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Conservation Issues

# Wetland Restoration for the Threatened Green and Golden Bell Frog (*Litoria aurea*): Development of a Breeding Habitat Designed to Passively Manage Chytrid-Induced Amphibian Disease and Exotic Fish

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## ABSTRACT

Amphibians are under threat from many drivers resulting in declining populations. Restoration and creation of habitat is a method used to reverse amphibian declines. The green and golden bell frog (*Litoria aurea*) is distributed in southeastern Australia, and is threatened by the fungal pathogen *Batrachochytrium dendobatidis* (chytrid), an introduced fish (the plague minnow, *Gambusia holbrooki*), and habitat loss. There have been numerous wetland restoration attempts to combat population declines in this species, which have been largely unsuccessful in producing persisting populations. Here we present a robust model for the creation of breeding habitat for the *L. aurea* population on Kooragang Island, New South Wales, which is based off thorough review of the literature and past pilot studies and experiments. We describe in detail the habitat, land use history, and wetland habitat design formulation and construction so that the context of the habitat creation is understood and so construction can be repeatable and the design can be further refined. The habitat features passive controls for chytrid and *G. holbrooki*, and contains the most optimum breeding habitat for *L. aurea* based upon current knowledge. This is the first attempt in our knowledge to create wetlands in an open system that have the potential to passively manage chytrid.

*Index terms:* adaptive management; amphibians; *Batachochytrium dendrobatidis*; *Gambusia holbrooki*; chytrid; restoration ecology

## INTRODUCTION

Amphibians worldwide are under considerable threat from multiple drivers (Wyman 1990). One major driver that stands out is the fungal pathogen *Batrachochytrium dendrobatidis* (hereafter chytrid), which causes the lethal disease chytridiomycosis (Skerratt et al. 2007). This disease has caused declines in over 200 amphibian species worldwide (Wake and Vredenburg 2008) and the estimated extinction of ~90 species (Scheele et al. 2019b). In addition to this driver, early estimates indicate that ~200 species of amphibians are impacted by habitat destruction (Stuart et al. 2004). There is evidence of additional decline in some species, caused by other drivers such as climate change, invasive species, and habitat degradation (Kats and Ferrer 2003; Hamer and McDonnell 2008; McMenamin et al. 2008). Given the widespread threats to amphibians, there is an urgent need for conservation measures to stabilize populations.

One of the most commonly implemented and most successful methods for reversing amphibian declines is the use of restoration ecology principles to reverse historical habitat degradation and create new habitats (Oertli et al. 2009). Here we define restoration ecology as the study of methods for recovering degraded ecosystems, reestablishing components of ecosystems, and creating new habitat for threatened species and communities (Young et al. 2005). Under this paradigm, we consider the following as restoration attempts: rehabilitation, habitat creation, habitat enhancement, and remediation (Aronson et al. 1993; IUCN 2012). Actions that fall under the restoration ecology paradigm include building abiotic structures for habitat and removing or adding species to an ecosystem. An example of the first is the construction of predator-proof fences as barriers that prevent predators from entering a site, thereby relieving a target species from predation pressure (Hayward and Kerley 2009). The second concept relates more to eradication, translocation, or revegetation (Sinclair and Krebs 2002; Munro et al. 2007; Parker 2008). The main criteria that determine success in restoration ecology projects is a clear idea of what the desired habitat is, along with measurable success parameters, and a feasible plan of action that is backed by robust scientific understanding (Hobbs and Norton 1996). In regard to amphibians, the major objective is often to make more breeding habitat available, which is usually achieved with wetland restoration or creation, and this may be supplemented with reintroduction or assisted colonization (Mitsch and Wilson 1996; Brown et al. 2012; Harding et al. 2016), but there is also a clear need for connectivity in terrestrial habitats (Gibbons 2003).

There are apparent worldwide trends associated with abiotic and biotic habitat features in successful amphibian wetland restoration projects. The most commonly cited successful abiotic design is that of a "wetland mosaic" (Petranka et al. 2007; Rannap et al. 2009; Hamer et al. 2012). A habitat mosaic is defined as "an area composed of multiple habitat types" and similar to this definition, a "wetland mosaic" refers to an area that contains multiple wetlands of varying features such as size, shape, and hydro-period (Wiens et al. 1993; Smith et al. 2007). Landscape placement is another key component of success, as restorations are more likely to succeed if they occur in close proximity to existing occupied wetlands (Lehtinen and Galatowitsch 2001).

From the biotic perspective, one of the most common causes of failure to produce a persisting amphibian population is the presence of invasive organisms, which includes predators, competitors, parasites, and diseases. Colonization by invasive predatory fish species has been well considered in the past (Beebee 1997; Baker and Halliday 1999; Julian et al. 2006). An important biotic concern in habitat restoration for amphibians is the control of wildlife diseases. Unfortunately, the impact of chytrid fungus has rendered restoration projects as failures (Stockwell et al. 2008) despite the observation that the physical habitat features are suitable for the target species. Dealing with disease in amphibian restoration ecology requires creating habitats that facilitate coexistence of target species and disease (Scheele et al. 2019a). This requires creating habitat that limits the disease while still remaining within parameter thresholds that are conducive to successful survival and reproduction of a target species (Scheele et al. 2019a).

The green and golden bell frog (*Litoria aurea*) is a threatened species found in southeastern Australia that has undergone severe declines facilitated by multiple threats (Mahony et al. 2013). Historically it was one of the most commonly encountered frogs in its distribution but has dramatically declined since the 1960s (White and Pyke 1996). This severe decline resulted in it being listed as "endangered" in New South Wales (Lunney et al. 1995) and "vulnerable" nationally (EPBC Act 1999). Three main drivers have been identified: habitat destruction, disease, and exotic fish (Mahony 1999). Numerous studies have investigated the roles of these threats, and it is recognized that they can act in a cumulative fashion to threaten populations (Mahony et al. 2013). *L. aurea* is highly susceptible to the amphibian disease chytridiomycosis and this is considered to be the primary driver of the decline (Stockwell et al. 2016). The introduced invasive fish the plague minnow (*Gambusia holbrooki*) exacerbates the impact of chytrid by depredating *L. aurea* larvae and eggs (Klop-Toker et al. 2017). While population strongholds remain in far eastern Victoria (Gillespie 1996), there are only 31 remaining in NSW with a 50% decrease in the number of populations in 12 y (White and Pyke 2008b).

There have been 16 wetland restoration and reintroduction projects to combat *L. aurea* decline, however 43% have not been successful in producing persisting populations (Mahony 1999; Pyke et al. 2008; see Appendix 1 for review). There are some key causes for the lack of success that can be identified from these past recovery attempts (see Appendix 1 for summary). Here we define a "population recovery attempt" as any action that was used to recover or enhance a population, including any kind of restoration, reintroduction, or other conservation translocations (IUCN and SSC 2013). All five population recovery attempts occurring in an area that did not have an existing *L. aurea* population were unsuccessful in creating persisting populations. There are two confirmed cases of *L. aurea* population recovery attempts failing to produce self-sustaining populations due to chytrid (Stockwell et al. 2008; White and Pyke 2008a), however 8 of 16 were not monitored for chytrid. Five of these populations are likely extirpated.

While chytrid is considered to be the primary cause of *L. aurea* population loss across its distribution (Mahony et al. 2013), there is an unusual situation that occurs near coastal habitats (Stockwell et al. 2015a, 2015b), where the frog can survive despite infection with chytrid and where *G. holbrooki* can tip the decline to local extinction if it is widespread in these coastal systems. There are four examples where *G. holbrooki* invaded created/restored wetlands (Pyke et al. 2008; White and Pyke 2008a; O'Meara and Darcovich 2015). Of these, two failed to produce population persistence (Pyke et al. 2008; White and Pyke 2008a). The two that were successful in creating persisting *L. aurea* populations despite *G. holbrooki* invasion had numerous freshwater wetlands (>50) within the landscape, where some of these wetlands, on a temporal basis during a breeding season, were not colonized by *G. holbrooki*. The attempts that failed had relatively smaller numbers of wetlands within the landscape (<10) and a higher proportion were colonized by *G. holbrooki*.

With many reintroduction and restoration attempts not succeeding, it is imperative that the ongoing declines must be met with a scientifically robust restoration ecology model to compensate for large-scale loss of *L. aurea* populations due to chytrid (Daly et al. 2008; Goldingay 2008; Pyke et al. 2008; White and Pyke 2008b; Mahony et al. 2013). The most widely used framework for restoration ecology procedures was developed by Hobbs and Norton (1996), which identifies seven steps: (1) identify processes leading to decline, (2) develop methods to reverse declines, (3) determine realistic goals for reestablishment, (4) develop easily observable measures of success, (5) develop practical techniques for implementing restoration goals, (6) document and communicate these techniques, and (7) monitor key system variables to assess progress and adjust procedures if necessary (adaptive management).

The aim of this case study, following this framework, is to provide comprehensive details of a wetland creation project on Kooragang Island (KI), New South Wales, that is based on past learnings to successfully produce a persisting *L. aurea* population. We aim to address parts 2–5 of the Hobbs and Norton (1996) restoration ecology framework and apply this to conservation of *L. aurea*; the answer to step (1) was addressed by Stockwell et al. (2016) and Mahony et al. (2013). More extensive descriptions of the population ecology, demography, habitat use, and population genetics in the restored habitat will be provided in future publications.

This case study is split into subsections that describe and review climate, habitat, land use history, design formulation, construction, monitoring, and adaptive management so that the context of habitat creation on KI is understood, and so that the construction of the habitat is repeatable and can be further refined.

## Study Site Description – Kooragang Island

**Location and Physical Description:** Kooragang Island is situated in the mouth of the Hunter River ~5 km northwest from the city of Newcastle (GPS: –32.9283, 151.7814; Brereton and Taylor-Wood 2010), on the eastern Australian seaboard

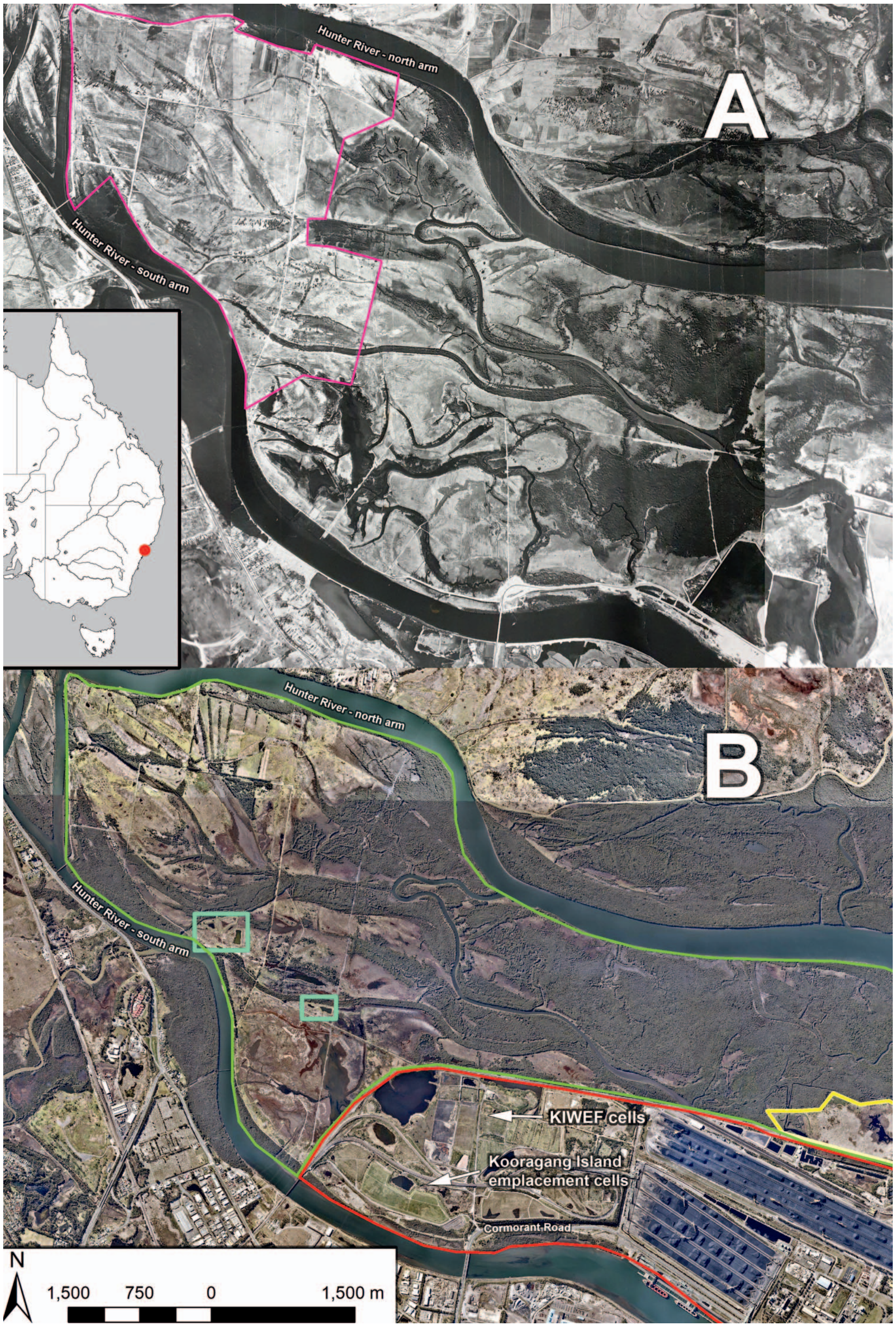


**Figure 1.**—Historic and current aerial photographs of Kooragang Island. (A) Historical photograph of Kooragang Island, year of image: 1965. Two images stitched together, photo numbers: 5,190 and 5,192 NSW (1464). Purple indicates area historically used for agriculture. (B) Aerial photograph from Nearmap (2019 15/08/2019). Blue-green rectangles indicate location of the created wetlands. Green indicates land managed by the NSW NPWS. Red indicates land zoned for industry. Yellow indicates the location of a RAMSAR listed wetland.

(Figure 1). It is a estuarine deltaic island that is ~11 km long and 3 km wide at its greatest width, and has a surface area of ~2560 ha (Streever 1998). Historically, it consisted of 10 islands in a deltaic system; the largest islands were named Ash Island, Spit Island, Dempsey Island, Moscheto Island, and Walsh Island, and all were combined by reclamation works in the mid-1900s (Johnston 1992). The islands were separated by narrow intertidal river channels (Williams et al. 2000). These islands were formed by alluvial deposits consisting mostly of clay with an underlying layer of medium- to coarse-grained silty sand, above bedrock of shales, mudstone, sandstone, coal seams, and tuff (Johnston 1992; McCotter 1996).

The islands have a low relief, with the average height above sea level being <1 m (Howe 2008). The highest and lowest astronomical tides for ocean and river entrance for the nearby Port Stephens area is 2.08 and –0.3 m above and below the Australian Height Datum (AHD), respectively (Maddox 2017). However, the highest tide that has been recorded within the Hunter Estuary where KI resides is 1.56 AHD, which was recorded on 9 June 2007 after heavy rainfall that caused flooding coincided with high tides (Rogers et al. 2013). Such flooding events triggered by heavy rainfall can cause an increase in salinity throughout the landscape and can also increase the height of the groundwater table (Rogers et al. 2013). During the calendar year there are routinely two occasions when king tides (i.e., >1.2 m) occur, and at these times large parts of the western half of the island are inundated by tidal waters.

**Climate:** KI experiences a stable coastal climate with mean daily temperatures for mid-summer (January) and mid-winter (July) of 25.6 °C and 16.8 °C, respectively, and mean daily minimum temperatures for the same months 19.2 °C and 8.5 °C, respectively (BOM 2018). The mean average annual rainfall is 1122 mm, where the wettest month is June, which has a mean rainfall of 125.2 mm (10th percentile = 30.4 mm, 90th percentile = 243.4 mm), and the driest month is September, which has a mean rainfall of 60.6 mm (10th percentile = 15.4 mm, 90th percentile = 129.9 mm) (Johnston 1992; BOM 2018). However, long-term averages hide a considerable level of stochasticity in annual and seasonal rainfall patterns. The region can undergo periods of drought where most sources of freshwater completely dry (Hamer 1998). The Hunter region where KI occurs has experienced drought in 9 out of the last 40 y (BOM 2018). In contrast, wet periods occur at least once per year when >140 mm of rain can fall within 2 d, which often results in local flooding (Brereton and Taylor-Wood 2010).

**Existing Habitat and Historical Alterations:** There have been significant habitat alterations due to changes in land use on KI over the past 200 y (Streever 1998; Symons and Russell 2005). Prior to European settlement, the island was likely composed of a mosaic of floodplain woodland and littoral rainforest (Dames and Moore 1978). However, clearance of this vegetation began in the early 1800s for timber and the conversion of land for cattle pastureland (Perry 1963). By 1821, most of the ash trees (*Elaeocarpus obovatus*) and other tree species were felled and harvested. Reclamation for agricultural activities began in the 1880s, and in the 1950s port infrastructure development resulted in all five islands being joined together to form what is now KI (Hamer 1998).

The earlier reclamation involved blocking of tidal creeks, draining of upland areas by channels, and cutting of natural levees along the creeks to create wet pastures (Howe 2008). This prevented the inflow of saltwater and presumably assisted in the drainage of the land. During this latter reclamation process, sediments were imported (usually dredged sands from the river derived from port deepening) to provide dry land for industrial infrastructure, particularly in the eastern portions of the island. Rail line construction began in 1966 to support a large port facility for coal loading (Williams et al. 1995). The rail line enclosed 829 ha of the southeastern end of the island, and was built on a 3 m high embankment above the floodplain using slag, which is a by-product of steel production. This rail line was placed across several of the larger estuarine creeks on the eastern end of the island, effectively blocking their flow. It also formed a large enclosed area that was gradually reclaimed over several decades for industrial purposes. The southwestern portion of the enclosed area was used as a certified heavy industrial waste site and was subdivided into 34 approximately equal "cells" that were bordered by a bund wall of slag. These were known as the KI waste emplacement facility (KIWEF) cells. These cells were gradually filled with waste products from the Newcastle steel works and other heavy industries. Sand and clay sediments dredged from the Hunter River were used to reclaim other areas of the island for light industrial use.

Further habitat alterations occurred after 1955 due to flood mitigation works on the Hunter River that were not directly on KI, but which altered tidal flow rates and heights. These works included installations of floodgates in Hexham Swamp, which is a large floodplain swamp area on the other side of the Hunter River, and further creek blockages on KI (NSW Department of Public Works 1963). This altered the hydrology of the island and gradually altered the boundaries of existing vegetation communities (MacDonald 2001). Once the blocked tidal creeks were reopened tidal inflow returned and mangroves have begun to recolonize these areas. There is a new "saline" influence, and at high tides saltwater spreads across salt marshes and saline meadows.

There have been several ownerships and rezoning of KI throughout history resulting in the alteration of ecological features. The traditional owners of this land were likely the Awabakal and the Worimi Aboriginal People, and there is evidence of their presence during the 1800s (Streever 1998). The first European ownership land title on KI dates back to 1827 when Alexander Scott was granted 1036 ha of land on Ash Island (the western section of what is now KI), where he settled by 1831. In 1866, Scott sold the land to various families, which formed the basis for a small agricultural community to exist. Much of the land was converted to agricultural fields for cattle (*Bos taurus*) and there is evidence of orange tree (*Citrus* × *sinensis*) plantations (Streever 1998). By 1893 the population of KI was around 450, most of which were farmers (Streever 1998). This small community continued until 1955 when a large flood devastated the island and the community. All families were forced to evacuate and most buildings were destroyed. After this incident, the state government took control of the land and leased it for grazing and rezoned portions of it as industrial land. By the 1960s, large-scale industrial developments had taken place, and by the 1970s ~27% of KI, mostly in the eastern section, had been reclaimed for industry (Symons and Russell 2005). Currently, KI is accessible by four road bridges.

The industrial rezoning of the island created considerable tension between the development of KI and the ecological values of the Hunter River and estuary. In 1983 a large section of the northern half of the island and the northern arm of the Hunter River and its embayments were declared a RAMSAR site since the habitats had long been recognized for their importance to migratory wader birds (Brereton and Taylor-Wood 2010). To address the tension between industrial development and the environmental values, the NSW government strategically focused port development on the east of the island, while the western portion of the island was dedicated to natural land. Private land ownership on the western portion of KI was resumed, and a large habitat restoration project commenced (KI Wetland Rehabilitation Project; Copeland 1997). This project included replanting of native trees, the opening of historically blocked or restricted tidal creeks, and wetland restoration. Not all human infrastructure was removed and several roads and power line easements remain and restrict the movement of surface water due to the low relief of the island. In 2007 the western portion of the island was gazetted to be included in the Hunter Wetlands National Park and management was transferred to the NSW National Parks and Wildlife Service (NPWS).

The vegetation communities that currently exist on KI are composed of five broad categories, each identified by characteristic dominant species (MacDonald 2001), and these occur in the landscape in a sequence from the tidal zone to the upper terrestrial zone. The river and tidal creeks are lined by mangrove forests, which give way to salt marsh, tidal swamp forests, wetlands, pastureland, and revegetated gum forests. The mangrove swamps are dominated by grey mangrove (*Avicennia marina*) with an understory of river mangrove (*Aegiceras corniculatum*) and occur at 0–0.7 m AHD (Buckney 1987). The salt marshes occur at a higher elevation than mangroves, usually around 0.1–1.1 m AHD, and are characterized mainly by *Sacrocornia quinqueflora*, *Sporobolus virginicus*, and *Juncus krausii* (Brereton and Taylor-Wood 2010). The tidal freshwater swamp forests are higher still, and are dominated by a canopy of *Causuarina glauca* (Brereton and Taylor-Wood 2010). The transition zone between the salt marsh and the freshwater swamp forest is well defined by plants that delineate these communities, however there are times, such as after the summer high tides, that these areas will be inundated with saline water, and at other times, such as following heavy rainfall, that they are inundated by freshwater. This balance is dynamic and relies on the interplay of these interactions. There are large shallow depressions in the upland areas (>1.1 m AHD) that are separated from the tidal creeks by natural levees. After rainfall wetlands form in these depressions, and those that occur at a low AHD can have a range of different salinity levels depending on tidal influence and rain. They are typically positioned in areas of higher AHD values compared to salt marsh. However, on KI they also occur in pastureland that has been bordered by roadways that prevent free surface water flow after rain, and therefore they can occur on AHD levels below 1.1 m. The dominant vegetation within wetlands depends on the salinity; the more brackish wetlands contain *Phragmites australis* and to a lesser extent *Bolboschoenus caldwelli*, while the freshwater wetlands primarily contain *Typha orientalis* and to a lesser extent *Bolboschoenus fluviatalis* and *Persicaria decipiens* (Winning 2006). At the highest AHD points that do not have tidal influence there are pasturelands from past agricultural use, and these are composed mainly of pasture grasses dominated by thick matts of kikuyu (*Pennisetum clandestinum*; Howe 2008). The most recent vegetation community to occur on the island is planted forests on the western end of the island, composed mainly of *Eucalyptus tereticornis* and *Eucalyptus punctata*, which have been established by the KI Rehabilitation Project (Howe 2008).

With the variety of swamp type vegetation communities that have remained on KI throughout recorded history, it is likely that the existence of a *Litoria aurea* population predates European settlement of Australia. *L. aurea* were first reported on KI in 1975 (Gosper 1975), and the first targeted surveys were conducted from 1996 to 2001 as part of a postgraduate research program (Hamer 1998). This led to the first population size estimate for the island of 1995 ± 315 for the year 2000 and 905 ± 143 in the year 2001 (Hamer and Mahony 2007). Yearly long-term monitoring has been conducted since 2011 and apparent fluctuations in the occupancy and population size of *L. aurea* on the island have been documented (Clulow et al. 2012, 2013, 2014; Campbell et al. 2015; McHenry et al. 2016).

**The Compensatory Habitat Proposal**

Under sustainable development and environmental protection legislation in Australia all developments are required to assess the impacts on native species and ecological communities, and where impacts are identified, to apply a response that considers in order: avoid → mitigate → compensate (EPBC Act 1999). As a consequence of industrial port development on KI, numerous projects have conducted ecological impact assessments, and have found the development directly or indirectly affecting the viability of a local populations of *L. aurea*. Usually, this is because there was an identified impact on wetlands that are occupied by adults, which are used for breeding, and/or terrestrial habitats that are used for foraging, shelter, and movement. For several of the spatially large infrastructure projects, it has not been possible to avoid or mitigate for an impact on site, and thus the approach has been to compensate. Under Australian environmental law, compensation may involve several possible actions including purchase of land that supports a viable population or rehabilitation of land to support a viable population. This is a controversial approach since it assumes “like for like” or a “no net loss” (Brooks et al. 2006; Gibbons and Lindenmayer 2007), and may be associated with a multiplier (e.g., 2:1, 5:1) where restoration is required. Another possibility is that the proponent can pay for research that is considered valuable in addressing a pertinent issue for the threatened species or ecological community.

The compensatory habitat that is the subject of this article was created in association with development works within the eastern end of KI. Although located on disturbed former industrial land, environmental assessments identified that *L. aurea* had since established in areas within the development footprint. A condition of regulatory approval required the development proponent (the proponent) to create and implement a compensatory habitat program (CHP) to compensate for the disturbance of *L. aurea* habitat.

The proponents’ CHP included the following primary elements:

1. Completion of a research and monitoring program, including construction of trial habitat.
2. Construction of compensatory habitat at a ratio of 2:1 to the area of habitat disturbed.

3. Monitoring and adaptive management of the compensatory habitat.

The proponent engaged the services of the University of Newcastle Conservation Science Research Group to assist with development and implementation of the CHP.

**Design Formulation and Site Selection of the Compensatory Habitat:** A rather unusual, but not unique, situation arose when selecting a location of the compensatory habitat. The government departments responsible for approval of the environmental compensation, based on scientific advice, considered that restoration of habitats on government owned land in the nearby KI Rehabilitation Project was a means to promote rehabilitation and positive outcome for the *L. aurea* population. Indeed the synoptic plan for the management of the *L. aurea* population on KI recognized that the frog population extended across the island and research had shown that frogs moved widely among wetlands (DECC NSW 2007), often across rails and roads and through large industrial infrastructure (Hamer et al. 2008). Thus, the CHP was considered a win–win situation, with restoration occurring on national parks land supporting the threatened frog population affected by development on the adjacent industrial lands.

This situation provided appropriate ecological features for the potential location of the compensatory habitat that cannot be underestimated. Firstly, the substrate and climate at the compensatory habitat site are part of the same landscape where the frog occurs. Secondly, *L. aurea* is known to disperse widely among wetlands (Hamer et al. 2008), and the potential compensatory habitat site was to occur within the movement distance of individuals, as the majority of KI is inhabited by this frog. The sites chosen for wetland creation were dominated by pasture plants (mostly kikuyu *Pennisetum clandestinum*) and had historically been drained. Based on botanical and hydrological studies of KI (MacDonald 2001; Winning and Saintilan 2009; Howe et al. 2010) the land chosen for restoration was likely to have been salt marsh prior to the reclamation of the land for pasture in the early 1900s. It is difficult to rule in or rule out the importance of landscape attributes (soils, climate, and hydrology) in other cases where habitat restoration and creation for *L. aurea* has not been successful. It was our observation that where *L. aurea* were successfully retained along with large developments, such as at the Sydney Olympic Parklands (Bower et al. 2013; Pickett et al. 2013), habitats were created nearby to previously occupied habitats, often in the same substrates.

This meant that other important features of *L. aurea* habitat could be the focus of the project, including wetland hydrology, the management of predatory fish and chytrid, and wetland vegetation composition. Wetland hydrology and its linkage to the occurrence of salt were investigated in the research phase of the program (Stockwell et al. 2015c; Valdez et al. 2015; Klop-Toker et al. 2016) to test the hypothesis that *L. aurea* populations persisted in the near coastal ecosystem (see Mahony et al. 2013) because salinity in the environment provided some form of protection against chytrid (Stockwell et al. 2015a, 2015b).

Linkage between wetland hydrology and salinity occurs in two spatial dimensions. There is a saline influence in wetlands on saline substrates and on lands adjacent to saline influences, such as adjacent to tidal creeks, salt marshes, and mud flats. At times of high rainfall, surface waters are fresh and spread out across salt marshes and flats, and for a period of time these become "freshwater habitats" with a saline influence. Daily saltwater intrusion occurs with tidal highs, and these are exacerbated at certain times with maximum high tides ("King tides," $\sim$2.1 m AHD). The second dimension is the groundwater that also varies on the continuum from fresh to saline, and depends on infiltration to the aquifer of freshwater from rainfall and the influence of saline inflow from the river. Groundwater influence occurs in wetlands that intersect with the shallow aquifer that is $<$1 m below the surface on KI, and in large low-lying areas where the aquifer is exposed on the surface. The interaction of tides and rainfall on an estuarine landscape with low relief results in a situation where the occurrence of freshwater wetlands and other related ecosystems such as salt marsh and saline meadows are dynamic (Howe et al. 2010). Nonetheless, the island can be divided into the major landscape zones described above (MacDonald 2001). In accordance with laboratory studies and field observations that *L. aurea* survival was higher in saline water, and that chytrid survival was reduced in saline water (Stockwell et al. 2012; Clulow et al. 2017), a concept plan was developed for the compensatory habitat to include some wetlands that have a saline influence due to intersection with the groundwater table. This plan also included a combination of wetlands with differing hydrologies in the landscape, permanent and ephemeral, and by having these wetlands in relatively close proximity (i.e., $\sim$50 m apart and well within the dispersal distance of *L. aurea*) so that frogs would be exposed to some levels of salinity as they moved among local wetlands. This structurally equates to the concept of a "wetland mosaic" (see Hamer et al. 2002a; Hamer and Mahony 2010; Mahony et al. 2013).

Prevention of the occupancy of wetlands by predatory fish is also related to wetland hydrology. In an estuarine ecosystem with low relief, surface flooding is common and most wetlands become connected at times of high rainfall, which can result in dispersal of invasive fish such as *Gambusia holbrooki* (Chapman and Kramer 1991). When rainfall is low, ephemeral wetlands dry, and any fish occupying them perish. However, the fish persist in permanent wetlands and can rapidly recolonize ephemeral wetlands when rainfall occurs. The outcome is a spatial and temporal pattern of wetland occupancy of fish that is related to rainfall and topography. It is evident that *G. holbrooki* has occurred for many years on KI, and we have detected them in over 80% of the wetlands (Hamer et al. 2002a, 2002b; McHenry et al. 2016, 2017). Nonetheless, although *L. aurea* persist on the island and have been observed breeding in wetlands with *G. holbrooki* (Hamer et al. 2002a), we consider that predation of larval *L. aurea* has a negative impact on their population size, which in turn makes the population more vulnerable to chytrid impacts. Our approach was to develop a concept design for wetland habitat that would reduce the likelihood of colonization by *G. holbrooki* through the creation of perimeter bunding around the edge of each wetland to prevent surface water ingress through overland flow.

Composition of wetland and terrestrial plant communities were an important part of the rehabilitation plans. Previous

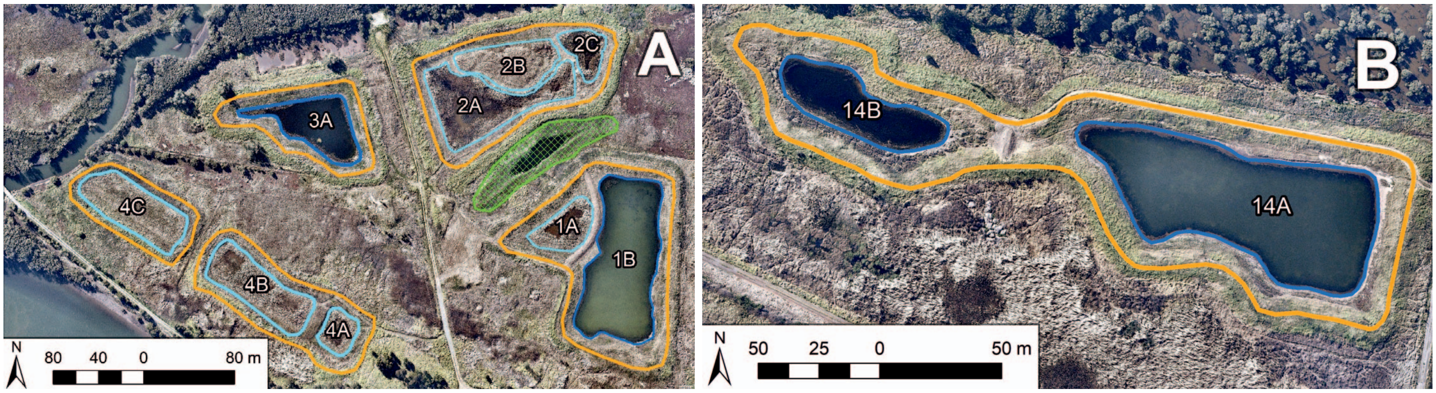


**Figure 2.**—Aerial photographs of the *L. aurea* compensatory wetlands. (A) Primary site. (B) Secondary site. Dark blue indicates permanent wetlands. Light blue indicates ephemeral wetlands. Orange indicates the extent of perimeter bund walls. Green cross-hatching indicates extent of natural wetland.

studies of wetland occupancy by *L. aurea* on KI had identified several emergent reeds as being favored (Hamer et al. 2002a); however, a similar study at Sydney Olympic Parkland revealed a different suite of plants (Midson 2010). Together the studies of wetland plants at occupied and unoccupied habitat give strong indication that plant structure is more important than floristic composition (Pollard 2009; Midson 2010; Garnham et al. 2015; Valdez et al. 2017a).

Selection of the site was made so that biodiversity of the area was maximized. This meant avoiding threatened communities such as salt marsh and mangroves, and choosing degraded communities. Selection of sites was also guided by the presence of acid sulphate soils; this substrate was avoided. Two sites fit for this purpose were identified (–32.852816, 151.710752 and –32.858765, 151.719757), where the former was the site allocated to the primary site of the wetland mosaic, and the latter was allocated to secondary site to increase landscape connectivity among wetlands. Both of these sites were rank cattle-pasture land. A detailed ecological risk assessment was made to determine if there would be negative impacts to any threatened species or communities that potentially occupy or use this habitat. It was deemed that there would be little impact to threatened species and communities, and that there would likely be positive outcomes for some species.

## Construction of the Compensatory Habitat

Construction of the CHP was guided by the concept design we developed, and this was subject to third party review and endorsement. This formed the basis of a detailed design and technical specification commissioned by the proponent.

A total of 2.6 ha of *L. aurea* breeding habitat was constructed in 2015/2016, which consisted of 11 wetlands (Figures 2A, 2B). There are four ephemeral, three semi-permanent, and four permanent wetlands (Table 1; Beranek and Mahony 2017). We define an "ephemeral wetland" as one that dries at least once during a year, and a "permanent wetland' as one that always retains water.

To prevent surface flow to the constructed wetlands, and the possibility that *G. holbrooki* would invade, earthen walls (bunding) were constructed around each wetland using the material excavated during the construction of the wetlands. The walls were constructed to be >0.6 m above ground level (agl) for ephemeral wetlands, ~1.2 m agl for permanent wetlands, and ~3.5 m wide, and were compacted to meet engineering stability standards.

Agricultural lime was applied to the base and sides of the wetland excavations at a rate of 1 kg/m$^2$, because the soils on KI have a potential for acid-sulfate formation. This was only performed for ephemeral wetlands, as lime could not be added to permanent wetland basins due to immediate groundwater recharge. Lime was also added to all excavated material that was used to form bunding walls.

Aquatic, riparian, and terrestrial zone vegetation was carefully planned in consultation and with consent of National Parks and Wildlife Services and the Office of Environment and Heritage New South Wales. Consideration was given to species known to be favorable for *L. aurea* habitat and also to the relationship between wetland hydrology and plant occurrence. The objective was to establish a diversity of native plants, and to avoid monocultures that often occur in restored wetlands. There were 12 local native wetland plant species that were allocated (Table 2). The majority were chosen as they had previously been shown to be positively associated with *L. aurea* abundance, occupancy, and shelter sites (Table 2). *Typha* sp. and *Phragmites australis* were avoided; they are known to be used by *L. aurea* but rapidly form monocultures and can result in the infilling by organic matter in shallow ephemeral wetlands. The chosen species offer different structures. For example, there are large species such as *Schoenoplectus vallidus*, which has straight erect leaves that can reach 2 m in height. In contrast, there are smaller species that have more dense leaf arrangements and have strappier leaves such as *Bolboschoenus fluviatilis*, which reaches ~1 m in height.

**Monitoring and Adaptive Management of the Compensatory Habitat:** The proponent responsible for the CHP was required to demonstrate success in terms of occupancy, breeding, habitat, and water quality. They were also required to manage the terrestrial components of the habitat areas (i.e., including terrestrial weed and feral animal management).

Measuring the success of the constructed wetlands was based on setting thresholds (key performance indicators, KPIs), that were prescribed in the proponents' CHP consent conditions and adaptive management triggers in case there were KPIs that were

**Table 1.**—Wetland hydrology classification, surface areas, and depths for the *L. aurea* breeding habitat. Ephemeral = dries at least once throughout the year. Permanent = never dries. Mean proportion of days dry per year was measured yearly from September 2016 to August 2019.

| Wetland code | Wetland type | Wetland area ($m^2$) | Depth (m) | Australian Height Datum (base of wetland) | Mean proportion of days dry/year ± SE | Mean breeding season salinity (ppt) ± SE |
|---|---|---|---|---|---|---|
| 1A | Ephemeral | 1447 | 0.7 | 0.3 | 0.15 ± 0.02 | 1.04 ± 0.10 |
| 1B | Permanent | 6250 | 1.8 | −0.8 | 0.00 ± 0.00 | 11.64 ± 0.43 |
| 2A | Ephemeral | 3380 | 0.4 | 0.5 | 0.21 ± 0.02 | 1.67 ± 0.27 |
| 2B | Ephemeral | 1790 | 0.3 | 0.7 | 0.37 ± 0.06 | 1.58 ± 0.52 |
| 2C | Ephemeral | 777 | 0.5 | 0.2 | 0.04 ± 0.02 | 2.73 ± 0.38 |
| 3A | Permanent | 2703 | 1.5 | −0.2 | 0.00 ± 0.00 | 4.71 ± 0.26 |
| 4A | Ephemeral | 720 | 0.5 | 0.9 | 0.32 ± 0.04 | 0.76 ± 0.47 |
| 4B | Ephemeral | 2218 | 0.4 | 0.9 | 0.35 ± 0.04 | 1.28 ± 0.73 |
| 4C | Ephemeral | 1949 | 0.5 | 0.8 | 0.51 ± 0.04 | 0.18 ± 0.03 |
| 14A | Permanent | 3700 | 1.5 | −0.7 | 0.00 ± 0.00 | 21.63 ± 0.70 |
| 14B | Permanent | 1102 | 1.5 | −0.5 | 0.00 ± 0.00 | 6.60 ± 0.28 |
| Total wetland area ($m^2$) = | | 26,036 | | | | |

not being met. Primary KPIs were set for adult occupancy and the occurrence of breeding. In addition, there were also secondary KPIs set for abiotic and biotic features of the wetlands that influence *L. aurea* occupancy, breeding, and recruitment. Secondary KPIs' conditions/parameters serve as management guidelines for the maintenance of the *L. aurea* breeding habitat and define "trigger conditions," which may require adaptive management. An adaptive management strategy was established for frog activity, water quality, vegetation, and *G. holbrooki* presence, such that actions were to be triggered once monitoring showed that some aspect of the habitat was outside the predetermined threshold values (Table 3).

This monitoring strategy of *L. aurea* frogs and tadpoles was considered robust to detect mature adults and breeding. It is likely that *L. aurea* has a similar high rate of detection to the closely related *L. raniformis*, where frogs of this species had a detection probability of ~0.7 (Heard et al. 2006). Monthly surveys for *L. aurea* tadpoles with Fyke nets is appropriate given that tadpoles of *L. raniformis* had the highest detection probability with this method compared to other techniques (Wassens et al. 2017). *L. aurea* has a tadpole development period of 1.5–3 mo (Anstis 2017), and therefore monthly Fyke netting surveys typically would achieve 2–3 valid samples. At a detection probability of 0.7, two samples would achieve 85% confidence in detection if present, and three samples would achieve 95% confidence in detection (Mackenzie and Royle 2005). Therefore, if 95% confidence is important for quality assurance, surveyors may wish to assure that three samples are taken by sampling on slightly shorter intervals (~3 wk). If applying this method to other species, sampling effort should account for detection rates and sample sufficiently to achieve an acceptable level of confidence in detection.

## DISCUSSION

Here we report on the creation of aquatic and terrestrial habitat that has been designed based on past learnings to passively manage threatening processes. We describe the physical and biotic components of the constructed habitat, design and construction phases, and monitoring and adaptive management. This wetland design addresses threats that have been detrimental to past *L. aurea* population restorations and provides optimal breeding habitat discerned from >20 y of research on this species (Valdez et al. 2017b).

The concept design was based on what was currently known about *L. aurea* breeding biology, and also of the sibling species *L. raniformis* (Pyke and White 2001; Pyke 2002; Wassens et al. 2010; Scroggie et al. 2019). This design also facilitates coexistence with chytrid by attempting to achieve the optimum tradeoff

**Table 2.**—Vegetation planted within the wetlands.

| Species | Common name | Total | Reference of positive associations with *L. aurea* |
|---|---|---|---|
| *Bolboschoenus caldwellii* | Marsh club-rush | 5262 | Valdez et al. (2016, 2017a) |
| *Bolboschoenus fluviatilis* | Marsh club-rush | 2270 | Valdez et al. (2016, 2017a) |
| *Carex appressa* | Tall sedge | 2993 | Patmore (2001) |
| *Eleocharis acuta* | Small spike-rush | 2993 | Patmore (2001) |
| *Eleoacharis sphacelata* | Tall spike-rush | 2993 | Patmore (2001) |
| *Juncus usitatus* | Common rush | 1505 | Valdez et al. (2016, 2017a) |
| *Lomandra longifolia* | Spiny-headed mat-rush | 1505 | Garnham et al. (2015) |
| *Persicaria decipiens* | Slender knotweed | 1505 | |
| *Persicaria praetermissa* | Spotted knotweed | 1505 | |
| *Ranunculus inundatus* | River buttercup | 160 | |
| *Schoenoplectus validus* | River club-rush | 160 | Midson (2010); Valdez et al. (2016, 2017a) |
| *Triglochin procera* | Water-ribbons | 160 | |
| | Total plants: | 23,006 | |

**Table 3.**—Adaptive management triggers for the success of the *L. aurea* compensatory wetlands.

| Objective | Monitoring | Trigger conditions | Adaptive management |
|---|---|---|---|
| Establishment of breeding events in two seasons. | Dip netting, Fyke netting, or tadpole traps. Once per month during breeding season. | If there is no evidence of breeding within a season. | 1. Enhance colonization using recorded bell frog calls (James et al. 2015). 2. Captive breeding and release of tadpoles. |
| Evidence of at least five reproductively mature individuals in wetlands where breeding events have occurred. | Visual encounter surveys, auditory surveys (once per two weeks), and sound recording devices. | If there is no evidence of reproductively mature individuals within a season. | |
| At least one permanent wetland retains water throughout the year. | Manual depth measuring and water depth loggers. | If no permanent wetlands hold water throughout a year. | Examine rates of change on depth loggers, seek hydrological expert advice, and explore the option of excavation to deepen wetland. |
| Achieve water quality results that are conducive for *L. aurea* habitat use and breeding. | Water quality monitoring once per week. Implement data loggers for salinity, water temperature, and dissolved oxygen. | If a water quality parameter consistently breaks set thresholds. | Assess if healthy *L. aurea* tadpoles are occupying the wetland. If so, adjust parameter thresholds. Investigation the cause, e.g., flooding, drought, acid sulphate soils, or nearby land-use changes. Consult specialist advice. |
| Maintain an open water percent cover of 25%. | 20 × 20 m vegetation plots three times per year. | Open water percent cover dropping below 25%. | Undertake emergent vegetation removal. |
| Keep breeding habitat free of aquatic noxious weeds. | 20 × 20 m vegetation plots three times per year. | Discovery of any aquatic noxious weeds in a wetland. | Manual removal of plants with follow-up removal until infestation is cleared. |
| Keep compensatory wetlands free of *G. holbrooki*. | Dip netting, Fyke netting, or tadpole traps. Once per month during breeding season. | Discovery of *G. holbrooki* in a wetland. | If found in an ephemeral wetland, no action is required as the wetland will dry and all *G. holbrooki* will desiccate, and thereby be passively treated. If found in a permanent wetland, netting should occur periodically to reduce population size. |

between limiting the occurrence of chytrid and promoting survival of *L. aurea*, with the model suggested by Stockwell et al. (2015a) and conceptualized by Scheele et al. (2019a). However, we extend this model by also including abiotic structures that prevent colonization by *G. holbrooki*, which we consider is a source of mortality in early stage development periods. Both ephemeral and permanent wetlands were incorporated in the design as there is evidence for *L. aurea* breeding in both situations (Hamer et al. 2002b). Deeper permanent wetlands were also incorporated in the landscape to provide drought refuge and connectivity, as isolated wetlands have been demonstrated to be negatively correlated with *L. aurea* occupancy (Bower et al. 2013; Valdez et al. 2015). Engineering the constructed wetlands so that salinity levels were outside the fundamental niche of chytrid required hydrological modeling to identify the optimum depth to enable salt influence from the groundwater. Several studies have shown *L. aurea* tadpoles can successfully grow in salinity levels of 9 ppt (Werkman 1999; Pyke et al. 2002; Stockwell 2011), and the wetlands were designed with the intent to maintain optimum salinity levels. Additionally, native flora that are positively associated with *L. aurea* occupancy such as *Juncus usitatus* and *Shoenoplectus litoralis* were planted in the wetlands (Hamer et al. 2002a; Valdez et al. 2016, 2017a).

The wetlands were designed to passively manage chytrid, which is one of the first attempts to control for this disease in an amphibian restoration project (Stockwell et al. 2015a; Klop-Toker et al. 2016, 2018; Scheele et al. 2019a). In theory, this is achieved by including wetlands of varying salinity and ephemerality, as chytrid has been shown to be less active with higher salinity concentrations, increased water temperature, and can be eliminated after wetland drying (Forrest and Schlaepfer 2011; Scheele et al. 2015; Stockwell et al. 2015c). In this scenario, wetlands that are deeper are permanent and more saline, as they intersect the groundwater table, whereas wetlands that are shallower are ephemeral and less saline, as they do not intersect the groundwater table. Therefore, chytrid may be passively managed in the ephemeral wetlands by desiccation and in the permanent wetlands by increased salinity.

The threat of *G. holbrooki* can be mitigated by removing dispersal pathways by constructing "bunds" around the periphery of the wetlands. These bunds act as a "predator-proof fence" for *L. aurea* breeding habitat (Hayward and Kerley 2009; Kerezsy 2015). *G. holbrooki* disperse using overland flow of water during floods to reach new water bodies (Chapman and Kramer 1991). However, there is anecdotal evidence that *G. holbrooki* are able to reach new water bodies via biological vectors, such as water birds (C. McHenry and A. White pers. comm.). Waterfowl have been found to transport killifish (*Austrolebias* sp.) eggs via gut passage (Silva et al. 2019). However, *G. holbrooki* give birth to live young and birds acting as a biological vector for their transport has not been experimentally tested. Overland flow is likely more important for dispersal. One shortcoming of bunding is that it limits the catchment size, which reduces the replenishment of water from rainfall. However, in this situation, we were able to use bunds due to the high groundwater table and hydrological predictions modeled the water level in respect to groundwater and rainfall for the local system. In theory, this will remove the threat of *G. holbrooki* predation on early life stage *L. aurea*, which should enable a higher survival rate and thereby enhance *L. aurea* populations (Morgan and Buttemer 1996). In conclusion, the approach used in this project showcases an effective way of designing and implementing a wetland restoration project for a threatened amphibian. First, the threats of the target organism or ecosystem requires thorough research so that passive and active management options can be explored and developed. Secondly, precise parameters were defined based on realistic objectives to control the threat, determined the success of the project, and which can be monitored accurately. Thirdly, adaptive management triggers and response actions were developed in case the restoration efforts did not reach targets.

## CONCLUSION

The breeding habitat design we present here is a result of >10 y of research, which accounts for the currently known optimal habitat for *L. aurea* and passive management controls for their threats. This includes passive management of the chytrid fungus, which currently has no known large-scale treatment (Garner et al. 2016). This case study is one of the first to implement passive control for the chytrid fungus by including habitat that naturally incorporates wetland drying and fluctuations of salinity, both of which are known to hinder chytrid (Scheele et al. 2015; Stockwell et al. 2015a; Clulow et al. 2017). In addition, there are clearly defined parameters to determine success, measured as thresholds coupled with adaptive management triggers. All of this careful planning comes together to provide a robust template to create a viable breeding population of *L. aurea*. Future publications will document the population dynamics of *L. aurea* in response to this created habitat and aim to further refine the design to benefit future wetland creation projects for amphibian conservation.

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