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Achieving no net loss in habitat offset of a threatened frog required high offset ratio and intensive monitoring

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ABSTRACT

The use of habitat offset to mitigate the impact of development on threatened species is becoming increasingly popular. Despite a plethora of theoretical work on the requirements of habitat offset to achieve no net loss, there are very few examples of successful habitat offset programs and monitoring regimes to detect success. We present a case study of a population of the threatened green and golden bell frog (*Litoria aurea*) which was impacted by urban development through the removal of nine ponds. Development was concurrent with habitat offset and construction of a large number of ponds which resulted in a 19-fold increase in available pond area. Through the use of mark recapture surveys, the population size was determined pre- and post-development. Despite the creation of ponds in the immediate vicinity of the development there was a decrease in the pond area and a measured decline in the population located within the area where the development occurred. However, the overall pond construction program also involved the addition of considerable habitat away from the immediate vicinity of the development which resulted in a 19-fold increase in pond area and an approximate 1.2–3.5-fold increase in population size. No net loss in population size to 95% confidence was achieved only when including all pond construction. This study demonstrated that to achieve no net loss for a habitat offset program can require extensive levels of habitat creation with intensive monitoring to detect it.

1. Introduction

Loss and alteration of habitat has seen the reduction of species at the local, national and global scale and these factors are listed as the most common cause of species decline (Butchart et al., 2010). Though the large-scale clearing of natural habitat for agriculture has recently declined in many developed countries, industrial and urban development continues to endanger many species and habitats over a wide geographical area (McKinney, 2002; Pauchard et al., 2006).

Habitat loss mitigation through the creation of new habitat has been an increasingly popular requirement for development approvals (Edgar et al., 2005; Madsen et al., 2010); wetland restoration or creation in the US alone increased from 7148 ha to 56,613 ha from 1992 to 2002 (ten Kate et al., 2004). The intention of habitat offset is to achieve 'no net loss' or ideally lead to a 'net gain' in the conservation value of an area impacted by development (Quintero and Mathur, 2011). For habitat offset concerning a single threatened species, this usually means no loss in population size or viability through the actions of a development. Successful implementation of habitat offset enables infrastructure projects to contribute to conservation efforts through mitigation programs, whilst long-term monitoring programs to evaluate success can provide much needed insight into the population dynamics of threatened species and communities (Quintero and Mathur, 2011).

The effectiveness of habitat offset has been widely debated, as the quality and extent of offset and level of monitoring and review are often insufficient to ensure that successful offset has been achieved (Maron et al., 2012; Matthews and Endress, 2008; Morris et al., 2006). The creation of habitat is made difficult by the level of uncertainty in the eventual outcome of the program. Though created habitat can resemble the composition of existing habitat, certain ecological processes can be difficult to restore, possibly reducing the compatibility for the target species or community (Moreno-Mateos et al., 2012). A time lag is also expected between the creation of habitat and habitation by the target species, as some habitat resources require later-stage succession (Moilanen et al., 2009; Vesk et al., 2008; Zedler, 1996). This can result in some developments proceeding before the offset habitat has the capacity to achieve no net loss. This time lag is pronounced in certain habitat such as woodlands and some grassland, but can be rapid in highly dynamic or transient systems, such as mudflats, salt marshes and freshwater wetlands (Morris et al., 2006).

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The uncertainty of success for the development of offset habitat has resulted in some broad recommendations for its implementation. Two of the major recommendations concern the size and location of habitat offset projects as a means of increasing the probability of creating the ecological processes required for success. A high offset ratio, where more habitat is created than lost, is recommended for species with a risk of failure (Bruggeman et al., 2005; Dunford et al., 2004; Moilanen et al., 2009). Under this circumstance, a small proportion of success within created habitat may still achieve no net loss as a large quantity of habitat is created. The second recommendation is to build offset at a close proximity to the lost habitat in an attempt to maintain the original composition, increase the probability of colonisation and to incorporate localised habitat characteristics or ecological processes (Moilanen et al., 2009). The final recommendation is to delay development so as to allow succession of offset habitat to achieve no net loss. However, the slow succession of some environments and the economic value of some developments to society mean that many developments proceed before this is achieved, and therefore management of the offset habitat is required to ensure successful mitigation (Morris et al., 2006).

The literature contains extensive theoretical justifications pertaining to the above recommendations (see Morris et al. (2006) for a summary). However, criticisms of habitat offset programs include that there is a consistent failure to monitor and report the success of offset (Edgar et al., 2005), and that success is frequently evaluated based on excessively lenient criteria (Matthews and Endress, 2008). Monitoring of habitat offset projects is required pre- and post-development to determine success, and long-term monitoring is required to evaluate sustainability of the population (Quintero and Mathur, 2011). A review of great crested newt (*Triturus cristatus*) habitat offset projects in the UK found that just 49% of projects included a post-development monitoring period. Furthermore, the average length of this monitoring period lasted 1.8 years, which would not account for any negative effects that succession may have on the population (Edgar et al., 2005).

We present a case study of a threatened species for habitat offset that was successful in achieving no net loss through the creation of large areas of habitat. This could be successfully evaluated with the use of long-term data that was collected for the target populations prior to and after a development that resulted in the loss of habitat. This case study highlights the complexity of dealing with habitat offsets for a species which is perceived to be 'straightforward' based on its biology and habitat requirements (see Section 2.1), and demonstrates that the level of effort required to successfully construct and monitor habitat offset may be drastically underestimated for most infrastructure projects.

2. Materials and methods

2.1. Study species and site

The green and golden bell frog (*Litoria aurea*) is native to the south-east coast of Australia and is listed as vulnerable by the IUCN (Hero et al., 2004). Populations have declined since the 1970s, contracting towards the coast with just 37 populations occurring in the state of New South Wales. This coastal contraction has placed the remaining populations of *L. aurea* under increased threat from urban development (White and Pyke, 2008b). *L. aurea* has been observed to rapidly inhabit ponds after creation and has the highest recorded fecundity for a native Australian frog (Hamer and Mahony, 2007). These traits make *L. aurea* a perceived ideal candidate for habitat offset as habitat can be rapidly created and inhabited.

One of the largest populations of *L. aurea* is found at Sydney Olympic Park, the site of Australia's biggest urban remediation projects (Darcovich and O'Meara, 2008). *L. aurea* was historically found throughout the park, including within a disused quarry, known as the Brickpit, which was conserved to maintain its population of *L. aurea*. Long-term monitoring has been commissioned by the Sydney Olympic Park Authority throughout the development period and has been maintained through the post-development period.

A development occurred in the Brickpit in 2000 which resulted in the loss of 9 of 26 ponds by flooding two lower levels of the quarry to create a water reservoir (Australian Museum Business Services, 1999). This equated to a loss of 3351 m^2 of pond surface area and 775 m of pond edge. As a mitigation measure, 19 ponds were constructed within the Brickpit. An additional 24 ponds were constructed throughout Sydney Olympic Park as part of the *L. aurea* management plan to conserve the population outside the Brickpit (Fig. 1). A requirement for any development in the Brickpit was that these external ponds were successfully colonised by *L. aurea* (Darcovich and O'Meara, 2008). These changes equated to the creation of 2249 m^2 of pond area in the Brickpit and 64,757 m^2 in total throughout Sydney Olympic Park (830 m and 6927 m of pond edge respectively; Table 1). These ponds were created within 2 km of the Brickpit, on top of historical locations for the species to remove the issue of proximity of offset habitat to removed habitat. Offset habitat outside the Brickpit was also created adjacent to already occupied ponds.

This study focused on two major offset areas outside of the Brickpit where *L. aurea* exhibit the highest abundance known as the Northern Water Feature and Narawang Wetland. It also includes a subset of the Brickpit ponds where abundance was highest, including most offset ponds within the Brickpit.

2.2. Monitoring

Monitoring of the population was conducted by different groups during the life of the project, resulting in variable methods and level of effort. These methods included auditory surveys, tadpole surveys, timed visual encounter surveys to determine relative abundance and mark recapture surveys. We have analysed the mark recapture data so as to determine the population size of the Brickpit and offset habitat wherever this data was available. Mark recapture involved repeated surveys of ponds where frogs were captured with a disposable plastic bag to prevent disease transmission. Frogs were scanned to detect a passive integrated transponder (PIT) tag, and newly encountered individuals were marked via subcutaneous insertion of a PIT tag in the dorsolateral region of the body and were then released at the site of capture.

Regular closed-population mark recapture surveys were conducted annually in the Brickpit from 2007 to 2011. Development of the Brickpit occurred from August 1999 to June 2000. Two closed-population mark recapture surveys were completed 9 and 6 months prior to the beginning of development within the brickpit. During the initial stages of development, frogs were removed from the development area to limit direct mortality of frogs. These frogs were relocated to ponds adjacent to the development area, and a single mark-recapture survey was conducted concurrently with this removal process. A single mark recapture survey was also conducted 10 months after completion of the development.

All surveys within the brickpit were conducted to follow the assumptions of the closed population model (Pollock et al., 1990). Consistent closed-population mark recapture surveys conform to the Pollock's robust design model which incorporates sampling at two temporal scales, known as primary and secondary sampling events (Kendall, 2001; Pollock, 1982). Primary sampling events were separated by long intervals at which migration, death and recruitment occur (open population). Within each primary sampling event, more than one secondary sampling occasion occurred over a short period during which the population can be as-

Fig. 1. Map of ponds at Sydney Olympic Park representing the ponds lost during the Brickpit development (black), existent ponds unaffected by development (dark grey) and ponds created for habitat offset program (light grey).

Table 1
Habitat offset ratios for the Brickpit development site population and the total population across Sydney Olympic Park using different descriptions of habitat:number of distinct ponds, surface area of ponds and length of pond edge.

	Pond number	Surface area	Pond edge
Brickpit	1:2.1	1:0.7	1:1.1
Total park	1:4.7	1:19.3	1:8.9

sumed as closed. By incorporating closed population estimates for abundance and open estimators for survival within the one model, the overall analysis is more robust than if these were estimated separately (Kendall, 2001). Pollock's robust design estimates:

- Apparent survival (φ) – probability an animal stays within the study area between primary surveys. Removal of an animal can occur through mortality or permanent emigration.
- Temporary emigration (γ) – probability an animal migrates away from the survey area for at least one sampling occasion and subsequently migrates back.
- Capture probability (p) – probability an individual is captured during a survey period
- Recapture probability (c) – probability a marked animal is captured during a survey period.
- Population size (N) – size of the target population.

The assumptions of Pollock's robust design are as follows:

- Capture and survival probability of each individual is independent of other individuals.
- No births, deaths or migration occur during secondary sampling occasions (closure).
- Survival probabilities are equal for all individuals in the population.

- Marks are unique and not lost or misread.
- Capture and marking do not affect survival or recapture rate.
- Marked individuals are a true representation of the population.

Robust design mark recapture surveys were also conducted for the Northern Water Feature and its surrounding ponds.

Competing models were tested by comparing the effect of keeping each parameter constant with varying parameters over time, and by determining whether capture and recapture probabilities were equal during a single sampling period. Where appropriate, capture and recapture probability was also modelled against the number of people in each survey, the length of the survey (recorded as number of nights) and the maximum temperature in the day preceding the survey from the Sydney Olympic Park Bureau of Meteorology weather station.

Model selection was based on Akaike's Information Criterion with correction for small samples (AICc) produced by program MARK. The most parsimonious model was determined as the model with the smallest AICc value. The ΔAIC value is the difference between a model and the most parsimonious model, and ΔAIC values of less than two could not be considered different enough to reject (Burnham and Anderson, 2002). To remedy this, each model was weighted according to the ΔAIC value and the parameter outputs were averaged according to the methods of Burnham and Anderson (2002). Models that failed to converge were removed from the candidate model set to prevent influence on the model averaging results.

Surveys to determine the relative abundance of *L. aurea* at each pond were undertaken consistently since 1996, and marking of individuals was introduced to these surveys in 2008. This marking data from 2009 to 2011 was used to determine population size of the third high-abundance offset habitat area within Sydney Olympic Park, the Narawang Wetland, using robust design mark recapture analysis. However, the assumption of closure was unlikely to be met for the secondary sampling periods as they occurred 1 month apart which was likely causing slight positive bias for population size in the Narawang Wetland (Kendall, 1999).

Population size was not estimated for existing habitat outside the Brickpit, mostly due to the low density of *L. aurea* within these ponds. Therefore, evaluation of no-net-loss is based on the assumption that the development within the Brickpit and the creation of offset habitat did not have a negative impact on the population of *L. aurea* within existing ponds outside the Brickpit.

For the purpose of this study we only used population size as it is a measure of 'no net loss.' In total, five mark recapture models were produced: three for the brickpit (1999–2000, 2000–2001 and 2007–2011) and one each for the Northern Water Feature and Narawang Wetland (2009–2011).

To determine whether no net loss was achieved, the population size of the Brickpit based on the 1999–2000 pre-development surveys was compared to the combined population size of the Brickpit, Northern Water Feature and Narawang Wetland in 2010. The upper 95% confidence interval of the population pre-development was used as the threshold of success whereby the lower 95% confidence interval of the post-development population size estimate had to reach this level so as to remove the issue of uncertainty.

3. Results

Effort in the mark recapture surveys within the Brickpit development site increased over the 13 years of study (Table 2). Therefore, the most parsimonious models for the five surveys were different (Table 3). The Brickpit mark recapture indicated a population decline from 276–551 in 1998 to 131–156 in 2010 (95% confidence intervals; Fig. 2), representing an approximate 1.5–2.8-fold loss in population size. However, the level of temporal variability in population size is high within the Brickpit, as is demonstrated by an approximate 100% increase between 2010 and 2011. The observed decline may therefore be an artefact of limited sampling as the range of population sizes were unknown prior to development within the Brickpit.

Areas of highest abundance within the offset habitat supported large populations according to the mark recapture analysis. The Northern Water Feature and the Narawang Wetland were found to have populations that ranged between 182–208 and 252–492 (95% confidence intervals) respectively.

Comparison of population size of the Brickpit prior to development and the total population size after development indicates that the offset program has been successful in achieving no net loss with an approximate 1.2–3.5-fold increase in population size. The lower confidence interval post-development exceeds the upper confidence interval pre-development and the threshold for success was therefore met (Fig. 3).

4. Discussion

The habitat offset program at Sydney Olympic Park was extensive in its attempt to achieve no net loss, with a 19-fold increase in pond area and 8.9-fold increase in pond edge. Despite these large increases in available habitat, there was not an equivalent increase in population size with an approximately 1.2–3.5 fold increase in population size. Despite some offset attempts within the Brickpit development site, an overall loss of habitat resulted in a decline in population size within the Brickpit.

The disparity between the amount of created habitat at Sydney Olympic Park and the increase in population size could indicate three different processes: *L. aurea* (1) was still in the process of colonising available habitat in 2010, (2) inhabited the offset habitat at a lower density or (3) could not colonise all available areas. *L. aurea* has the highest recorded fecundity of any Australian frog and has been noted for its ability to rapidly colonise ponds after their creation (Goldingay and Newell, 2005; Hamer and Mahony, 2007). It is therefore unlikely that the colonisation process extended the entire decade of this study making the first explanation unlikely. Created wetlands have been shown to experience lower vegetation structure, biodiversity, invertebrate assemblages and productivity than natural wetlands (Brown et al., 1997; Moreno-Mateos et al., 2012; Stanczak and Keiper, 2004). These factors could result in a lower density for *L. aurea* or if productivity is too low could result in unviable populations over the long-term. Design of ponds for habitat offset was based on the known habitat features for this species (Darcovich and O'Meara, 2008; Pyke and White, 1996), although it is possible that an obscure habitat feature has not been

Table 2
Summary of captures for each mark recapture survey. Captures are the number of frogs captured during a survey, with the number of recaptures indicating how many of these animals were previously captured in the survey.

	1998	1999	2000	2001	2007	2008	2009	2010	2011
Captures	156	96	89	69	79	122	193	206	301
Individuals	135	87	79	61	69	103	135	142	221
Recaptures	21	9	11	8	10	19	71	88	103

Table 3
The most parsimonious models from each candidate model set for mark recapture survey. Comparison of models for parsimony used Akaike's Information Criterion (AICc) and models with ΔAICc < 2 are included within the table. Model parameters include: apparent survival (φ), temporary emigration (γ'' and γ'), capture probability (p) and population size (N) and included the effects of time (t), the number nights a survey lasted (surveyNights), the mean number of people in the survey over the number of survey nights (meanSurveyors). Some parameters did not vary (.) and some were constrained as indicated by the "=" symbol.

Model	ΔAICc
Narawang	
φ (t) $p(t) = c\gamma'' = 0\gamma' = 0N(.)$	0
Northern Water Feature	
$\varphi(.)$ $p(t) = c\gamma'' = 0\gamma' = 0N(t)$	0
Brickpit 1999–2000	
$\varphi(.)$ p(surveyNights) $= c\gamma'' = 0\gamma' = 0N(t)$	0
$\varphi(.)$ p(surveyNights) $= c\gamma'' = 0\gamma' = 0N(.)$	1.7
Brickpit 2000–2001	
$\varphi(.)$ $p(t) = c\gamma'' = 0\gamma'' = 0\gamma' = 0N(t)$	0
Brickpit 2007–2011	
$\varphi(t)$ p(meanSurveyors) $= c\gamma''(.)\ \gamma'(.)\ N(t)$	0
$\varphi(t)$ $p(t) = c\gamma''(.)\ \gamma'(.)\ N(t)$	1.76

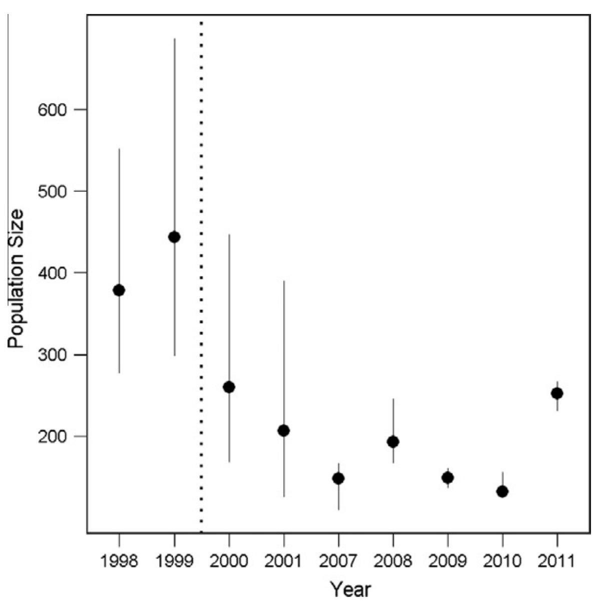

Fig. 2. Size of the Brickpit population over time. Dotted line indicates beginning of development pressures between August 1999 and June 2000. Note that frogs were moved from development site in 2000–2001. Error bars indicate 95% confidence intervals.

identified as relocations for this species are often unsuccessful (Stockwell et al., 2008; White and Pyke, 2008a). This would result in uninhabited areas where this unknown habitat feature was absent. This last point is possible due to the unknown role of habitat in controlling pathogens; particularly the chytrid fungus which is known to affect this species and population (Penman et al., 2008).

Determining the threshold for success is a shortfall for many offset programs (Matthews and Endress, 2008), but is simplified for single species projects which can use estimates of population size. The threshold for success was reached for this offset project when offset outside of the Brickpit was included, and this threshold incorporated uncertainty in the population size estimate. The pre-development population size estimate ranged from 276 to 551 (95% confidence intervals). In order to achieve no net loss, it

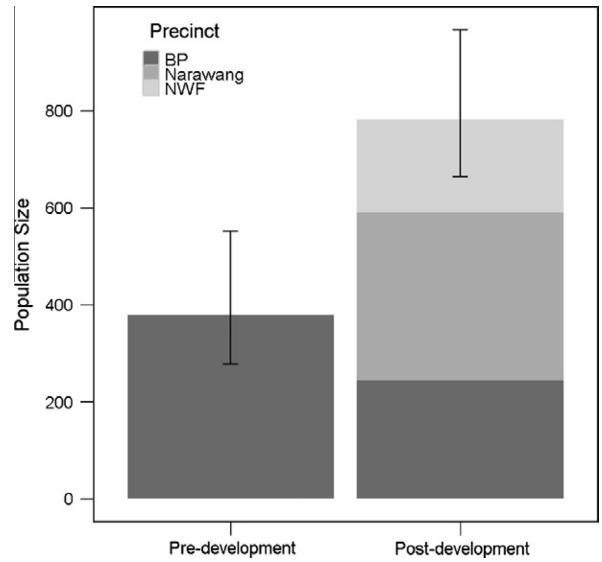

Fig. 3. Population size of the Brickpit development site before development compared to the combined populations of the Brickpit and two offset areas (Northern Water Feature and Narawang Wetland) 10 years after development. Despite a population decline within the Brickpit, the overall habitat offset program produced no net loss in the population size. Error bars indicate 95% confidence intervals.

was also necessary for the population to have increased rather than remaining the same after development occurred, as the large amount of uncertainty increased the upper confidence limit which was used as the threshold for success. Ideally, a large amount of effort should be invested in the initial phase of habitat offset projects when the pre-development population size is assessed, so that the threshold for success can be determined more precisely. Monitoring regimes should be explicit in the minimum level of uncertainty and flexible so as to allow increased monitoring until this level is reached.

Determining the success threshold is complicated by temporal variability in population size. Determining population variability requires long-term data, as rare impacts on population size such as 1-in-50 year floods, cannot be detected without repeated surveys (Pimm and Redfearn, 1988). It is common practice for the development process to be rushed, but attempts should be made to incorporate as many pre-development population size estimates as possible so as to determine some level of population variability. Temporal variability was evident with this population, as the mark recapture results indicated a highly variable population size with an approximately 100% increase between 2010 and 2011. Failure to incorporate long-term data could result in a population size estimate that is extreme for that population and not a representation of the norm.

Use of a single metric for success is likely to be an over-simplification of the viability of a population subject to habitat loss and offset (Traill et al., 2007). The habitat offset program at Sydney Olympic Park resulted in an increased population size over a larger distribution. This could have both positive and negative impacts for population viability, particularly for populations acting as multiple metapopulations. Creation of offset habitat that is further from the existing population than the lost habitat would likely reduce the level of migration. If migration is substantially reduced, the capacity for nearby populations to be recolonised after extinction is diminished. Alternately, an increased spread of the population will mean localised events that negatively impact a metapopulation will have less impact on the population as a whole (Hanski, 1999). The impact on habitat offset increasing the distribution of a population on population viability requires further study.

This study has focused on a single species for determining success in a habitat offset program and therefore does not address other functions of the habitat. The lower density of *L. aurea* within the offset area suggests these wetland systems were acting differently to the original habitat. Studies have demonstrated that constructed wetlands can be rapidly colonised by certain species (e.g. Stanczak and Keiper, 2004), but other important indices of ecological function require long periods of time to recover (Craft et al., 2003; Hossler et al., 2011). Further research is required into the benefits and losses for these mitigation techniques which may aid conservation of single species but reduce ecosystem function and change species composition.

Habitat offset is increasingly popular for the mitigation of species prone to decline from human development. However, the implementation of offset and monitoring of results are often insufficient for achieving and detecting no net loss for a population. The offset program at Sydney Olympic Park was extensive in both aspects as the level of offset required was much more than is usually recommended (Briggs et al., 2009; Moilanen et al., 2009), and monitoring aimed to produce precise and robust estimates over a long period post-development.

5. Conclusions

Habitat offset projects have the capacity to contribute to conservation efforts when successfully implemented if they achieve net gain (Quintero and Mathur, 2011). Additionally, the use of long-term monitoring data to determine success can increase knowledge of population dynamics, for both natural populations and those exposed to human-induced pressures. However, habitat offset aimed at achieving and detecting no net loss can only be successful where the offset ratio is large, monitoring is long-term, robust and precise and funding is available to substantially increase the amount of habitat if monitoring indicates that this is necessary. This is the major short-fall of most offset programs, and this paper illustrates that even for species that are perceivably ideal for habitat offset, a large amount of effort is required for successful outcomes.

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