Reducing weed biomass by burning and grazing can adversely affect frogs

Deborah S. Bower[1], Leonie E. Valentine[2], Anthony C. Grice[3] and Lin Schwarzkopf[2]

[1] Institute for Applied Ecology, University of Canberra, Australian Capital Territory 2601, Australia
[2] School of Tropical Biology, James Cook University, Queensland 4811, Australia
[3] Sustainable Ecosystems, Davies Laboratory, CSIRO, Queensland 4811, Australia

Summary The impacts of weed management on native biodiversity are often overlooked. Burning and grazing plots of land, in isolation and in combination, were used to experimentally reduce the biomass of introduced para grass (*Urochloa mutica* (Forssk.) T.Q.Nguyen)) in a North Queensland wetland. Frogs were monitored to assess the impact of these management trials. Marbled frogs (*Limnodynastes convexiusculus* Macleay) declined in response to all management treatments, and their abundance was correlated with vegetation biomass. The abundance of spotted marsh frogs (*Limnodynastes tasmaniensis* Günther) was not related to weed control treatments, but was influenced by the distance of the experimental plot from the nearest woodland. The decline of these frog species in response to management trials indicates that knowledge about impacts of planned weed control is critical, to inform management of taxa that may be affected.

Keywords Weed, para grass, *Urochloa mutica*, amphibians, frogs, fire, grazing, wetland management.

INTRODUCTION

Few studies have examined the influence of weed control measures on biodiversity and assemblage composition, as weed removal is generally considered beneficial for invaded areas (Zavaleta *et al.* 2001). However, there can be several potential flow-on effects to faunal assemblages in the wake of large-scale weed control (Zavaleta *et al.* 2001). First, and most obviously, successful weed control removes weeds. Although weeds typically have a negative effect on biodiversity once established (Sakai *et al.* 2001), the process of weed removal can have inadvertent and undesirable impacts in itself (Zavaleta *et al.* 2001). Secondly, control measures often change physical properties of an area, such as vegetation structure and growth form, altering microhabitat and shelter (Williams *et al.* 2005). Changes to the habitat structure following weed removal may cause a shift in the faunal assemblage after the control measure is applied. In order to make informed management decisions about weed removal, we must be able to quantify the response of faunal assemblages to weed control and understand which species are successful in the presence of the weed.

In Australia, para grass is an exotic weed, capable of destroying ecosystem structure and function (Humphries *et al.* 1991). Introduced to Australia from Africa in the late 1800s, to increase pasture productivity (Guenni *et al.* 2002), para grass has since spread outside grazed areas, where it monopolises large areas of land (Humphries *et al.* 1991). Para grass reduces biodiversity by out competing complex mosaics of native vegetation (Ferdinands *et al*. 2005). The invasion of para grass into native habitat drastically changes composition and structure of the habitat and increases fuel load and intensity of fires (Humphries *et al.* 1991). Such changes in vegetation composition and ecosystem function may influence fauna that used the habitat prior to weed invasion.

Fire and grazing were selected for reduction of para grass because they were readily available, cost-effective management options that reduce vegetation biomass. Previous studies have found that grazing reduces biomass and cover of para grass, but the remaining mats of trampled para grass prevent the recruitment and growth of native species (Williams *et al.* 2005). While fire also reduces biomass, the patchy nature of burns often leaves large areas of standing para grass. Using a combination of burning and grazing may decrease the extent of para grass; controlling the invader more effectively while still allowing native plant species to regenerate (Williams *et al*. 2005).

To study the influence of weed control measures on fauna, managers require faunal groups whose sensitivity to environmental change can be detected. Frogs are considered useful indicators of environmental health, although some studies have had difficulties attributing population responses to specific causal factors (Bamford 1992). We examined responses of the frog assemblage to changes in vegetation structure caused by burning and grazing to investigate the influence of weed control methods on fauna that inhabit para grass.

MATERIALS AND METHODS

Study sites were located at a seasonal wetland, the Town Common Conservation Park, Townsville, North Queensland (19°12'28.69"S, 146°44'25.19"E). Originally common grazing land, the Town Common became

a conservation reserve in 1985, due to the high species richness of water birds. Para grass was first introduced to the Town Common as a fodder for cattle over 30 years ago, and is now the dominant vegetation on the flood plain. Prior to the introduction of para grass, the flood plains were probably dominated by salt-water couch (*Paspalum distichum* L.), native sedges (*Eleocharis dulcis* Hensch. and *Cyperus scariosus* R.Br.) and grasses (*Leersia hexandra* Sw., P. Williams pers. comm.).

Experimental design   Para grass-dominated habitat was divided into 12 plots, 200 × 300 m each. Treatments were randomly assigned to the plots. Experimental treatments, each replicated three times, included plots that were: burnt and grazed, burnt only, grazed only, and neither burnt nor grazed as controls. Plots exposed to fire were burnt at the end of August 2004. Grazed plots were intensely grazed by cattle, *Bos indicus* (Linnaeus), for three months from December 2004. A model predicting required grazing intensity for 50% of grass to be removed, determined the number of cattle allocated to each plot. The model assumed that one cow ate 9 kg of forage per day.

Amphibian sampling   Faunal sampling was conducted between 28 July to 29 August 2005, 12 months following burning and eight months following the removal of cattle. Frog abundance was surveyed using pitfall traps that were open for 21 days. Each plot contained three trap arrays, consisting of four pitfall traps each. The data collected from all traps within a plot were pooled in analyses. Pitfall traps were 20 L plastic buckets with the lip of the opening level with the ground. Arrays were located 50 m apart along the long axis of the plot. In each plot the centre array had 1 mm mesh drift fence 50 cm high, the other two arrays had 50% shade cloth drift fence 25 cm high. Traps were checked daily and captured frogs were identified and tagged using a treatment-specific mark by clipping a small notch of skin off one hind toe before release.

Vegetation sampling   Vegetation biomass and composition were estimated using BOTANAL. Fifty 1 m^2 quadrats were spaced along two parallel transects on the long axis of each plot. Calibration relationships were developed by estimating biomass in nine 1 m^2 quadrats, harvesting, drying, and weighing all herbage in them. This technique provided biomass (kg ha^{-1}), on a plot-by-plot basis. The distance from the centre of each plot to the nearest woodland was also recorded.

Statistical analysis   Overall abundance of frogs and the abundance of each frog species were compared between treatments using two-way fully orthogonal

ANOVAs (SPSS, v.11) with burning and grazing as factors. This design examines the impacts of burning and grazing and any interaction between the two factors. Overall frog abundance and the abundance of each species were correlated with vegetation variables and distance to the nearest woodland using Spearman's Rank Correlation Coefficient (SPSS, v.11). Count data was examined for homogeneity and normality using Levene's tests, and visual examination of boxplots and residual plots. All abundance data were square-root transformed to meet assumptions of normality. The distribution of vegetation data recorded as percentages were normalised by arc-sine transformation of the proportional data.

RESULTS

A total of 450 individual frogs from four species were trapped during the survey period. Overall frog abundance did not significantly differ between treatments. Frog abundance was dominated by *L. tasmaniensis* (64% of all frogs captured), and *L. convexiusculus* (28% of all frogs captured). Although there were no significant differences between factors in the abundance of *L. tasmaniensis*, both burning (ANOVA F$_{1,8}$ = 3.225, P = 0.001) and grazing (ANOVA F$_{1,8}$ = 19.436, P = 0.002) significantly reduced the abundance of *Limnodynastes convexiusculus* (Figure 1). There was also a significant interaction between the burning and grazing (ANOVA F$_{1,8}$ = 8.449, P = 0.020), indicating that the application of either treatment reduced the abundance of *L. convexiusculus* (Figure 1). In addition, there was a trend for slightly higher abundances of *L. convexiusculus* in burnt-only compared to grazed only treatments (Figure 1). The abundance of the other frog species, *Crinia deserticola* Liem and Ingram and *Litoria nastua* Gray, showed no significant differences between treatments.
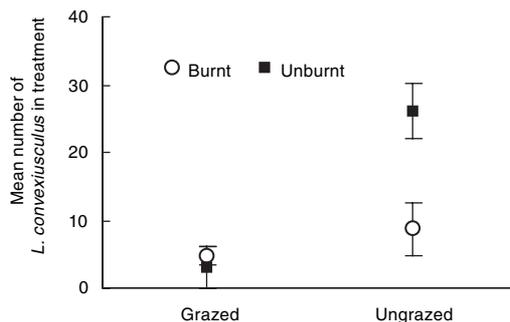
Figure 1.   Influence of burning grazed and ungrazed para grass on *Limnodynastes convexiusculus* abundance (±95% CI).

Frog abundance and vegetation biomass correlations Overall frog abundance did not correlate with vegetation biomass. However, the abundance of one frog species, *L. convexiusculus* was positively correlated with plant biomass (Spearman's rank, rho = 0.637, P = 0.026, r^2 = 0.53; Figure 2).

Distance to nearest woodland Distance to the nearest woodland varied among plots but was not significantly different among treatments. Abundance of *L. tasmaniensis*, the dominant frog species, was negatively correlated with distance from woodland (Spearman's rank, rho = −0.598, P = 0.04, r^2 = 49.98; Figure 3); these frogs occurred in higher abundances in plots adjacent to woodland habitat.

DISCUSSION

Frog responses to weed reduction treatments In our study burning and grazing did not significantly change the overall abundance of frogs. However, the abundance of *L. convexiusculus*, the second most abundant frog species, was reduced by both weed management techniques. Although the impact of grazing on amphibians is poorly documented, previous research indicates that frogs may respond negatively to grazing (Jansen and Healy 2003). In contrast, previous studies have observed an increase (Bamford 1992), decrease (Masters 1996) and no change (Bamford 1992) in frog abundance in response to burning. Even though these studies document disparate responses of frogs to fire, they suggest that the removal of ground cover may be an important factor influencing habitat use by frogs. As the abundance of *L. convexiusculus* was positively correlated with vegetation biomass, our results suggest that *L. convexiusculus* are adversely affected the removal of vegetation cover caused by burning and grazing.

Limnodynastes convexiusculus are ground-dwelling myobatrachids, which probably use vegetation for finding food and shelter, and the removal of vegetation may alter the availability of these resources. Indeed, burning may initially remove invertebrates, which are prey for many species of reptile and amphibians (Bamford 1992). Although, in our study, grazing appeared to reduce *L. convexiusculus* numbers to even lower numbers than burning. As fires typically burn in a mosaic fashion, frogs may seek shelter in remaining unburnt patches of vegetation. In contrast, grazing uniformly reduced and compacted vegetation in plots (Bower personal observation), leaving little opportunity for shelter. Further, the negative impacts of grazing on frogs may increase with grazing intensity (Jansen and Healy 2003).
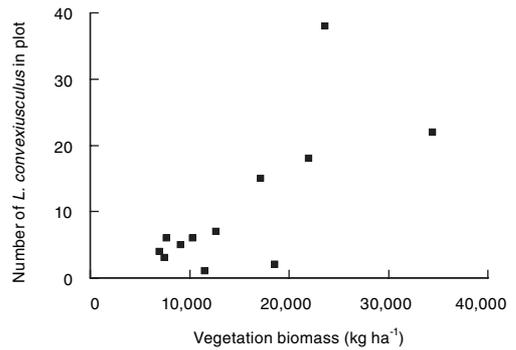
Figure 2. Response of *Limnodynastes convexiusculus* abundance in experimental plots with varying vegetation biomass.
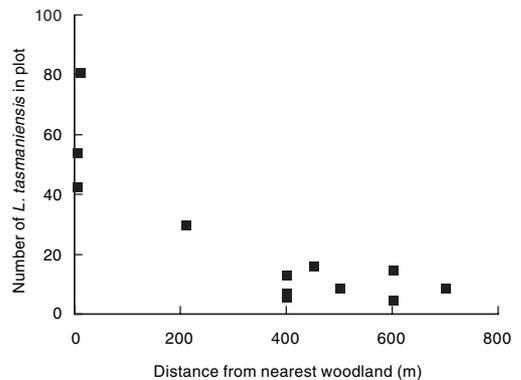
Figure 3. Response of *Limnodynastes tasmaniensis* abundance in experimental plots with varying distances from woodland habitat.

Frog relationships with plot characteristics *Limnodynastes tasmaniensis* accounted for more than half of the total frog captures, and this species did not respond to the management treatments applied to para grass. Instead, *L. tasmaniensis* abundance was related to the sampling distance from the woodland; frog abundance decreased with increasing distance from the woodland habitat. Such a relationship suggests that para grass is suboptimal habitat for this species, even though they can survive in para grass in low densities. In contrast, the strong positive relationship between *L. convexiusculus* and vegetation biomass suggests that a complex vegetation structure may be very important to this frog's ecology.

Implications for weed management Although amphibians are not always successfully used to quantify ecosystem responses to manipulation (Bamford 1992), our study indicates that some species of frogs can be useful indicators of disturbance. However, closely related frog species showed very different responses to the disturbances applied in these treatments.

The decline of some frogs during the management of para grass, illustrates the importance of setting priorities when managing habitats. The reduction of some frog species using the area may be undesirable, desirable, or neutral to management goals. In this case, a drastic reduction in population sizes of *L. convexiusculus*, a relatively common species in other, surrounding habitats, may be an acceptable consequence of controlling the para grass. However, if this frog species played a critical role in the ecosystem function of wetlands, unrelated to the para grass infestation, then a reduction in this species may be more damaging. Thus, a decline of native fauna adapted to live in weedy habitats could be a consequence of weed removal, and the consequences of that decline must be weighed against the management goals of weed removal in every case. In any case, knowledge of the wider implications of reducing para grass cover is valuable to management agencies setting priorities for conserving wetlands.

REFERENCES
Bamford, M.J. (1992). The impact of fire and increasing time after fire upon *Heleioporus eyrie*, *Limnodynastes dorsalis* and *Myobatrachus gouldii* (Anura: Leptodactylidae) in *Banksia* woodland near Perth, Western Australia. *Wildlife Research* 19, 169-78.

Ferdinands, K., Beggs, K. and Whitehead, P. (2005). Biodiversity and invasive grass species: multiple-use or monoculture? *Wildlife Research* 32, 447-57.

Guenni, O., Marin, D. and Baruch, Z. (2002). Responses to drought of five *Brachiaria* species. I. Biomass production, leaf growth, root distribution, water use and forage quality. *Plant and Soil* 243, 229-41.

Humphries, S.E., Groves, R.H. and Mitchell, D.S. (1991). Plant invasions of Australian ecosystems: a status review and management directions. Report to the Australian National Parks and Wildlife Service, Endangered Species Program and CSIRO. *Kowari* 2 (Canberra).

Jansen, A. and Healy, M. (2003). Frog communities and wetland condition: relationships with grazing by domestic livestock along an Australian floodplain river. *Biological Conservation* 109, 207-19.

Masters, P. (1996). The effects of fire-driven succession on reptiles in spinifex grasslands. *Wildlife Research* 23, 39-48.

Sakai, A.K., Allendorf, F.W., Holt, J.S., Lodge, D.M., Molofsky, J., With, K.A., Baughman, S., Cabin, R.J., Cohen, J.E., Ellstrand, N.C., McCauley, D.E., O'Neil, P., Parker, I.M., Thompson, J.N. and Weller, S.G. (2001). The population biology of invasive species. *Annual Review of Ecology and Systematics* 32, 305-32.

Williams, P.R., Collins, E.M. and Grice, A.C. (2005). Cattle grazing for para grass management in a mixed species wetland of north-eastern Australia. *Ecological Management and Restoration* 6, 75-6.

Zavaleta, E.S., Hobbs, R.J. and Mooney, H.A. (2001). Viewing invasive species removal in a whole ecosystem context. *Trends in Ecology and Evolution* 16, 454-9.