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Biological Conservation 119 (2004) 61–71
BIOLOGICAL CONSERVATION
www.elsevier.com/locate/biocon

A comparison of constructed and natural habitat for frog conservation in an Australian agricultural landscape

Donna Hazell [a,*], Jean-Marc Hero [b], David Lindenmayer [a], Ross Cunningham [a]

[a] Centre for Resource and Environmental Studies, The Australian National University, Canberra, ACT., 0200, Australia
[b] School of Environmental and Applied Sciences, Griffith University Gold Coast, PMB 50, Gold Coast Mail Centre, QLD, 9726 Australia

Received 3 December 2002; received in revised form 16 October 2003; accepted 22 October 2003

Abstract

Constructed ponds are an important consideration in the conservation of wetland biota in agricultural landscapes. Twenty-two natural ponds and 22 adjacent constructed ponds (farm dams) were surveyed on the Southern Tablelands of New South Wales to compare patterns of use by frogs and develop frog conservation recommendations. Farm dams supported similar numbers of frog species to natural ponds, although differences in frog assemblage were observed between the pond types. *Limnodynastes tasmaniensis* and *Uperolia laevigata* were significantly more likely to occur at farm dams while *L. peronii* was more likely to occur at natural ponds. Results suggest waterbodies with high levels of emergent vegetation cover that lack fish are likely to support a high number of frog species, regardless of origin (i.e. natural or constructed). However, it is important for landholders to conserve natural waterbodies as these environments appear likely to support frog species that do not use farm dams.

Keywords: Frogs; Wetlands; Fish; Farms; Australia

1. Introduction

Human impacts on the environment have caused habitat loss worldwide (McNeely et al., 1995). This has raised the profile of habitat construction, which plays an important role in conservation (Anderson, 1995). One example is the construction of ponds, which are considered important habitat for a range of aquatic species (e.g. Fry, 1991; Hull and Boothby, 1995; Bignal and McCracken, 1996; Boothby, 1999). Many frogs require aquatic environments in which to breed and their use of constructed ponds as habitat has been the focus of several studies (e.g. Laan and Verboon, 1990; Hecnar and M'Closkey, 1997; Stumpel and van der Voet, 1998; Baker and Halliday, 1999).

In Australia, many natural wetlands have been destroyed or severely degraded (State of the Environment Advisory Council, 1996) but constructed ponds (in the form of farm dams) are widespread, particularly in agricultural areas of south-eastern Australia. Farm dams are an important element in the conservation of wetland biota as they may provide an alternative to natural habitat. They have been promoted as valuable habitat for water birds (Hill and Edquist, 1982; Smith, 1983; Brouwer, 1995). Hazell et al. (2001) demonstrated that a range of frogs use farm dams, but their capacity for providing suitable frog habitat as an alternative to natural wetlands is unknown.

Farm dams differ from natural ponds in several ways. While farm dams may be constructed almost anywhere, natural ponds only occur where surface water collects or is discharged from sub-surface flow. Farm dams have been described as less complex in structure and biological diversity than natural ponds (Brock and Jarman, 2000). For these reasons, we predict that frogs would show a preference for natural ponds and that natural habitat would support a greater number of species than constructed habitat.

This study compares the ways in which frogs use natural and constructed ponds in the upper Shoalhaven catchment on the Southern Tablelands of south-eastern

* Corresponding author. Fax: +61-2-6125-0757.
E-mail address: dhazell@cres.anu.edu.au (D. Hazell).

0006-3207/$ - see front matter
doi:10.1016/j.biocon.2003.10.022

Australia. Differences in the characteristics of natural ponds and farm dams were quantified. The objective was to determine whether natural ponds supported more frog species and/or larger populations of frogs than farm dams. We also set out to determine if particular species of frog were more likely to occur in natural ponds than farm dams. Finally, we recommend ways in which waterbodies in agricultural landscapes may be managed in order to maximise habitat value for frogs.

2. Methods

2.1. Study area

The upper Shoalhaven catchment is situated on the Southern Tablelands of New South Wales in south-eastern Australia. This region was substantially altered through European-derived land management practices introduced in the early to mid 1800s. These included clearing forest, planting crops, grazing horses, sheep and cattle, and gold mining (Duffy, 1969; Ellis, 1997; McGowan, 2000). The upper Shoalhaven catchment consists predominantly of grazing country and has been described in detail by Hazell et al. (2001).

Clearing of native vegetation, draining of swamps, grazing and stream disturbances (such as creek crossings) are likely to have had major impacts on stream systems of the region (Eyles, 1977a; Prosser, 1991; Starr, 1999). Prior to European settlement, low-energy stream systems of intermittent or discontinuous flow were common across this region, and contained deep, steep-sided ponds that were relatively permanent (Eyles, 1977b; Brierley and Fryirs, 1999). Many of these chain-of-ponds systems were quickly converted to incised channels as a result of the introduction of European land use practices (Eyles, 1977b; Wasson et al., 1998). Potential ecological implications of these changes have been discussed by Hazell et al. (2003).

2.2. Experimental design

We undertook a catchment-wide search for natural ponds using local knowledge and an analysis of maps and aerial photographs. All waterbodies created through non-human, biophysical processes were considered natural. This included billabongs (or ox-bow lakes), other wetlands, and chain-of-ponds. Billabongs are lentic waterbodies that occur on riverine floodplains, formed by the geomorphic action of an adjacent flowing stream system (Hillman, 1986). Wetlands occur where depressions, seepages or springs (sometimes combined with impermeable soils) allow water to accumulate at the surface (Campbell, 1983). Chain-of-ponds systems were described in Section 2.1.

Twenty-two natural pond systems were studied (Fig. 1). Some pond systems consisted of areas or drainage lines with multiple ponds. In these cases, each pond was mapped and numbered. A study pond was then randomly selected from the numbered ponds within each system. A farm dam was matched with each study pond on the basis of comparable management practices (i.e. levels of stock access and stock type) and surrounding landscape features. This was to control for the effects of these variables on frog habitat suitability. Each farm dam was at least 1 km from its respective natural pond. It was assumed this would minimise frog movement between study sites on a given night and ensure that frogs calling from one study site could not be heard from another. Farm dams were no further than 3 km from their respective natural ponds to ensure that climate and landscape features (such as topography and surrounding native vegetation cover) were similar. Farm dams that were less than 10 years old were excluded from the study to ensure that frog communities had time to colonise these waterbodies.

2.3. Frog data

Sites were surveyed during the spring/summer of 1999/2000. This is the main activity period for a majority of the frog species known to occur within the region (see Dankers, 1977; Humphries, personal observation). Surveys commenced once spring breeding species became active. Surveying on a given night ceased if the night air temperature dropped below the long term recorded mean minimum daily temperature for the spring months in the catchment (approximately 5 °C). Reference sites (i.e. waterbodies where certain frog species were known to occur) were visited when night air temperatures dropped below 10 °C to confirm calling activity. Other sites were surveyed only if frogs were calling at the reference site. This was considered necessary as minimal temperature thresholds for calling activity can vary on a night to night basis, depending upon time of the season, humidity, temperature during the day and previous rainfall (Dankers, 1977). Each site was surveyed at night on two separate occasions. Sites in a given dam/pond pair were sampled on the same night under similar air temperature conditions. Standardised frog survey techniques were employed at each site. This involved audio strip transect sampling and visual transect sampling (Jaeger, 1994; Zimmerman, 1994). The influence of weather on the behaviour and activity of frogs was considered when scheduling surveys (see Crump, 1994; Hazell et al., 2001 for further details). Order of site visits was random (i.e. farm dams were not always surveyed first).

Each study site occurred along a moisture pathway or drainage line. Aural transects surveys were conducted for 20 min along these drainage lines in order to capture

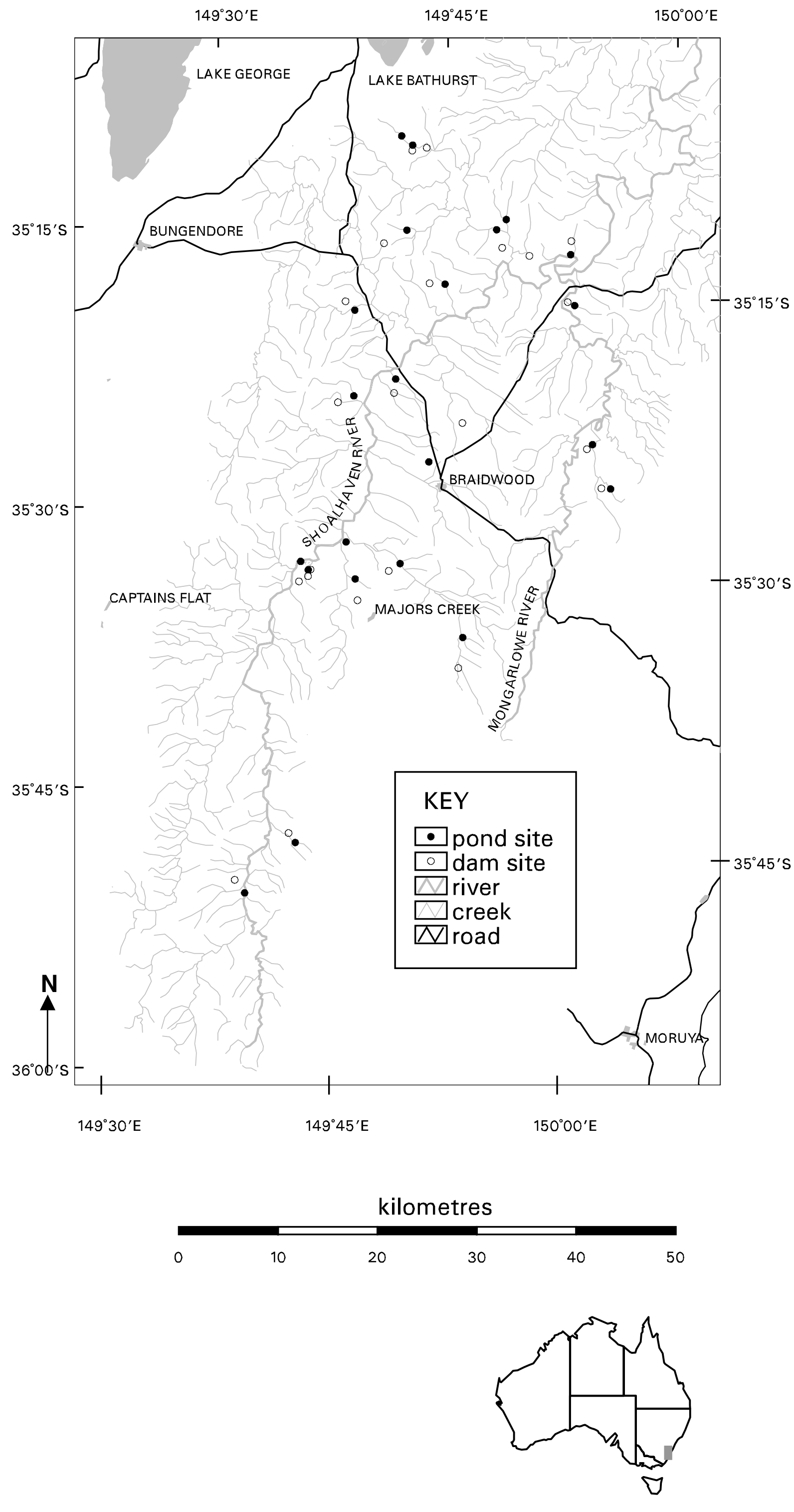

Fig. 1. Location of study sites within the upper Shoalhaven catchment, south-eastern Australia. The map of Australia indicates location of the catchment in grey.

species calling from terrestrial, semi-aquatic and aquatic microhabitats associated with each waterbody. Each aural transect was 50 m long with the waterbody as the central point. This distance was considered adequate to capture a variety of potential calling sites. We estimated chorus size across the entire transect for each species using logarithmic-scaled categories (category 0 = 0 frog; 1 = 1–5 frogs; 2 = 6–10; 3 = 11–20; 4 = 21–50; 5 = 50+). Category values were averaged across the two visits at each site. This provided an index of male frog response for comparison between waterbodies surveyed on the same night.

We surveyed four visual strip transects for frogs at each site. The margin of each pond was divided into broad microhabitat environments (i.e. areas with bare riparian zone, areas with emergent vegetation in the shallow water, areas with riparian shrub or canopy cover). Transects were randomly placed within each of the microhabitat zones present. The coverage of each microhabitat was taken into account. For example, when four microhabitats were present one transect was randomly placed within each microhabitat. If there were only three microhabitats present then three transects were randomly placed within each microhabitat with the fourth transect randomly placed within the microhabitat with the greatest coverage. Each transect consisted of a 5 m long and 2 m wide strip. The waters edge was used as the lengthwise, centre-line for this strip so that both riparian vegetation and shallow water were included in the survey. Species were recorded as present if any individuals were observed along the visual strip transects or if they were heard calling along the audio strip transect. Species that were not detected at a site were recorded as absent.

2.4. Fish and habitat data

Thirteen of the natural ponds were chain-of-ponds stream systems, seven were ponds associated with flowing stream systems (such as billabongs), and two were other wetlands or swamps not associated with flowing stream systems. Most sites had been modified to some extent. Several had been deepened or widened. Water had been diverted in some pond systems. In one instance, a history of gold dredging resulted in the destruction of natural ponds and the construction of artificial ponds in the same area. The study site selected from this highly disturbed natural system was included as a natural pond, but was observed closely during data analysis.

Fish are major predators of frogs during their larval stage (Heyer et al., 1975) and are therefore an important consideration when examining habitat use. We surveyed for fish by dragging a seine net through each waterbody. This net was 16 m long by 1.5 m high with mesh square diameter of 8 mm. Small fish were targeted with 10 dipnet sweeps. Fish species were identified from McDowall (1996). Fish also were recorded as present at a site if they were observed during night surveys. We estimated the percentage of the water margin with emergent vegetation cover for each waterbody. Riparian vegetation was surveyed within a 2 m wide band around the bank of each waterbody. This area was surveyed at ground level for percentage cover of bare ground and at grass tussock height (0.5 m) for percentage cover.

Four water depth measurements were taken at regular intervals around each waterbody at a distance of 1 m from the waters edge. These measurements were used to indicate the availability of shallow water at the pond edge. We took several depth measurements at the centre of each water body to establish maximum depth to the nearest 50 mm. We also recorded electrical conductivity, pH, dissolved oxygen and temperature using a WTW Multiline P4 universal pocket meter. These parameters were measured during night-time surveys at each site for consistency. Measurements were taken 30 cm from the waters edge.

2.5. Statistical analysis

We used logistic regression analysis (McCullagh and Nelder, 1989) to develop habitat models for each frog species. Sample size ($n = 44$) did not allow the inclusion of all habitat variables in these analyses. Five potential explanatory variables were chosen a priori. These were pond type (PT – recorded as either natural pond or farm dam), percentage of the water margin with emergent vegetation (EV), maximum recorded depth (DEPTH), presence of fish (FISH), and percentage cover of bare ground in the riparian zone (BG) (see Table 1). These variables were chosen because of their likely biological importance (e.g. fish are known to be a major predator for frogs) or because they had proved to be significant explanatory variables for habitat use in previous studies (see Hazell et al., 2001). All other variables are presented as additional information on farm dams and natural ponds (Table 1). Variables chosen for the logistic regression analysis were first examined for collinearity using scatterplots. When collinearity was evident between two explanatory variables, the variable considered to be of least biological importance was removed from further analysis. Sensitivity and specificity were used as a measure of logistic model accuracy (Fielding and Bell, 1997). A prediction of 0.5 or higher was considered as a predicted presence to calculate these measures.

We examined the index of chorus abundance with a mixed model for each species (Searle et al., 1992). This was to determine if there was a relationship between the number of male frogs calling and the pond type. The variable 'pair' was included as a random effect in 2the mixed model to identify similarities in chorus size within dam/pond pairs. Such similarities may result through sites being surveyed on the same night, or being located close together. Within-pair similarity can obscure overall differences between farm dams and natural ponds.

Abundance data with large numbers of zero values (i.e. greater than 30%) were analysed in the mixed model with the zero abundance datapoints removed. This was necessary to satisfy the underlying distributional properties of data required for mixed models. A mixed model was also used to examine species richness against those

Table 1
Summary statistics for continuous habitat attributes measured at farm dams and natural ponds

Pond type variable	Modelling code	Units	Farm dam		Natural pond		Paired *t*-test (*t*) or Wilcoxon signed-rank (*z*)
			Mean (SE)	*n*	Mean ± (SE)	*n*	
Percentage of 2 m wide riparian zone without ground cover vegetation	BG	%	25.9 (4.5)	22	3.9 (1.3)	22	$z = 3.521$, 19df, $p < 0.001$
Percentage of waterbody perimeter with emergent vegetation at edge	EV	%	57.3 (7.9)	22	94.5 (1.7)	22	$z = 3.04$, 20df, $p = 0.002$
Maximum waterbody depth	DEPTH	cm	198.5 (25.4)	20	128.6 (22.2)	22	$t = 2.91$, 19df, $p = 0.009$
Average water depth 1 m out from waters edge		cm	19.8 (1.4)	22	36.0 (6.0)	21	$t = -2.79$, 20df, $p = 0.011$
pH			7.1 (0.2)	22	6.9 (0.2)	22	NS
Electrical conductivity		μs/cm	97.8 (22.1)	22	260.1 (72.6)	22	$z = 2.68$, 21df, $p = 0.001$
Percentage of 2 m wide riparian zone with tussock cover		%	26.8 (6.3)	22	60.0 (8.0)	22	$z = 3.23$, 21df, $p = 0.001$
Surface area of waterbody during breeding season		m^2	1131.5 (367.0)	21	1342.6 (533.9)	21	NS
Maximum dissolved oxygen (saturation)		%	103.8 (3.9)	22	72.8 (5.1)	22	$t = 4.40$, 21df, $p < 0.001$

Variables with modelling codes were chosen a priori for statistical analysis. This also included categorical variables of pond type (PT) and the presence of fish (FISH). Wilcoxon signed-rank test was applied when assumptions for the paired *t*-test could not be met. NS = not significant.

habitat attributes included in the logistic models as defined above. Further details on mixed models are presented in Hazell et al. (2001).

The proportion of male frogs actively chorusing in a population can vary on a daily or seasonal basis depending upon environmental conditions such as temperature and rainfall (Dankers, 1977; Humphries, 1979). Chorus sizes recorded at different sites on different dates are therefore unlikely to represent comparable population proportions. Between-site comparisons of chorus size are only valid if they represent a consistent sub-set of the population (Zimmerman, 1994). It was not possible to survey all sites simultaneously in this study. Survey effort on any given night was therefore evenly distributed between farm dams and natural ponds to ensure that chorus size could be compared between pond types.

2.6. Engaging landholders

Farmers and other landholders provided information on current and historical land management. This was made possible by working closely with landholders while engaging in ecological research on their land. One of the authors (Hazell) developed ethical and methodological principles that guided how this research was undertaken on private land. These principles detail appropriate ways to approach landholders and develop working relationships, the importance of encouraging participation, showing respect and recognising landholders rights to data and knowledge, as well as the importance of communication. This approach has considerable value with respect to developing meaningful management recommendations for frog conservation on private land.

3. Results

3.1. Frogs

At least one species of frog was recorded at each of the 44 waterbodies surveyed. We recorded 12 frog species across the 22 natural ponds. Ten species were recorded across the corresponding farm dams (Fig. 2). The number of species recorded at each farm dam varied from two to eight while the number of species recorded at each natural pond varied from one to eight (Fig. 3). An average of 5.0(±0.3 SE) species was recorded from farm dams while the corresponding value from natural ponds was 4.5(±0.4 SE).

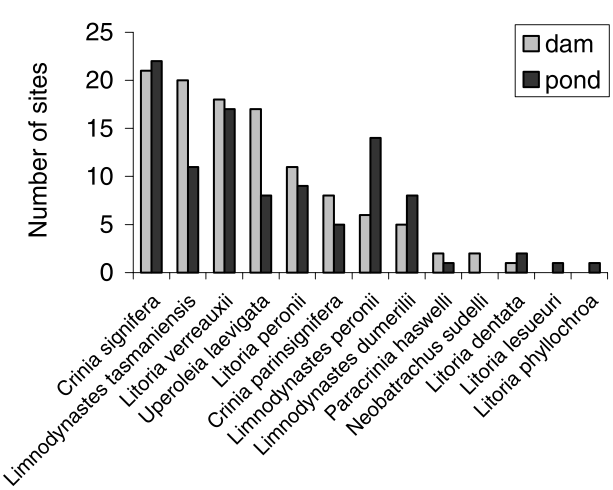

Fig. 2. Total number of sites at which each individual frog species was recorded. Sites are shown as farm dams or natural ponds.

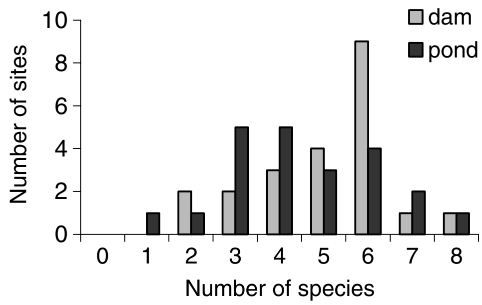

Fig. 3. Total number of frog species recorded at each farm dam and natural pond site.

Crinia signifera was recorded more often than any other species at both natural ponds (100%) and farm dams (95%). The two least common species (*Litoria lesueuri* and *L. phyllochroa*) were only recorded at natural ponds (Fig. 2). A correlation in species presence within dam/pond pairs was found for *C. parinsignifera* (Pearson chi-squared value $= 12.63$, $p < 0.001$) and *Limnodynastes peronii* (Pearson chi square value $= 4.48$, $p = 0.034$). In logistic regression models for these species, response to pond type may be obscured by spatial similarities in the data. Logistic models for these species were interpreted with consideration of this.

3.2. Habitat characteristics and predators

Electrical conductivity (EC) was significantly higher on average in natural ponds than in farm dams (see Table 1). pH values for both farm dams and natural ponds were primarily neutral, although alkaline extremes were recorded for both pond types. Maximum dissolved oxygen levels were significantly lower in natural ponds than farm dams with dissolved oxygen well below saturation level at the former (Table 1). Natural ponds also had significantly less bare ground and more tussock cover in the riparian zone with significantly more emergent vegetation cover in the shallow water zone (Table 1). Natural ponds were not as deep as farm dams at maximum depth, although they had less shallow water on average (Table 1). High levels of turbidity were observed in many of the natural ponds, particularly the chain-of-ponds systems.

Fish were present at 12 natural ponds. We identified the introduced mosquito fish (*Gambusia holbrooki*) in five ponds and the mountain galaxia (*Galaxias olidus*) in five ponds. The two species co-occurred in one of these ponds. Fish were seen, but not identified at the remaining three ponds. We recorded *G. olidus* at only one farm dam.

3.3. Models for the probability of occurrence of individual frog species

Data were available to fit logistic regression models for seven of the 13 frog species recorded. Two of the habitat attributes chosen for statistical analysis were collinear (emergent vegetation at the water margin (EV) and bare ground in the riparian zone (BG)). For this reason only one of these variables (EV) was included in the analysis. None of the habitat attributes chosen for the analysis were found to be useful explanatory variables for the occurrence of *C. parinsignifera*, *L. dumerilii* and *L. verreauxii*. Logistic regression models were developed for *Limnodynastes peronii*, *L. tasmaniensis*, *Litoria peronii* and *Uperoleia laevigata*.

Pond type (PT) was a significant predictor of the probability of occurrence of three species. *Limnodynastes peronii* was more likely to occur at natural ponds while *L. tasmaniensis* and *U. laevigata* were more likely to occur at farm dams (Fig. 4 and Table 2). The presence of fish was a significant, negative explanatory variable for the presence of all four species modelled. In the case of *L. tasmaniensis*, the presence of fish was interchangeable with pond type. The percentage of the water margin with emergent vegetation was a significant positive predictor for *U. laevigata* (Fig. 4) and *Litoria*

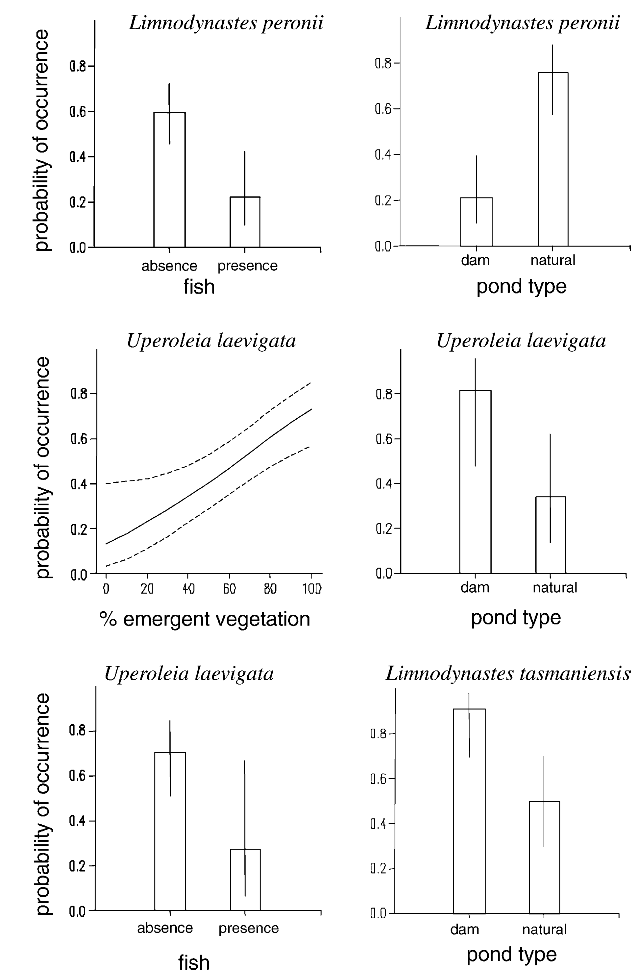

Fig. 4. Predicted values and associated 95% confidence estimates from a model of habitat factors significantly related to the occurrence of *U. laevigata*, *L. tasmaniensis* and *L. peronii*.

Table 2
Presence/absence models for individual species and adequacy of predictions

Species	Fixed effect	Estimate (SE)	Change in deviance	$p(x^2_{t-1})$	Error	Sen	Spe
Limnodynastes peronii	Constant	−0.92 (0.48)			0.27	0.45	0.96
	PT (n)	3.15 (1.15)	12.28	<0.001			
	FISH	−2.58 (1.19)	6.63	0.01			
L. tasmaniensis							
Model 1	Constant	2.30 (0.74)			0.3	0.65	0.85
	PT (n)	−2.30 (0.85)	9.51	0.002			
Model 2	Constant	1.91 (0.53)			0.18	0.87	0.69
	FISH	−2.72 (0.80)	13.52	<0.001			
Litoria peronii	Constant	−1.581 (0.91)					
	EV	0.024 (0.012)	4.85	0.028	0.32	0.60	0.75
	FISH	−1.50 (0.77)	4.06	0.040			
Uperoleia laevigata	Constant	−0.69 (0.88)			0.20	0.88	0.68
	EV	0.05 (0.02)	7.96	0.004			
	PT (n)	−3.07 (1.43)	6.28	0.01			
	FISH	−2.93 (1.10)	8.96	0.002			

Fitted values of 0.5 and above were considered as presence predictions. Error rate is the proportion of the predictions that were incorrect. Sensitivity (Sen) measures how well the model predicts species presence while specificity (Spe) measures how well the model predicts absence. Values are expressed as a proportion of 1. PT = pond type, FISH = fish presence, EV = percent emergent vegetation cover.

peronii (Table 2). All species models had relatively low levels of prediction error (Table 2).

3.4. Chorus size

Chorus size was compared between pond types using mixed models for seven species: *C. signifera*, *C. parinsignifera*, *Limnodynastes peronii*, *L. tasmaniensis*, *Litoria peronii*, *L. verreauxii* and *U. laevigata*. Five of these species were absent from 30% of the sites or more (see Table 3 for summary statistics). Pond type was found to be a useful explanatory variable for *L. tasmaniensis,* with chorus size significantly higher at natural ponds than farm dams (Table 4). There was no evidence of correlation in chorus size within dam/pond pairs.

3.5. Species richness model

Pond type was examined first in the species richness mixed model in the absence of all other variables. There was no significant increase in the amount of deviance explained by the model with the inclusion of pond type.

Table 4
Table of effects for mixed model of *Limnodynastes tasmaniensis* chorus data

Fixed effect	Estimate	SE	Change in deviance	$p(x^2_{t-1})$
Constant	1.33	0.22		
PT (n)	0.91	0.40	4.73	0.03

Standard error of difference is shown for the categorical variable pond type (PT).

Pond type was then examined in relation to all other fixed effects to determine if there were any relationships between species richness and habitat attributes dependent upon pond type. No such relationships were found. A significant relationship was found between species richness and percentage of the water margin with emergent vegetation (Table 5). The presence/absence of fish was also a significant explanatory variable (Table 5). Intra-dam/pond pair correlation in species richness values was low, estimated at 0.17. Fifty-two percent of the variation in species richness between dam/pond pairs was explained by the fixed effects (pond type, percent emergent vegetation cover at water margin and the

Table 3
Summary statistics for chorus index at dams and ponds

Species	Dam		Natural pond	
	Mean (SE)	*n*	Mean (SE)	*n*
Crinia parinsignifera	1.38 (0.28)	8	1.9 (0.7)	5
Crinia signifera	1.57 (0.15)	21	2.07 (0.21)	21
Limnodynastes tasmaniensis[a]	1.33 (0.19)	20	2.2 (0.4)	10
Limnodynastes peronii	1.13 (0.38)	4	1.36 (0.23)	14
Litoria peronii	0.89 (0.22)	9	0.78 (0.12)	9
Litoria verreauxii	1.56 (0.30)	17	1.65 (0.26)	17
Uperoleia laevigata	1.88 (0.21)	17	1.81 (0.33)	8

Sites with zero abundance have been removed.
[a] Indicates a significant difference was recorded for the mixed model of chorus index.

Table 5
Table of effects for frog species richness mixed model

Fixed effect	Estimate	Mean	SE	Change in deviance	$p(x^2_{t-1})$
Constant	5.28		0.29		
EV	0.02		0.01	4.46	0.04
FISH					
Present	−1.72	3.56	0.55	8.69	0.003
Absent		5.28			

Standard error of difference is shown for the categorical variable of fish presence (FISH). EV = percent emergent vegetation cover at waters edge. Mean shows mean species richness at sites where fish were recorded present and absent.

presence of fish). Fixed effects explained 5% of the variation in species richness within dam/pond pairs.

4. Discussion

4.1. Species response to pond type

Contrasting responses to pond type were found at the species level. *Limnodynastes peronii* appeared to show a preference for natural ponds while *U. laevigata* appeared to show a preference for farm dams. While *L. tasmaniensis* was also more commonly recorded at farm dams it is important to note that pond type was interchangeable with the presence of fish in the logistic model for this species. This suggests *L. tasmaniensis* is more likely to occur at waterbodies where no fish are present (regardless of their origin). In the case of *L. peronii* and *U. laevigata*, pond type remained a useful explanatory variable for species occurrence in the presence of all other explanatory variables. This suggests that these frog species are responding to differences between natural ponds and farm dams that were not captured by the variables analysed. It is interesting to note that during a survey of 75 waterbodies in 1998 *L. peronii* was recorded at four out of five semi-natural ponds surveyed, but only four out of 70 farm dams (see Hazell, 2001). This species is clearly more common in natural ponds than farm dams within the study area.

Results of this study suggest that there are a range of biotic and abiotic differences between the two pond types. Characteristics such as electrical conductivity, dissolved oxygen, availability of shallow water and tussock cover in the riparian zone all warrant further attention in understanding how frogs may respond to differences between natural and constructed waterbodies.

No preference for either pond type (on the basis of species occurrence and/or chorus size) was shown by *C. parinsignifera*, *L. dumerilii*, *L. verreauxii* or *C. signifera*. This result is not surprising, as these species use a broad range of breeding habitats (see Hazell et al., 2003). Results from this study show that farm dams are providing these species with habitat requirements that are adequate to attract adult frogs. Species that prefer farm dams, or show no preference for either pond type, are likely to have benefited from the widespread development of this pond type.

Two species recorded at natural ponds were absent from farm dams (*L. leseueri* and *L. phyllochroa*). These species are considered obligate stream-dwelling species (Gillespie and Hines, 1999), as opposed to pond-dwelling and were recorded at a chain-of-ponds and a stream-side pond respectively. Their absence from farm dams reflects differences in the dynamics and nature of farm dams and natural ponds. Chain-of-ponds systems occur in substantial drainage lines and experience far greater levels of flow (albeit periodically) than farm dams. In addition, farm dams are rarely located directly adjacent to streams as they are not required in these environments. These differences emphasise the importance of conserving natural ponds in maintaining a diversity of pond habitats.

On the basis of adult frog occurrence it appears that constructed ponds are capable of supporting the same number of species as natural ponds. However, results at the individual species level suggest that species composition will vary between the two pond types.

4.2. Fish predation

The presence of fish within a waterbody was a useful predictor of the occurrence of all four species modelled, as well as species richness (Tables 2 and 5). Fish can influence the composition of the tadpole community and reduce tadpole species richness (Woodward, 1983; Bradford, 1989; Werner and McPeek, 1994; Hero et al., 2001). There are several potential explanations for a fish/adult frog relationship in the current study. First, it is possible that species composition in the adult frog assemblage is determined by tadpole/predator relationships. In this situation tadpole/habitat dynamics may be more influential than adult frog/habitat dynamics in regulating frog populations. Adult habitat does however, require consideration in the current study as characteristics of terrestrial habitat have been shown to be useful explanatory variables for adult species occurrence and richness in the study area (see Hazell et al., 2001).

Second, adult frogs may be responding to the presence of fish. This is plausible, as choice of oviposition site has been shown to be influenced by the presence of both predators and competitors within a waterbody (Resetarits and Wilbur, 1989; Hopey and Petranka, 1994; Petranka et al., 1994; Holomuzki, 1995). Third, frogs may be responding to habitat characteristics that are correlated with the presence of fish, or frogs and fish may be responding to the same habitat characteristics. Experimental manipulation of fish predators would be required to determine which explanation is correct. It is important to note that there is limited published work in highly modified landscapes comparing species structure across tadpole and adult frog assemblages or the extent to which aquatic predators control frog assemblages in such environments.

4.3. Implications for frog conservation

The construction of waterbodies can have both positive and negative effects on biota at landscape and regional scales. In South Africa, Little and Crowe (1994) suggested that farm dams had formed a new biotope that complemented prefarming avian diversity. In the semi-arid rangelands of Australia, the introduction of surface water has threatened the persistence of a range of biota (James et al., 1999). There is no evidence that the construction of farm dams in south-eastern Australia has had a deleterious effect on pond-breeding frogs. However, habitat and frog assemblage differences between farm dams and natural ponds suggest that further loss of natural ponds will result in the simplification of pond environments available as frog habitat.

The results of this study suggest that waterbodies with high levels of emergent vegetation cover that do not contain fish are likely to support a high number of frog species, regardless of their origin (i.e. natural or constructed). However, it is important for landholders to conserve any natural waterbodies on their property as these environments are likely to support particular frog species that do not use farm dams. On the basis of these results, landholders who wish to maximise the value of pond habitat on their land should avoid releasing native fish in half of their farm dams. This will provide frogs with a mosaic of ponds and dams, with and without fish. Results suggest that farm dams built for agricultural purposes will provide suitable habitat for attracting a range of frog species. However, if landholders wish to construct a waterbody specifically for conservation purposes they should consider the benefits of temporary ponds in providing habitat that is free from predators such as fish.

Species occurrence provides an indication of what habitat is suitable and what is not. However, it is limited in its ability to indicate how suitable habitat is (i.e., habitat quality). Habitat quality is an important consideration in conservation decisions. Abundance data can assist in examining this issue. This is demonstrated by the results obtained for *L. tasmaniensis.* Farm dams have clearly provided additional habitat for this species (see Fig. 4). However, when the species was present at natural ponds it occurred in significantly higher numbers (Table 4). This suggests certain natural ponds provide the highest quality habitat (and thus conservation value) for this species. Further information on habitat quality would be provided through the collection of comparable abundance data at the adult and tadpole stage across many farm dams and natural ponds. Differences in habitat quality between farm dams and natural ponds may be particularly evident during extended dry periods when frogs only occupy highly favourable environments. Populations of *L. tasmaniensis* may contract back to certain natural ponds during such periods. A long-term study would be required to determine if such dynamics do indeed occur.

5. Conclusions

These results suggest that both natural and constructed ponds have a conservation role in providing habitat for frogs and conserving heterogeneity in waterbody characteristics across the landscape. Differences in habitat quality require further investigation, particularly in relation to population recruitment, which was not examined by the current study. This issue requires collection of detailed frog data and would be more suitably undertaken on a smaller sub-set of frog species.

Conservation recommendations developed from this research have been integrated into a publication specifically designed for assisting landholders with wildlife conservation (Lindenmayer et al., 2003). This is considered as a vital endpoint to ecological research on private land to maximise the chances of such research actually achieving on-ground conservation outcomes.

Acknowledgements

This research was undertaken as part of a PhD at the Australian National University and was made possible through an Australian Postgraduate Award and funds and resources provided by CSIRO Sustainable Ecosystems and the Centre for Resource and Environmental Studies, ANU. Encouragement and assistance with project development was kindly provided by Brendan Mackey, Denis Saunders, Will Osborne, Ian White and Henry Nix. John Stein assisted greatly with the map. Thanks also to Christine Donnelly, NSW Fisheries, NSW Department of Land and Water Conservation, Sydney Catchment Authority and the Braidwood Rural Lands Protection Board. The manuscript was greatly improved by comments made by Dale Roberts and Andrew Hull. This research would not have been possible without the enthusiasm and interest shown by private landholders of the upper Shoalhaven catchment.

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