85	Garvey (2005)
Carrying capacity (S.D.)	1000	Troy and Coulson (1993), Ben-Ami (2005)
Annual change in carrying capacity, *K*	None	
Initial population	547	Robertshaw and Harden (1986), Troy and Coulson (1993)
Basic supplementation regime	None	
Basic harvesting regime	None	

Fractions were not allowed in the model.

Sexual maturity typically occurs between 15 and 18 months (Merchant, 2002). For modeling purposes we set adulthood at one year as Vortex only processes integer values. We set the percentage of adult females in the breeding pool at 70% on the basis of breeding rates from other studies (Paplinksa, 2005). In New South Wales, swamp wallabies have been found to be both seasonal and continuous breeders, with high predation pressure thought to catalyze a continuous reproductive cycle (Robertshaw and Harden, 1986). The maximal potential breeding frequency is one offspring every eight months (Merchant, 2002) with no hierarchy in access to females by males (Crebbin, 1982). Mating can require eight days before the presence of an established fetus while the gestation period is longer than the oestrous cycle (Kirkpatrick, 1970; Strahan, 2002). After a gestation of 33–38 days a single young is born. Weaning occurs at around 15 months although embryonic diapause enhances the reproductive rate. Since gestation lasts eight to nine months we assumed that 75% of female have one young per year and 25% have two young per year. The effect of environmental variation on reproductive success was set at 5% due to the generalist nature of the swamp wallaby, its ability to exploit a variety of dietary resources and its resiliency to extreme climatic conditions.

The population of swamp wallabies within Muogamarra Nature Reserve has not been completely surveyed. Transect surveys were not possible because of the rugged terrain and dense vegetation (West, 1992). Population surveys from other studies have observed density estimates ranging from 17 per km^2 (Robertshaw and Harden, 1986) to 32 per km^2 (Troy and Coulson, 1993). With a reserve area of 22.34 km^2 (Thomas and Benson, 1985), we estimated that the population should range from 380 to 715 individuals. Taking a conservative approach, the lower and upper bounds of the wallaby population in the reserve were set at 300 and 800 individuals respectively.

Detailed fire records for the past 25 years show that two large-scale fires occurred within the last 10 years, although there were none in the previous 15 years (NSW NPWS, 2005). Reserve management includes regular small-scale prescribed burns around access roads and residences as well as within the reserve (NSW NPWS, 2005). Radio-tracking of mature individuals during fire events in the reserve indicated that survival during wildfires was higher in areas where the fuel load was decreased by prescribed burns (Garvey, 2005).

Non-catastrophe mortality in Muogamarra Nature Reserve was attributed to disease, road-kill, predation and natural death. Road-kill is a significant contributor to macropodid mortality, particularly so for wallaby species (Taylor and Goldingay, 2003; Ramp et al., 2006). From dietary studies it is know that at some locations, in particular in coastal New South Wales, swamp wallabies are the preferred wildlife dietary item of dingoes (Robertshaw and Harden, 1986), feral and domestic dogs (Lunney et al., 1990; Meek and Triggs, 1998), as well as a significant component of fox diets (Augee et al., 1996). The presence and predation of swamp wallabies by dogs and foxes has been confirmed at Muogamarra Nature Reserve through foot print surveys, collection of fecal pellets and photography (Ben-Ami, 2005). Of four (out of 22) radio-collared wallabies that died over the course of one year, two deaths were the result of predation while two were the result of natural causes.

Sensitivity analysis

Due to the uncertain nature of parameters in PVA models it is important to assess predictions over the range of possible parameter values, especially when PVA models are used to rank management options (Possingham et al., 1993). The sensitivity analysis procedures in this study follow the specifications indicated by McCarthy et al. (1996). Parameters incorporated in the sensitivity analysis included fecundity, mortality, reproduction, catastrophes and initial population size (Table 2). The variation of each selected parameter reflected the scale of uncertainty in its estimation as well variability thought to occur in natural populations. Five hundred random combinations of these parameters were derived from the uncertainty ranges.

Ten simulations of the PVA models were run for each parameter combination over a 100 year period. Correlation analysis was conducted to identify significant relationships among variable model parameters. Five thousand replicates were used in a logistic regression model to explore the relationship between the 'variable' parameters and extinction risk and parameters interactions. This level of replication was found to be sufficient for identifying sensitive parameters (McCarthy et al., 1995). The independent variables of the logistic regression were the model parameters. The dependent variable was whether the population fell to zero (0), or not (1), for each of the 500 random combinations of parameters.

Table 2 Influence of plausible ranges of parameters on population viability

Parameter	Plausible range of values	Wald	Sig
Females with one birth	5%,15%, 25%	4.5 (−)	*
Females (0<=age<=1)	5%, 10%, 20%, 30%, 35%	21.8 (−)	***
Adult females (1<=age<=10)	5%, 10%, 15%, 20%, 25%	22.9 (−)	***
Males (0<=age<=1)	5%, 10%, 20%, 30%, 35%	0.6 (−)	0.448
Adult males (1<=age<=10)	5%, 10%, 15%, 20%, 25%	0.3 (−)	0.595
Effect of the 1st wildfire on reproduction[a]	0.2, 0.4, 0.6, 0.8, 1.0	69.5 (+)	***
Effect of the 1st wildfire on survival[b]	0.75, 0.85, 0.95	62.8 (+)	***
Effect of the 2nd wildfire on reproduction[a]	0.2, 0.4, 0.6, 0.8, 1.0	44.6 (+)	***
Effect of the 2nd wildfire on survival[b]	0.75, 0.85, 0.95	27.0 (+)	***
Initial population	300, 547, 800	12.8 (+)	0.584

If the Wald statistic is significant then the parameter is useful to the model. The size of the Wald statistic indicates the relative influence of the parameter on the model. The direction of the parameters' influence on the survival or extinction of the population is indicated by (+) and (−) respectively. A significance of $P < 0.05$ is identified by '*' and $P < 0.001$ by '***'.

[a]Proportion of wallabies surviving.

[b]Proportion of wallabies reproducing.

In the model, the sign of the Wald statistic indicated whether a particular variable contributes to population decline (negative sign) or growth (positive sign). The magnitude of the Wald statistic relative to the range of the uncertainty of the parameter indicated the magnitude of the sensitivity. The model was also simulated 1,000 times for each variable parameter combination using the most extreme values (either contributing to population decline or growth) in order to obtain more precise predictions of population viability under extreme circumstances.

Management actions

The influence of various management actions, targeting those parameters deemed to be sensitive, was assessed. This was done for each parameter separately and in combination with other sensitive parameters. Values for the sensitive parameters were chosen to reflect the potential effect of management actions determined *a priori* (Table 3). A 'predator control - ongoing' option considered prevention of domestic dog predation of swamp wallabies on an ongoing basis. In the 'fire prevention' option, we assumed that through increased surveillance a second large-scale wildfire could be prevented. A second management option targeting fire, 'prescribed burns,' modeled an increase in ongoing prescribed burns throughout the reserve, rather than just on the periphery, such that whilst large-scale fires still occurred they were of low intensity. In the 'predator control after large-scale fires' option, domestic dogs would be actively removed only after large-scale wildfire, when prey are most sensitive to predation. The aim of this particular management option was to highlight the benefit of strategic short-term enforcement of dog removal. Finally, the 'road-kill' management option modeled the prevention of road-kill on the F3 Freeway and the Pacific Highway that run parallel to each other and mark the eastern border of the reserve. Values for the influence of the various management options on parameters in the model were determined from the known biology of the swamp wallaby, related examples from previous studies, and the unique characteristics of the study site. The model was applied 1,000 times for each management action over a 100 year period.

Table 3 The influence of management actions on population extinction

Management action	Parameters	P	PE_{50}	PE_{100}	R
Prevention of the 2nd wildfire:	1. % Probability of the 2nd wildfire	0	0	0	8
Implementing control burns	1. Effect on survival (after 1st fire)	.90	0	0	2
	2. Effect on survival (after 2nd fire)	.90			
Minimizing predation caused mortality	1. Juvenile female (0<=age<=1)	15	0	0	14
	2. Adult female (1<=age<=10)	10			
Predator control after wild fires	1. Effect on survival (after 1st fire)	.90	0	0	2
	2. Effect on survival (after 2nd fire)	.90			
Reducing road-kill	1. Juvenile female (0<=age<=1)	20	1	13	−1
	2. Adult female (1<=age<=10)	15			

P is the estimated parameter value in the management action; PE_{50} and PE_{100} are the probabilities of population extinction at 50 and 100 years; R is the mean growth. The higher the R value the greater the effectiveness of the population management action in increasing the swamp wallaby population in the reserve. A negative R value implies population decline. All values are in percent.

Results

The basic model

The basic PVA model using the best parameter estimates indicated a drop in population from an initial 547 wallabies in Muogamarra Nature Reserve to 104 ± 139 (mean of successful populations ± SD, Fig. 2), with a 12% chance of extinction within the next 50 years (Fig. 3(b)). In contrast, over a period of 100 years, the model indicated a final population of 51 ± 80 wallabies (Fig. 2) with a 71% chance of extinction (Fig. 3(b)).

Sensitivity analysis

A best-case scenario returned a population growth rate of 0.53, capping at the carrying capacity of 1,000 (Fig. 3(a)) and having an extinction probability of 0% (Fig. 3(b)). In contrast, the worst-case scenario returned a growth rate of −0.67 and extinction within 20

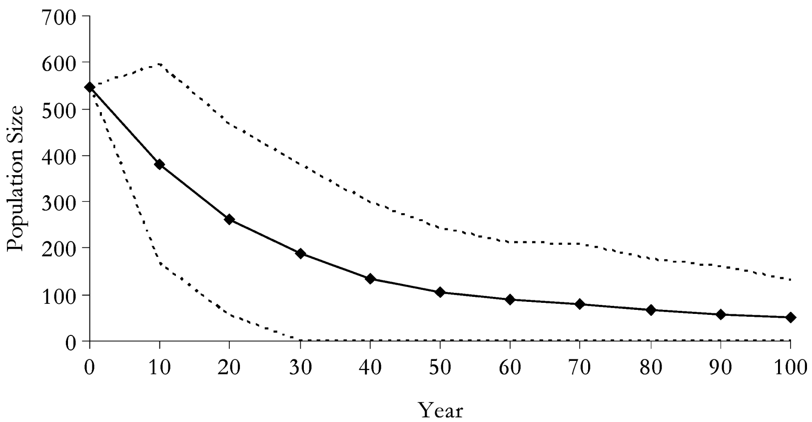

Fig. 2 Best estimate for mean populations (±SD). The solid line indicates the base line model derived from the best estimates. The stippled line indicates the upper and lower limits of the standard deviation

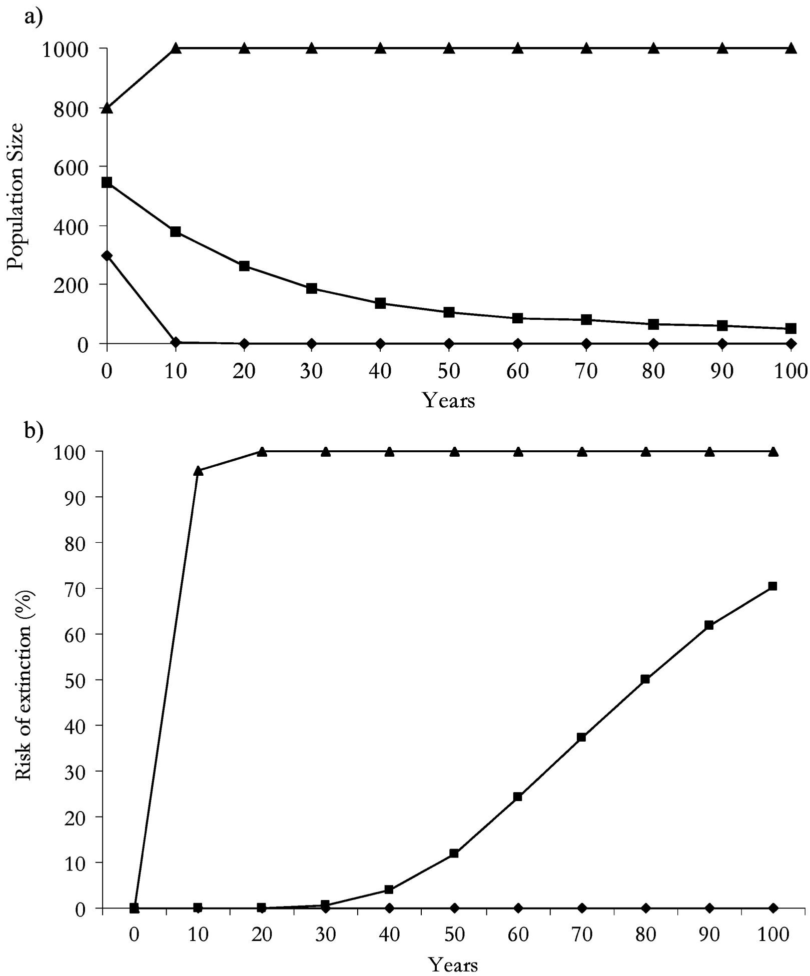

Fig. 3 Best and worst case scenarios for population viability in Muogamarra Nature Reserve over the next 100 years. The graphs indicate: (a) The population size over the next 100 years, and (b) the risk of extinction over the next 100 years. Diamonds indicate the low estimate (lowest values in Table 2); squares indicate the best estimate (Table 1); and triangles indicate the upper estimate (high values in Table 2)

years (Fig. 3(a)). Even after 10 years the population had a probability of extinction of 95.8% (Fig. 3(b)).

Of the variable parameters tested (Table 2), neither annual male mortality nor initial population had any significant influence on population viability. Survival and reproductive rates were most strongly influencing population viability after the first large-scale catastrophe (fire) that had a 95% probability of happening within a 50-year period (Table 2). The same parameters also strongly influenced viability after the occurrence of a second catastrophe (large-scale fire) that had a 50% probability of occurring. Increasing mortality rates (separate from catastrophic events) of adult and young females and decreasing annual female fecundity also had a significantly negative influence on the long-term viability of the population.

Only small correlations were found between the variable parameters used in the sensitivity analysis. Survival of the first catastrophe (large scale fire) and reproduction after the second (large scale fire) were positively correlated ($R^2 = 0.34$, $P < 0.05$); while reproduction after the first and survival after the second were also positively correlated ($R^2 = 0.24$. $P < 0.05$).

Management actions

Listed in order of effectiveness the management options are: 'predator control - ongoing,' 'fire prevention,' 'prescribed burns,' 'predator control after large-scale fires' and 'road-kill prevention' (Table 3). Ongoing predator control was the most effective management action examined (Table 3), as the swamp wallaby is the preferred dietary item of dogs and also a significant component of the fox diet. With the complete removal of predation, the annual mortality of young (< one year old) and adult females decreased by about half of the current best estimates of 25 and 20%, respectively. Under this scenario, the population is projected to experience a positive growth trend of 14% per year (Table 3).

If only one major fire were to occur every 50 years, rather than two, no other management actions would be necessary (Table 3). Under this scenario, over the next 50 and 100 years, the population would only be limited by the carrying capacity of the reserve and the risk of extinction would be minimal. In the last 10 years two major wildfires occurred in the area. If the necessary resources are not available for the presumably high increase in surveillance, a second option could be to increase the extent of prescribed burns in the reserve. This action increases the chance of swamp wallabies surviving the wildfire, and, as indicated by the sensitivity analysis, even a slight increase in swamp wallaby survival of large-scale wildfires, from 85 to 90%, can ensure the population's survival (Table 3).

If, for cost or logistical reasons, predation could only be reduced sporadically, then the optimal time for this would be immediately following large-scale wildfires. Predator control directly after wildlife led to only a small increase in swamp wallaby survival (5%), but this small increase had a strong influence on long-term population viability with a forecasted growth rate of 2% per year and 0% probability of extinction in both 50 and 100 years (Table 3). The reduction or elimination of road-kill along the Pacific Highway and the F3 Freeway adjacent to Muogamarra Nature Reserve may also decrease annual mortality and therefore greatly increase the chance of the population's survival. However, even if road-kill was completely eliminated from the system, resulting in reduction of adult and young female mortality, the risk of population extinction would not be completely removed, resulting in forecasted extinction values of 1 and 13% at 50 and 100 years and a growth rate of −1% per year (Table 3).

Discussion

The best estimate model indicated that over the course of 50 to 100 years the swamp wallaby population in Muogamarra Nature Reserve would decline and possibly become extinct. However, considerable uncertainty in the model was exposed by the best and worst case scenarios, alternatively predicting a population explosion or rapid decline. This variation highlights the uncertainty of PVA predictions (McCarthy et al., 1996; Lindenmayer et al., 2000), even when information on the subject species is robust. But what this information also tells us is that persistence of the swamp wallaby in Muogamarra Nature Reserve is dependent upon variation in a number of identified parameters.

The best estimate model predicts that the decline in the swamp wallaby population would occur slowly and perhaps go unnoticed. To avoid such a decline the factors causing uncertainty in the model have been identified and should be addressed. Of the ten parameters initially considered to be variable (Table 2), the models were most sensitive to reproductive decline and survivorship after the first catastrophe (95% chance of occurring in 50 years); to reproductive decline and survivorship after a second catastrophe (50% chance of occurring in 50 years); to the mortality rates of female juveniles and adults not related to catastrophes; and finally to fecundity levels not related to catastrophes.

Contribution to urban wildlife management

Many reviews of PVA indicate that its primary benefit lies in evaluating and ranking the effectiveness of various management options, not in the actual population predictions (Bertram, 1980; Lacy and Clark, 1993; Possingham et al., 1993; Beissinger and Westphal, 1998; Lindenmayer et al., 2003). High quality data, sensitivity testing and analysis are essential in population modeling as model validation is desirable but often unachievable due to the prodigious observation period required. However, where modeling validation has been possible, improved modeling accuracy has been correlated with particular population and habitat characteristics (Lindenmayer and Lacy, 2002), such as a singular population and uniform habitat present in this study. While in themselves they may not be accurate, modeling predictions may be useful for planning that accounts for worst case scenarios (i.e. extinction). Plausible estimates for the probability of extinction can be an integral part of the planning process, such as the designating of levels of threat towards a given species and setting priorities for subsequent conservation action to ensure the persistence of the target species (Lindenmayer et al., 2003). In this regard, PVA becomes a part of a program of active adaptive management (Walters and Holling, 1990).

By assessing the impact of population parameters on the likelihood of population persistence, urban wildlife managers can then model how various management actions will impact on a population in an otherwise complex system. Resources can be invested into the optimal management options, but where one option is too costly or difficult to implement, a combination of less effective management options may achieve the desired effect. For example, the ongoing control of predators and the elimination of one out of two catastrophes (wildfires) would be the most effective management options in MNR, but due to the needs of local dog owners, the cost of policing dog presence in the reserve and the flammability of the eucalypt forest, these are probably not realistic management options. As an alternative, the modeling indicates that by establishing a reserve-wide and ongoing prescribed burning program, which decreases wildfire intensity and increases wallaby survivorship, at the same time as controlling for dog presence in the reserve after the fires (both prescribed and wild), the swamp wallaby population will not only persist, but flourish. Thus, the key to peri-urban PVA and sensitivity analysis is that urban wildlife managers can formulate effective and pragmatic strategic management options within a complex environment, whose impacts on local wildlife populations would otherwise be difficult to predict.

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