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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-