

## Research Article

# Frog Communities in Fire-Disturbed Forests of the Peruvian Amazon

ELEANOR WARREN-THOMAS<sup>1</sup>, MARY MENTON<sup>1,2</sup>, JUSMELL HUAMÁN<sup>3</sup>, RUTH FRISANCHO VARGAS<sup>3</sup>, EMMA WADLEY<sup>1</sup>, NICHOLAS PRICE<sup>1</sup> AND JAN CHRISTOPH AXMACHER<sup>4</sup>

<sup>1</sup>*University of Oxford, Department of Plant Sciences, South Parks Road, Oxford, OX1 3RB.*

<sup>2</sup>*CIFOR, CIP, Apartado 1558, Lima 12, Perú.*

<sup>3</sup>*Universidad Nacional Amazónica de Madre de Dios, Facultad de Ingeniería y Facultad de Ecoturismo, Madre de Dios, Perú.*

<sup>4</sup>*UCL Department of Geography, University College London, Pearson Building, Gower Street, London WC1E 6BT, UK.*

E-mail address: <sup>1</sup>*em.warren.thomas@gmail.com*

**ABSTRACT** - Amphibian declines are a pressing global concern. The lowland rainforests of the south-eastern Amazon harbour exceptionally high amphibian diversity, but also face a range of threats including habitat modification caused by forest fires. In this study, we sampled amphibians in areas of forest in Madre de Dios, Peru, that were affected by anthropogenic fires following severe drought in 2005. Two forest types, bamboo and terra-firme, were assessed. Forty-two anuran species were recorded in 22 survey nights. Amphibian diversity and abundance were not significantly different in burned areas of either forest type, and amphibian community composition did not change significantly between burned and unburned forests within any forest type, while bamboo forest was found to support a distinctly different amphibian assemblage to terra-firme forest. Our results suggest that further sampling over wider spatial and temporal scales to encompass a greater range of fire impacts could consolidate insights into the effects of fire on anuran communities in this region, and help to highlight the conservation value of these disturbed forests. These preliminary results are novel and enhance our understanding of how tropical forest fires may affect amphibian communities. The data also highlight the conservation value of forests affected by a fire event, as they harboured large numbers of anuran species known in the region. This is of particular interest for those species that have so far only been recorded outside of protected areas in Madre de Dios, such as *Ranitomeya cf. ventrimaculata* and *Osteocephalus buckleyi*.

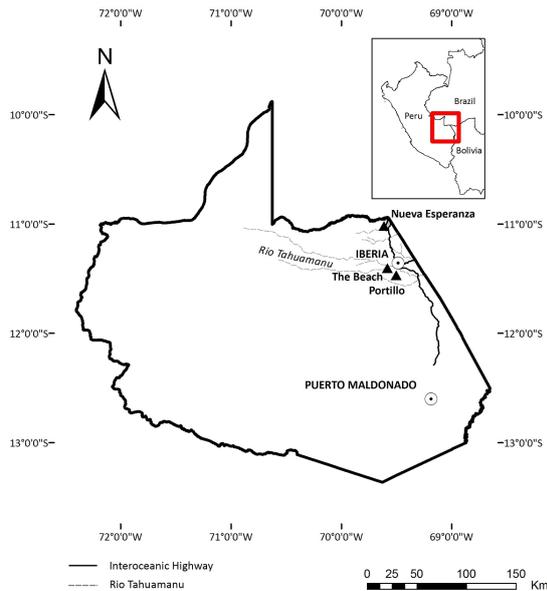
## INTRODUCTION

Amphibian declines are a pressing global problem, with neotropical species declining particularly rapidly (Stuart et al., 2004). Habitat disturbance, pollution and disease are considered major drivers of these declines (Beebee & Griffiths, 2005; IUCN, 2005; Wake & Vredenburg, 2008), with habitat loss and degradation thought to be affecting almost 4000 amphibian species worldwide (IUCN et al., 2008). Therefore, improving understanding of how human perturbation of tropical forests

affects amphibian communities has been identified as a priority by the IUCN Amphibian Conservation Action Plan (IUCN, 2005).

As an area of extremely high biological diversity, the south-western Amazon was listed as one of the original 25 global 'biodiversity hotspots' (Myers et al., 2000). In the Peruvian department of Madre de Dios that lies at the south-western edge of the Amazon Basin, this diversity includes an exceptionally high number of amphibian and reptile species. While the department represents only 1% of the Amazon

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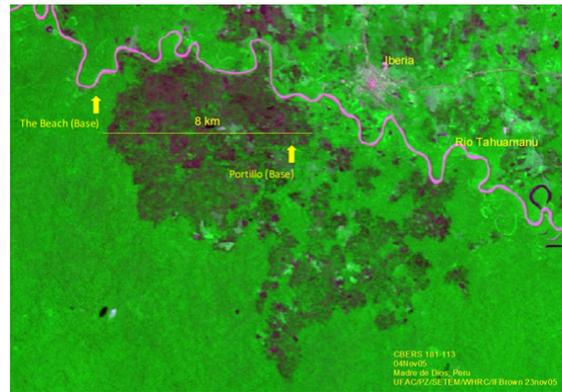
**Figure 1.** Study area in context of tri-national border (inset) and study sites in relation to Inter-Oceanic Highway (shown on map as completed in 2009).

Basin land area, it supports an estimated 30% of the known Amazonian herpetofauna, including 114 amphibian species (Duellman, 2005; von May et al., 2008).

The forests of Madre de Dios, which covered 94% of its area in 2011, are under threat of loss through clearance for settlements and agriculture (Oliveira et al., 2007) and artisanal gold mining, during which large areas of pristine primary forest are cleared and mercury is released into waterways and sediments (Swenson et al., 2011). Forest degradation also affects large areas, for example through selective logging and fire. Habitat disturbance by fire is thought to have a particularly strong effect on tropical forest biodiversity (Barlow & Silveira, 2009). It is currently emerging as an increasingly prevalent threat to moist tropical lowland rainforests (Cochrane, 2003; Barlow & Peres, 2008).

In closed-canopy rainforest, fire results in a dramatic alteration of forest structure within 1-3 years with widespread mortality of large trees, upper canopy thinning and increases in bamboo stems and pioneer tree species (Barlow & Peres, 2008). These changes are expected to affect amphibian communities via alterations to habitat structure, microclimate and vegetation species composition (Crump, 1994; Urbina-Cardona et al., 2006).

In 2005 a severe drought affected the whole



**Figure 2.** China-Brazil Earth Resources Satellite Program (CBERS) image of burned forest, showing dark purple against green primary forest in colour reproduction. The Rio Tahuamanu and urban areas show pink.

of Amazonia (Marengo et al., 2008; Zeng et al., 2008). In Madre de Dios, this was characterised by a virtual absence of rainfall for several months (Brown et al., 2006). This coincided with the building of a new highway in the south-western Amazon, where the increase in human activity (Perz et al., 2011; Southworth et al., 2011) led to accidental transitions of fires from settled areas into the unusually dry adjacent primary rainforests (Brown et al., 2006). This stimulated unprecedented forest burning across the region, affecting more than 467,000 ha of forest and open areas in Acre (Vasconcelos & Brown, 2007) and at least 10,000 ha of forest in Madre de Dios (Brown et al., 2006).

Relatively few studies have previously assessed the consequences of fire for amphibian communities in moist lowland tropical forests. Fredericksen & Fredericksen (2002) studied a tropical forest landscape in Bolivia disturbed by both fire and logging, and chiefly focussed on the impact of logging gaps, finding only four amphibian species. Other studies in South America directly assessing the impact of fire have focussed on open areas surrounded by forest (Papp & Papp, 2000), restinga forests along the Atlantic Coast (Rocha & Ariani, 2008) or savannah forests where fire is a natural part of the ecosystem (Cano & Leynaud, 2009).

The aim of this study was therefore to make a preliminary assessment of anuran species richness, abundance and community composition in forests affected by fire during the 2005 drought, and to compare these data to

Forest type	Study site name	Burn status	Number of transects
Bamboo	The Beach	Burned	5
	The Beach	Unburned	5
Terra-firme	Nueva Esperanza	Burned	2
	Nueva Esperanza	Unburned	4
	Portillo	Burned	4
	Portillo	Unburned	2

**Table 1.** Transects surveyed in each forest type, showing burn status and location of transects. Two localities were sampled in terra-firme forest, the results of which were pooled in the analysis (Nueva Esperanza and Portillo). One locality was sampled in bamboo forest (The Beach).

nearby non-impacted forests.

## METHODS

### Site description

The study sites lie close to the Inter-Oceanic Highway in Madre de Dios, south-eastern Peru (Figure 1), paving of which began in 2006 (Southworth *et al.*, 2011). Significant migration of people to the area in advance of the road paving led to some of the greatest extent of fire damage occurring here in 2005, associated with settlement activity (Perz *et al.*, 2011; Southworth *et al.*, 2011).

Four major forest types found across the south-western Amazon were present: terra-firme forest, floodplain forest, bamboo forest and palm swamp (Pitman *et al.*, 1999). These forest types are strong predictors of amphibian assemblages, with a high proportion of the regional amphibian diversity attributable to a high beta-diversity across the forest types (von May *et al.*, 2010). The overall amphibian species pool of Madre de Dios is well-documented through a number of long-term studies, and species lists are available for a number of localities in nearby protected areas (Doan & Arriaga, 2002; Duellman, 2005; von May *et al.*, 2010).

Meteorological records from the Los Amigos Biological Station (CICRA) approximately 140 km from the study site indicate that annual precipitation ranges between 2700 and 3000 mm, with a mean annual temperature between 21°C and 26°C (von May *et al.*, 2010). A distinct period of reduced rainfall with slightly cooler temperatures occurs between June and September. This dry season is also characterised

by periodic cold weather events driven by southerly winds from Patagonia known as *friajes*. During these events, windspeeds are elevated and air temperatures can drop as low as 5°C at night.

### Study sites

Images captured by the China-Brazil Earth Resources Satellites (CBERS) in 2005 were used to identify areas of burned forest (Figure 2) and to select three study sites located outside protected areas. One study site was known to have been well established bamboo forest before the fires, with the other two study sites located within closed-canopy terra-firme forest.

Terra-firme forests are characterised by high (>30 m) closed canopies with many emergent trees and high tree species diversity. In contrast, bamboo forests are dominated by *Guadua sarcocarpa* and *G. weberbaueri*, rising to ~25 m, with reduced tree species diversity and fewer high-canopy trees (Griscom & Ashton, 2006; Griscom *et al.*, 2007). These forests form where disturbance events in the past opened the canopy, allowing invasion of bamboo and formation of a self-perpetuating cycle of tree damage and bamboo regrowth (Griscom & Ashton, 2003; Griscom & Ashton, 2006).

Sites were chosen near the boundary between large continuous burned areas and unburned areas in order to allow pairing of sampling sites with minimal micro-geographic variation. The first terra-firme site 'Nueva Esperanza' (11° 1'13.07"S, 69°36'38.72"W), was located approximately 4 km from the highway. The second terra-firme site, 'Portillo' (11°28'6.60"S, 69°30'2.66"W), was located 6 km south of the

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Species	Bamboo		Terra Firme		Sampling method
	Burn	Unburn	Burn	Unburn	
<i>Allobates conspicuus</i>	0	0	0	1	O
<i>Ameerega hahneli</i>	1	6	1	0	T
<i>Ameerega petersi</i>	0	1	0	0	T
<i>Ameerega trivittata</i>	0	0	4	2	Both
<i>Chiasmocleis bassleri</i>	0	0	1	1	Both
<i>Chiasmocleis ventrimaculata</i>	1	0	0	0	T
<i>Teratohyla midas</i>	0	1	0	0	O
<i>Dendropsophus acreanus</i>	0	0	0	1	O
<i>Dendropsophus allenorum</i>	1	0	0	0	T
<i>Dendropsophus koechlini</i>	1	2	0	0	T
<i>Dendropsophus leali</i>	0	2	0	0	T
<i>Dendropsophus leucophyllatus</i>	0	1	0	1	T
<i>Dendropsophus parviceps</i>	0	5	1	3	T
<i>Dendropsophus schubarti</i>	2	3	0	0	T
<i>Dendropsophus xapuriensis</i>	0	0	0	1	T
<i>Elachistocleis bicolor</i>	6	1	1	0	Both
<i>Eleutherodactylus sp.</i>	1	0	0	0	T
<i>Engystomops petersi</i>	0	0	0	1	O
<i>Hamptophryne boliviana</i>	0	9	0	0	T
<i>Hypsiboas fasciatus</i>	31	37	3	1	Both
<i>Hypsiboas lanciformis</i>	0	0	1	0	O
<i>Hypsiboas sp.</i>	0	1	0	0	T
<i>Leptodactylus andreae</i>	4	0	0	10	Both
<i>Leptodactylus cf. petersii</i>	0	1	0	0	T
<i>Leptodactylus lineatus</i>	0	0	2	0	T
<i>Leptodactylus rhodonotus</i>	0	1	0	0	O
<i>Leptodactylus sp.</i>	1	0	0	0	T
<i>Oreobates quixensis</i>	0	0	1	0	T
<i>Osteocephalus cabrerai</i>	0	0	0	1	T
<i>Osteocephalus leprieurii</i>	0	0	3	1	T
<i>Osteocephalus sp.</i>	0	0	4	2	Both
<i>Phyllomedusa palliata</i>	1	2	0	1	T
<i>Pristimantis peruvianus</i>	1	3	6	17	Both
<i>Ranitomeya cf ventrimaculata</i>	0	0	0	3	T
<i>Rhinella margaritifera</i>	0	5	3	0	T
<i>Scarthyla cf. goinorum</i>	1	0	0	0	O
<i>Scinax garbei</i>	3	2	0	0	T
<i>Scinax ictericus</i>	16	15	1	1	Both
<i>Scinax pedromedinae</i>	0	1	0	0	T
<i>Scinax ruber</i>	1	1	0	0	T
<i>Trachycephalus venulosus</i>	1	0	0	0	O

**Previous page: Table 2.** Anuran species list, showing number of individuals recorded in each forest type. Sampling methods: O = opportunistic, T = transect, Both = individuals found both on transects and opportunistically.

town Iberia on the Inter-Oceanic Highway (5km south of the Tahuamanu River). The bamboo-dominated site ‘The Beach’ (11°24’14.14”S, 69°34’50.79”W) was situated next to the Tahuamanu River 9 km east of Iberia (Figure 1).

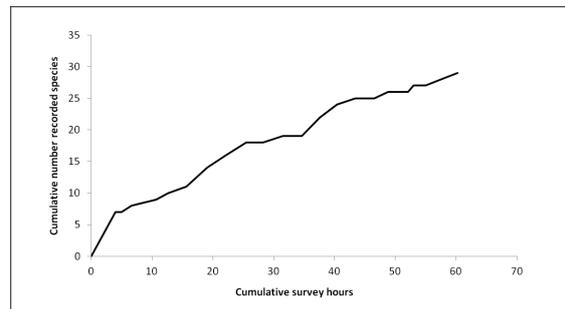
### Study plots

We set up randomly oriented 250 m transects at 100 m intervals along a main path at all sites. In the bamboo forest site, five transects were cut in burned areas and five in unburned areas. At the two terra-firme forest sites, two transects were cut in burned areas and four in unburned at Nueva Esperanza, and four in burned and two in unburned at Portillo. Data from these two terra-firme sites were combined in the analysis to give six transects in burned forest and six transects in unburned forest (Table 1).

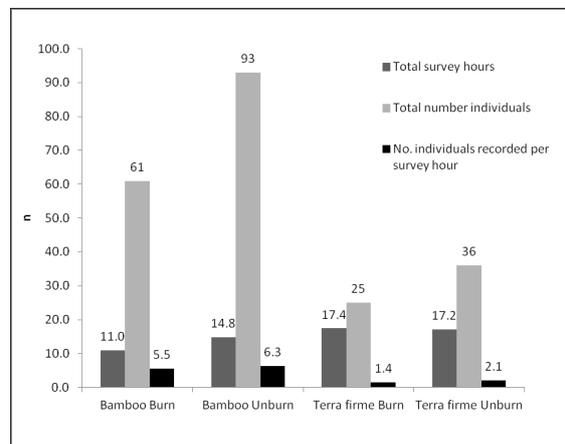
Transects were cut just wide enough to allow passage through the undergrowth and were left undisturbed for two days after cutting to allow any disturbed amphibians to return to the area.

### Sampling

Each transect was surveyed once using a Visual Encounter Survey (VES) between 18:00 h and 00:00 h according to the protocol described by Doan (2003). Sampling was completed over 22 nights in July and August 2009. Night time VES were chosen because comparative studies have found this method to be best for detecting inter-site variation in amphibian species richness (Pearman *et al.*, 1995), encompassing unique species (Doan, 2003) and finding the greatest amphibian abundance and species richness in short-term studies (Rödel & Ernst, 2004; Donnelly *et al.*, 2005). Two to four surveyors completed each transect, all with comparable surveying experience. The length of time spent searching each transect was recorded, allowing expression of abundance as a product of search effort. It cannot be assumed that the abundances recorded represent actual abundance and abundance is therefore defined as ‘estimated activity abundance’ (Pearman *et al.*, 1995). Amphibians that were encountered opportunistically, for example on the way to



**Figure 3.** Species accumulation curve, showing the number of cumulative species recorded with cumulative survey hours. The curve shows no sign of stabilisation.



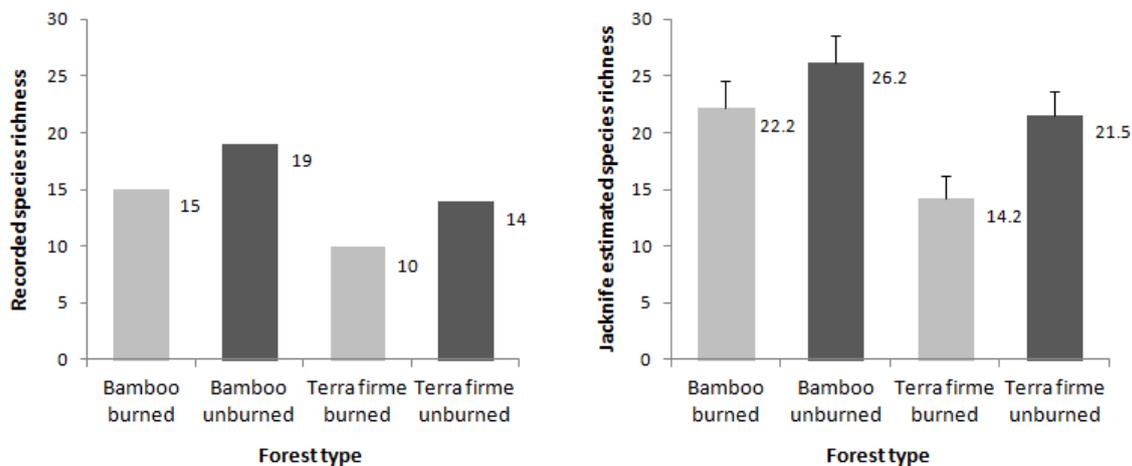
**Figure 4.** Anuran abundance. Three bars represent: number of survey hours in each forest type, number of individuals encountered in each forest type (sum of all surveys), and the number of individuals recorded by survey hour.

transects, were also identified to add to the overall species list, but they were not included in any statistical analysis.

Individuals were either identified immediately or retained until the following morning, when they were photographed and identified using a number of published field guides (Bartlett & Bartlett, 2003; Knell *et al.*, 2004; Duellman, 2005; von May *et al.*, 2007; Barbosa de Souza, 2009) before being released. No voucher specimens were taken.

Tree density was recorded along the same 250 m transects during daylight hours after the completion of amphibian surveys. All living and standing dead trees >10 cm DBH were recorded within 5 m on either side of transects. Sapling stem density and bamboo data were

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**Figure 5.** Anuran species richness. Recorded species richness is shown in the left panel, and Jackknife estimated species richness in the right panel, for each forest type. Error bar on Jackknife estimated species richness represents the calculated standard deviation.

collected within two 50 m x 1 m plots at the start and end of each transect. Within each plot, all tree saplings with <10 cm DBH (or diameter <10 cm immediately below branching if between 1 m and 1.3 m high) and height  $\geq 1$  m were recorded, with bamboo stems within this size class recorded separately.

### Data analysis

Comparisons of species richness between sites do not incorporate key information about species abundances or sample completeness patterns. Therefore, species diversity indices taking into account both species richness and relative abundances or estimators of true species richness are useful. Due to the low frequency of many species in the data, the use of diversity indices was difficult, but the data did allow meaningful calculations of non-parametric species richness estimators. These indicators approximate the total species richness of the site, had sampling continued until no additional species were found, based on the number of singletons and doubletons in the sample. The Jackknife estimator (Jack 1) was selected and calculated using EstimateS 8.2 (Colwell, 2009). Species accumulation curves were also created to assess sample completeness.

### RESULTS

Over the 22 sampling nights, 254 anuran specimens comprising 33 species were sampled

on the transects. A further 39 individuals comprising 17 species were caught opportunistically, including nine additional species that were not recorded during VES on transects (Table 2). Fourteen species were only found in unburned forests, while nine were only recorded in burned forest. A species accumulation curve for all anurans collected on transects does not show any levelling off, indicating that sampling was incomplete (Figure 3).

Significant reductions in large tree density were recorded in burned areas of terra-firme forest (Mann Whitney tests: Portillo  $w = 10$ ,  $p = 0.01$ ; Nueva Esperanza  $w = 6$ ,  $p = 0.05$ ), while there was no significant difference in bamboo forest (Mann Whitney test  $w = 12.5$ ,  $p = 0.07$ ). The density of standing dead trees was higher in burned areas at all sites.

The total number of recorded anuran individuals (burned and unburned data) was higher in bamboo forest than in terra-firme forest (Mann-Whitney  $w = 79$ ,  $p = <0.001$ ) (Figure 4). Within both forest types, more individuals were recorded in unburned areas than burned areas (Figure 4), but these differences were not statistically significant (bamboo: Mann-Whitney  $w = 27$ ,  $p = 0.92$ ; terra-firme: Mann-Whitney  $w = 32.5$ ,  $p = 0.31$ ).

Recorded anuran species richness was higher in unburned areas of both forest types (Figure 5), but again, differences were non-significant



a) *Rhinella margaritifera* (Bufonidae)



b) *Ranitomeya cf. ventrimaculata* (Dendrobatidae)



c) *Pristimantis peruvianus* (Strabomantidae)



d) *Dendropsophus leucophyllatus* (Hylidae)



e) *Leptodactylus rhodonotus* (Leptodactylidae)



f) *Chiasmocleis bassleri* (Microhylidae)

**Figure 6.** Anuran families recorded during sampling with representative species: a) Bufonidae – the true toads b) Dendrobatidae – poison dart frogs c) Strabomantidae – diverse group, all species thought to be direct-developing d) Hylidae – tree frogs and allies e) Leptodactylidae and f) Microhylidae – large and diverse group containing many different guilds.

(bamboo Mann-Whitney  $w = 25$ ,  $p = 0.25$ ; terra-firme Mann-Whitney  $w = 29$ ,  $p = 0.40$ ). No correlation could be found between recorded or estimated anuran species richness and any of the measured vegetation variables. In relation to anuran family spectra, the vast majority of

individuals sampled in bamboo forest were from the Hylidae (75%), while in terra-firme forest, Hylidae comprised 38%, while members of other families like Strabomantidae (22%), Leptodactylidae (17%), Dendrobatidae (16%) and Bufonidae (5%) were also strongly present

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(Table 2). These families contain species with strongly varying life history characteristics (Figure 6).

*Pristimantis peruvianus* (Strabomantidae) was found in higher numbers in terra-firme forest (n = 15) compared to bamboo forest (n = 4), while the two Hylidae species *Hypsiboas fasciatus* (n = 61 in bamboo forest, n = 4 in terra-firme) and *Scinax ictericus* (n = 26 in bamboo forest, n = 1 in terra-firme) were much more abundant in the bamboo forest. These two species represented a substantial proportion of the total specimens in the bamboo forest and contributed considerably to the difference in species composition between the forest types (Table 2).

### DISCUSSION

While burned areas of both bamboo and terra-firme forest appeared to support lower anuran species richness and abundance, these differences proved not statistically significant. This may be due to the short sampling period, as species accumulation curves and Jackknife estimations of the true species richness suggested that sampling was incomplete in both forest types. Long term datasets in the region have yielded up to 67 anuran species at a single site (Duellman, 2005), while Doan and Arriaga (2002) recorded 76 anuran species across five sites with 23 months. In context, our species list of 38 named species and 42 species overall recorded in such a short sampling period nonetheless highlights the exceptional anuran diversity at our study sites even after disturbance events such as the 2005 fires.

All of the recorded species are known to be tolerant to some degree of habitat modification, and even in unburned areas, no primary forest specialist species were recorded. This is a likely indication that our sampling sites lay within transition zones or edge areas, rather than within core areas of burned and unburned forest sites. Additionally, some highly generalistic species were recorded, like *Elachistocleis bicolor*, a species exclusively found in bamboo forests and more often in burned areas. This species has been recorded in savannah habitats subjected to various burning and grazing regimens in northeast Argentina (Cano & Leynaud, 2009) and survives in substantially disturbed and modified habitats. It may therefore

act as an indicator for habitat modification. Only one direct developing species (*Pristimantis peruvianus*) was recorded, occurring in notably higher numbers in terra-firme forest. All other recorded species have aquatic larval stages.

Overall, our data suggest that to fully assess the impact of forest fire on amphibian communities, particularly on primary forest specialists, more extensive sampling over a wider spatial scale between unburned and burned areas stretching over a large number of survey nights is required.

The finding that bamboo forest supported a different assemblage of anuran species with a higher species richness and abundance than terra-firme forest is in contrast to the findings of von May et al. (2010), who found no difference in community composition or species diversity between these forest types. This may be due to differing water availability between the two forest types during this study. While all sites were criss-crossed by small streams and close to large rivers, large patches of *Heliconia* sp. at the bamboo site suggested particularly high levels of soil moisture (Griscom et al., 2007). Increases in diversity and abundance of amphibians are expected close to water bodies due to enhanced numbers of species that rely on them for breeding (Rojas-Ahumada & Menin, 2010), but general water availability becomes particularly important for some anurans in the dry season (Suazo-Ortuño et al., 2008; Urbina-Cardona et al., 2006) when the present study took place.

Supporting this was the dominance of Hylidae at the bamboo forest site. Members of this predominantly arboreal family often only breed in temporary ponds (Bartlett & Bartlett, 2003), making them particularly reliant on areas with higher water availability.

While the short sampling period and the incompleteness of sampling means that these results must be considered as preliminary, the findings are still important for our understanding of how tropical forest fires may affect amphibian communities. The results support previous work (von May et al., 2010) in highlighting the importance of identifying distinct forest types when conducting amphibian studies in this region. Our data also highlight the conservation value of forests degraded by a burn event and of associated edge areas, as even in this short

study, a large proportion of anuran species known to the region were recorded. This is of particular interest for those species that have so far only been recorded outside of protected areas in Madre de Dios, such as *Ranitomeya cf ventrimaculata* and *Osteocephalus buckleyi* (von May *et al.*, 2008). In the eastern Amazon, it has been reported that 38% of burned forests were subsequently deforested (Alencar *et al.*, 2011) suggesting that the risk of further degradation or removal of burned forest may be high. Taking measures to prevent further degradation of the burned areas might therefore be more important for the preservation of anuran communities in this region than previously thought.

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