






# Fighting the flames: site-specific effects determine species richness of Australian frogs after fire

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## ABSTRACT

**Context.** Fire has played an integral role in regulating patterns of biodiversity for millions of years. However, anthropogenic disturbance and climate change has altered fire activity – driving increases in both fire severity and scale. The effect fire now has on the persistence of biodiversity is poorly known, especially for frogs. Studies examining frog responses to fire usually have small sample sizes, focus upon small geographic areas and are based on low-severity fires, which can mean results are not applicable to high-severity fires, such as those expected under future climate change. **Aims.** Our aims were to examine (1) persistence of frog species, measured by species richness, up to 18 months post-fire, and (2) the effects of varying fire severity on frog species richness and recovery, where we expected higher fire severity to lead to lower species richness after fire. **Methods.** Using large-scale citizen science data from the Australian Museum’s FrogID project, coupled with remotely sensed fire data, we present a spatially and taxonomically broad analysis examining post-fire recovery responses for Australian frog species after the 2019/2020 ‘Black Summer’ bushfires. **Key results.** We reveal no overall decrease in the species richness of Australian frogs both in the short- and long-term post-fire. Furthermore, species richness did not decline with increasing fire severity. Instead, species richness and its response to fire was highly site-specific. **Conclusions.** We provide evidence that widespread and common Australian frog species have persisted post-fire in most sites and concluded that this is potentially due to their ability to shelter from fire adequately and/or La Niña-driven high rainfall offering conditions conducive to breeding activity and persistence. **Implications.** We show how citizen science provides critical data for conservation, especially in response to unprecedented disturbance events, such as the 2019/2020 megafires. Our research also highlights the need for ongoing and targeted scientific monitoring, especially for less common or threatened species.

**Keywords:** citizen science, climate change, conservation, ecology, fire, frogs, remote-sensing, species richness.

## Introduction

Fire has regulated patterns of biodiversity for millions of years, but recent anthropogenic disturbances are altering fire impacts on the persistence of biodiversity (Kelly *et al.* 2020). Anthropogenically mediated climate change has increased fire severity and duration and geographic scale across the globe (Moritz *et al.* 2012; Jolly *et al.* 2015), exemplified by recent fires in western USA and south-eastern Australia (Nolan *et al.* 2020; Ward *et al.* 2020). Changes in fire regime are driven in part by: climatic variables, extending periods of dry, hot and windy weather; vegetation changes due to unprecedented precipitation periods; and increased likelihood of ignitions (Williams *et al.* 2009; Moritz *et al.* 2012; Kelly *et al.* 2020).

Many species have adapted to specific fire regimes, so disruption can impact both populations and entire ecosystems (Kelly *et al.* 2020). As fire increases in severity, frequency and scale under climate change, it can directly increase mortality rates, reduce available resources and habitat and fragment populations (Ward *et al.* 2020), increasing extinction rates of biodiversity (Gill and Bradstock 1995). Despite this potentially deleterious outcome, responses to fire are poorly documented, particularly for understudied

taxa like amphibians, invertebrates and reptiles (Dale *et al.* 2001; Pastro *et al.* 2014; dos Anjos *et al.* 2021).

Frogs are already experiencing population declines globally due to various anthropogenic threats (Houlahan *et al.* 2000; Campbell Grant *et al.* 2020). They are likely to be particularly susceptible to high mortality rates or population decline with fire, given their low vagility and reduced ability to flee fire fronts, reliance upon water sources and specific habitat requirements that may be degraded post-fire (Rowley *et al.* 2020). They may also be increasingly vulnerable to extinction due to increased inbreeding, lower population numbers and low immigration rates with fires (Potvin *et al.* 2017). The potential vulnerability of frogs to fire, and other key threats such as chytridiomycosis (Berger *et al.* 2016), make studying the response of frogs to fire critically important.

Most of our understanding of how frogs respond to fire is biased towards North American species (dos Anjos *et al.* 2021), with few investigations in Australia by comparison. However, effects on Australian species vary. Some studies show frog species are not obviously affected by fire (Bamford 1992; Lowe *et al.* 2013; Rowley *et al.* 2020), whereas others show some species experiencing population declines. For example, *Litoria littlejohni* (Daly and Craven 2007) and *Geocrinia lutea* (Driscoll and Roberts 1997) experienced declines, and one study even documented a local population extinction (Lemckert 2000). Despite the contributions of finer-scale or localised studies elucidating the effects of fire on Australian frogs, we lack evidence across larger spatial scales or for mega-fires, such as those experienced in 2019/2020, and those that are predicted to increase under future climate change (Lee *et al.* 2022). Further, such conflicting evidence for a few species limits our ability to extrapolate the impact of fires from previous research.

Fires can be unpredictable in space and time and consequently, assembling resources to study their effects on short notice can be difficult. With the rise of citizen science, it is possible to investigate impacts of fire over large spatial scales and on a time-sensitive basis (Kirchhoff *et al.* 2021). Indeed, citizen science was instrumental in providing preliminary data on short-term persistence of frogs across large spatial scales after the 'Black Summer' bushfires in Australia, with at least 45 out of 66 species detected, and all 33 species with more than five calling records pre-fire recorded calling post-fire (Rowley *et al.* 2020). However, this dataset was only short term, failed to sample range-restricted and rainforest species and focused on summer-breeding. This is particularly relevant because although some individuals may survive the immediate impact of a fire, they are persisting in a burnt landscape with fewer resources and potentially experiencing starvation, as well as an increased risk of predation (Nimmo *et al.* 2019; Dickman and McDonald 2020). Additionally, the impact of varied fire severity on frogs was not assessed.

In this study, we used large-scale citizen science data, coupled with two remotely sensed datasets from the extensive

and high-severity 2019/20 Australian bushfire season to examine: (1) persistence of frog species, measured by species richness, 18 months post-fire across Australia; and (2) effects of varying fire severity on frog species richness and recovery in eastern Australia, where megafires were concentrated. More specifically, we hypothesised that fire would have a negative effect on species richness, particularly at higher severity levels.

## Materials and methods

### FrogID data

Frog location data were derived from the Australian Museum's FrogID project, whereby users submit a 20–60-s audio recording of a frog advertisement call through the FrogID smartphone app, hereafter known as a 'submission' (Rowley *et al.* 2019). A submission can contain multiple frogs calling, forming 'records'. Species calling in each submission are identified based upon call, location, habitat and time of year. Calls are compared with a large database of recordings from known species. Frog species that are difficult to identify are listened to by multiple people and occasionally analysed in sound analysis software. If uncertainty remains over the species identification, we classify the frog as 'unidentified', and the recording does not contribute to the FrogID dataset. FrogID is a relatively non-structured citizen science project, with the majority of records from more populated areas. However, the dataset includes records from over 35% of the country (using 0.5 decimal degree grid cells; Cutajar *et al.* 2022). These data from FrogID indicate breeding effort and habitat, reflected in metadata identifying time, date and location (latitude, longitude and an estimate of location accuracy). Data were filtered by location accuracy, with records over 3000 m discarded, because this measurement commonly means the app could not take the user's location. Over 96% of records had an accuracy of less than 100 m.

### Remote sensing

To estimate time and extent of fire, we used multiple sources of remotely sensed data. First, we extracted data regarding the time of fire from the Digital Earth Australia (DEA) Hotspots resource (<https://hotspots.dea.ga.gov.au/>). We placed a 0.005° (approximately 550 m) buffer around each FrogID record and determined the highest temperature recorded within it, assuming that the highest temperature record coincided with the closest time when the fire front passed each location (Rowley *et al.* 2020). Records were then clipped to shapefiles of fire extent from the National Indicative Aggregated Fire Index (NIAFI) (<http://www.environment.gov.au/fed/catalog/search/resource/details.page?uuid=%7B9ACDCB09-0364-4FE8-9459-2A56C792C743%7D>) to ensure all records were

from burnt regions. DEA/NIAFI data were split into pre-fire (from 2017 until date of fire), short-term post-fire (up to 13 April 2020) and long-term post-fire (after 13 April 2020 until 10 April 2022). To investigate the effect of fire severity, we used Fire Extent and Severity Mapping (FESM) data (<https://datasets.seed.nsw.gov.au/dataset/fire-extent-and-severity-mapping-fesm>) to allocate each FrogID record to a fire severity category (2 = low, 3 = moderate, 4 = high, 5 = extreme). Fire severity category 1 refers to areas that were burnt but the extent is unclear and not reliable. Due to unreliability, we excluded this category from our analysis. The FESM data have been generated from a collaboration with the NSW Rural Fire Service and the NSW Government Department of Planning, Industry, and Environment Remote Sensing and Regulatory Mapping team (who have developed a semiautomated approach to mapping fire extent and severity through a machine-learning framework based on Sentinel 2 satellite imagery). These data were restricted to New South Wales (NSW) and the Australian Capital Territory (ACT), where fires were highly concentrated. Comparable data were not available for other states or territories. We also note that FESM data are an estimate of fire severity based on training data and are broad-scale, and some errors in severity classification may occur (White and Gibson 2022). However, these errors are seen when distinguishing between low- and moderate-severity fires, and between high- and extreme-severity fires (White and Gibson 2022). Despite these potential uncertainties, FESM was the most logistically feasible dataset for our large-scale analysis and has been the preferred choice in numerous remote-sensing studies (Bilney *et al.* 2022; Beranek *et al.* 2023; Law *et al.* 2022, 2023). More information regarding the FESM dataset can be found here <https://datasets.seed.nsw.gov.au/dataset/fire-extent-and-severity-mapping-fesm/resource/73b509a1-7674-45c2-adac-173ccd7d917a>. For FESM derived data, we similarly filtered FrogID data into two categories: pre-fire and post-fire.

Species' habitat preferences can also influence the likelihood they are susceptible to potential fires – for example, which refugia they are likely to shelter in, and their dependency on fire-sensitive habitat (Mahony *et al.* 2023). As such, we examined the composition of FrogID records before, short-term, and long-term after fire in three categories: their ecological grouping (terrestrial breeders, ephemeral pond, permanent water, permanent stream associated and bog or soak); their dependence on fire-sensitive vegetation (five categories: 0–20%, 21–40%, 41–60%, 61–80%, 81–100%); and likely refugia used to shelter from fire (burrow, hollow, under debris, under rocks, under leaf litter or topsoil, dense riparian vegetation and wetland).

## Data analysis

All data analyses were conducted in R ver. 4.3.0 (R Core Team 2021), relying heavily on the tidyverse workflow (Wickham 2017). To account for sampling bias inherent to citizen

science data, we created 'field sampling sites' by placing 1-km grids over Australia (5 km for DEA Hotspot/NIAFI data due to lower resolution) and only sampled grids (or sites) in fire-affected areas with both before and after records for frogs. This limited incorrect species richness counts (response variable of interest) in an area that may have only been sampled before fire and not after (either due to no frogs calling or absence of a citizen scientist recording). This also removed grids that were only sampled post-fire. Two additional measures to limit bias from any uneven sampling were implemented. The first was only grids with at least five submissions both before and after fire were included for species richness to control 'between grid' variation. This limited our sample size for number of grid/sites but provided greater confidence in any changes in species richness seen at each site. The second was to account for 'within grid' variation and ensured even-sampling pre- and post-fire within a grid. For example, if a grid had 18 submissions pre-fire and 32 submissions post-fire, only 18 submissions from post-fire were randomly sampled 10 times and then averaged to calculate species richness. Most post-fire grids were re-sampled because we had significantly more data post-fire ( $n = 9017$ ) than pre-fire ( $n = 1733$ ). Because we randomly sampled species richness to account for sampling biases before and after fire, the composition of the species changed among random samples. Therefore, our analysis focused on community-level metrics (i.e. species richness) as opposed to species-level metrics. However, we calculated the percentage of presence records of frog species across sampling grids in areas burnt in each severity level (low, moderate, high, extreme; Supplementary Tables S4–S7). For each grid, we calculated the Keith vegetation value, a highly detailed classification of vegetation types in Australia (Keith 2004). Our data fell into four main vegetation types: forest (including wet sclerophyll, dry sclerophyll, and rainforest); woodlands; scrubs/heath; and cleared.

To compare frog species richness before and after fire across Australia, we used two generalised additive models (GAMs) – one for DEA Hotspot/NIAFI data and one for FESM data (Wood 2017). For the DEA Hotspot/NIAFI model, the response variable was species richness, with two fixed effects: fire, and the number of samples (additional control for difference in sampling size between grids). Location was added as a bivariate smoothed term ( $k = 10$ ). Similarly, for the FESM model, the response variable was change in species richness from before to after fire, with three fixed effects: severity of fire; vegetation type and the number of samples; and location (bivariate smoothed term,  $k = 10$ ).

## Results

Overall, 36 well sampled species ( $\geq 5$  records before fire) called in the short-term after fire, and all 49 species previously recorded with FrogID in FESM burnt areas called long-term

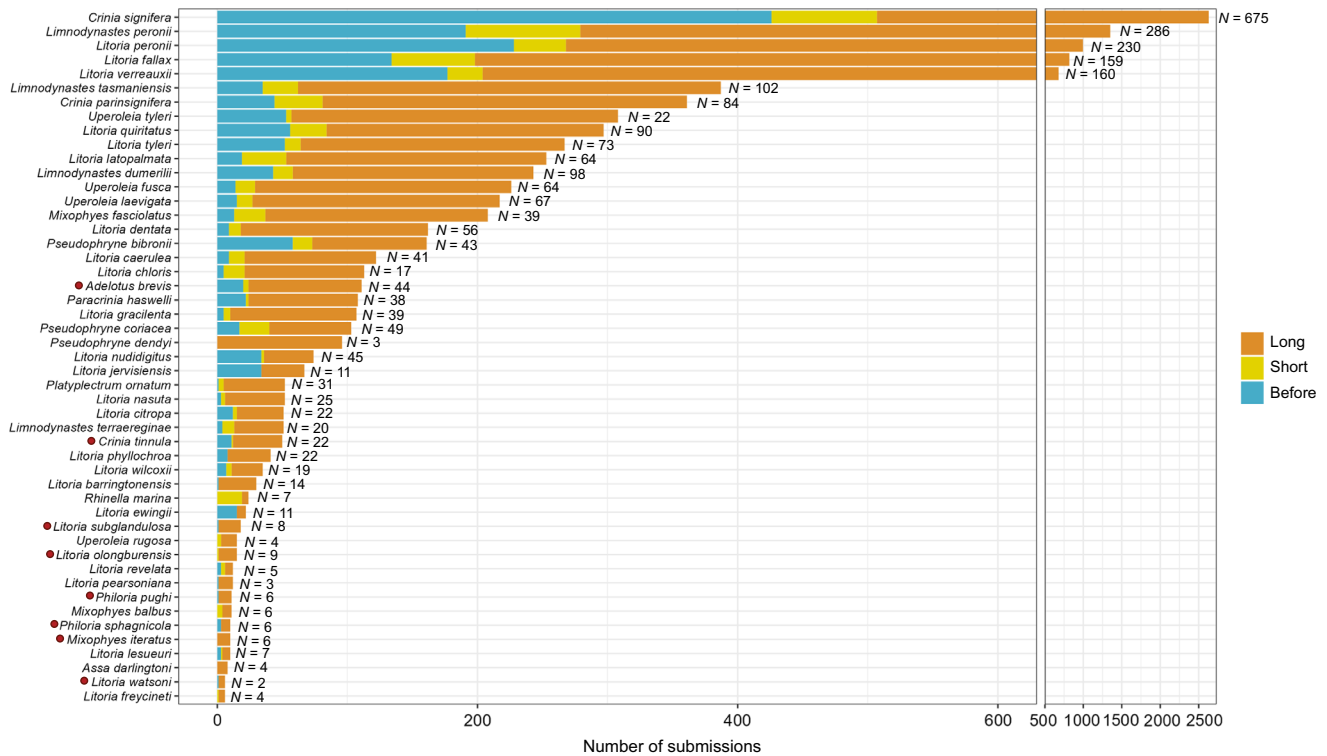
post-fire. These species were taxonomically diverse, representing three of five native frog families in Australia (Fig. 1). Frogs responded similarly in the NIAFI/DEA Hotspot data (Fig. S1). About 58% of the frog species ever detected in FESM burnt areas by FrogID (including those with <5 records) before the fire were calling after 3 months, increasing to 72% after 6 months, 92% after 12 months and 98% after 18 months, until all were calling by around 2 years post-fire (Fig. S2). Frogs with the highest number of records post-fire were of low conservation concern (categorised as ‘least concern’ by the IUCN). Seven threatened species, *Crinia tinnula* (VU), *Litoria subglandulosa* (VU), *Litoria olongburensis* (VU), *Philoria pughii* (EN), *Philoria sphagnicola* (EN), *Mixophyes iteratus* (EN) and *Litoria watsoni* (EN), and one near-threatened species, *Adelotus brevis*, called long-term after fire but had low numbers of records (Fig. 1).

Frogs from all ecological groups (terrestrial, ephemeral pond, permanent pond, stream associated, terrestrial and bog/soak) were present after fire, with comparably proportional records between groups for before, short- and longer-term after fire (Fig. S3). However, frog species associated with permanent waterbodies were the best represented in the dataset, and there were low number of records for frogs associated with bogs/soaks and those that relied upon both terrestrial and ephemeral ponds (Fig. S3). Frogs that had a

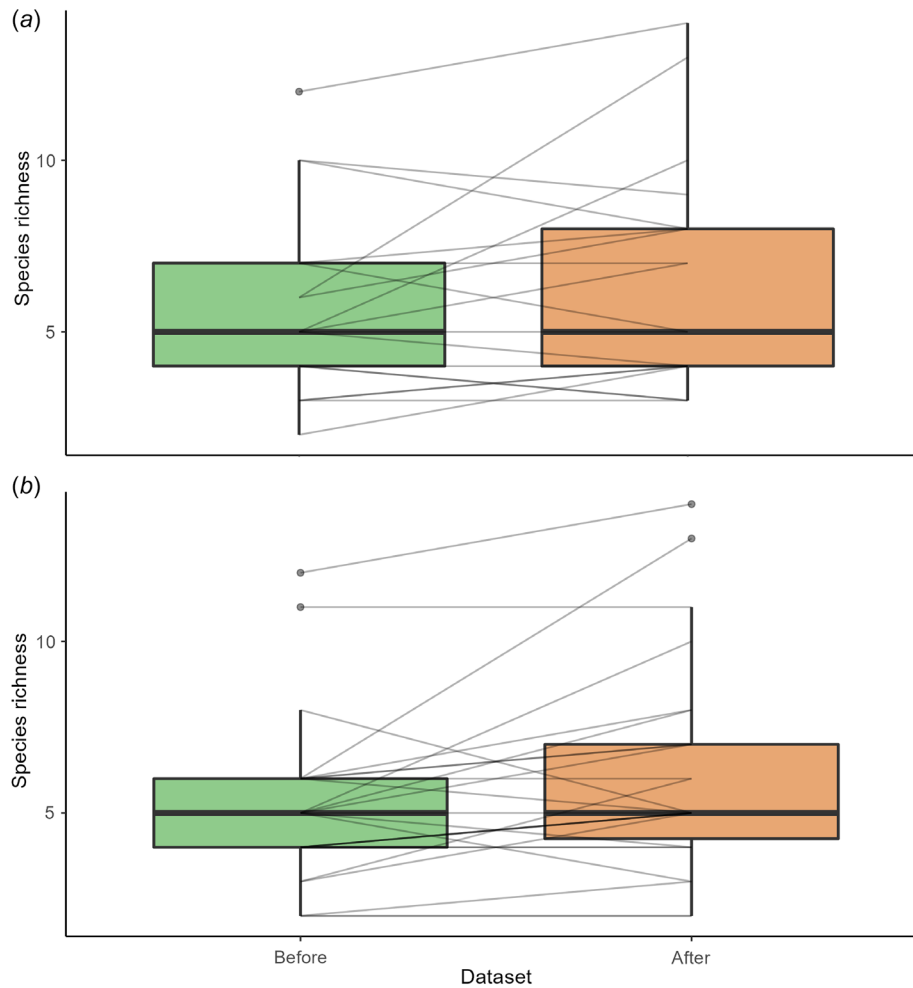
low (0–20%) dependence on fire-sensitive vegetation were best represented in our data, and those with a high dependence were present post-fire but had a low number of records (Fig. S4). The most common frog species in our dataset were those that found refuge under debris during fire; however, frogs from all refugia groupings were present post-fire, albeit in lower numbers (Fig. S5). Mean species richness did not significantly change post-fire compared with pre-fire (estimate = -0.130,  $z = -0.986$ ,  $P = 0.324$ ; Fig. 2). Instead, changes in species richness were site specific, increasing for nine sites, remaining unchanged for four sites and decreasing for six sites post-fire (Fig. 3; Table S1). Although species richness was, on average, higher in low-severity burn areas, species richness did not significantly differ with varying fire severity level, even after accounting for vegetation type and sample size ( $P > 0.05$ ; Table 1, Fig. 4). Similarly for DEA Hotspot/NIAFI data, changes in species richness were site-specific (Table S2). Specific changes to species and certain ecological groups varied across severity levels (Tables S3–S7).

### Discussion

We present a geographically and taxonomically broad analysis examining post-fire responses for frog species



**Fig. 1.** Well sampled frog species with high numbers of FrogID submissions ( $n \geq 5$ ) calling before fire (blue), short-term after fire (yellow) and longer-term after fire (orange) using New South Wales Fire Extent and Severity Mapping (FESM) data. All species were found to be calling in the long-term. Red dots indicate threatened and near-threatened species. Horizontal axis is truncated due to high numbers of records for some species.  $N$  = unique number of sites.

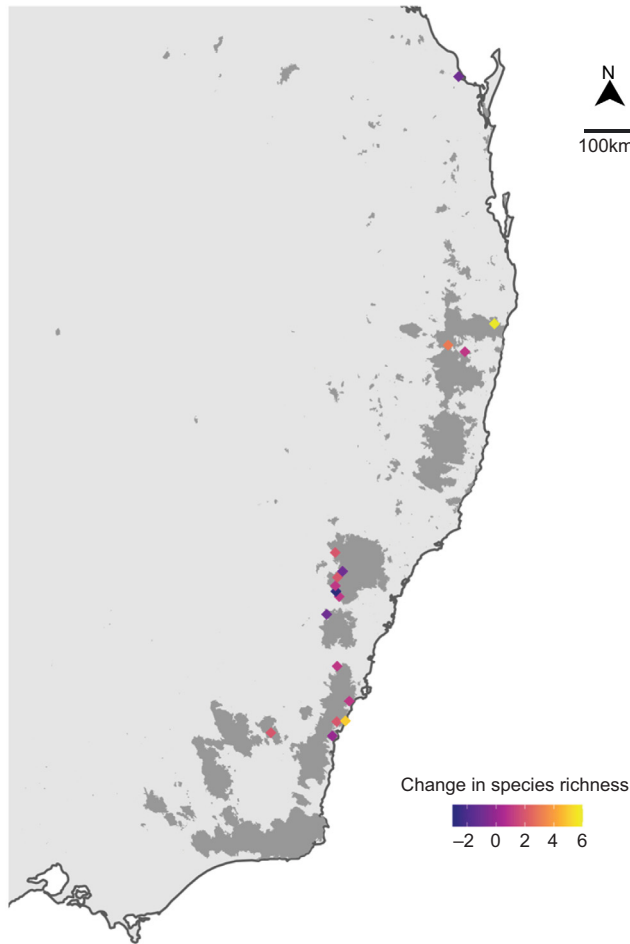


**Fig. 2.** (a) Overall species richness for bushfire-affected areas before and after fire for DEA Hotspot/NIAFI datasets encompassing New South Wales and Victoria. (b) FESM fire severity dataset encompassing only New South Wales. Slope of lines represent direction of change in species richness for each sampling site (either increase, decrease, or no change).

across south-eastern Australia – a continent increasingly affected by climate change and consequently increasing fire severity and extent (Bureau of Meteorology, CSIRO 2020; Lawrence *et al.* 2022). Encouragingly, Australian frog species richness did not decline overall after fire even with increasing fire severity, against our original hypothesis. Species richness increased in some survey areas but decreased in other areas, suggesting site-specific effects.

These variations in frog species richness at site level is consistent with other studies, identifying some increases (Brown *et al.* 2011; Allingham and Harvey 2013; Grundel *et al.* 2015; Mester *et al.* 2015) but also decreases after fire (Cano and Leynaud 2010; Constible *et al.* 2016). Such dichotomous responses occur in Australian frogs, with no discernible effects of fire detected for *Heleioporus eyrei*, *Litoria olongburensis*, *L. freycineti* or *Crinia tinnula*, but a negative impact was documented for both *Litoria littlejohni* (Daly and Craven 2007) and *Geocrinia lutea* (Driscoll and Roberts 1997).

Furthermore, Beranek *et al.* (2023) found that amphibian metacommunities in the southern regions of NSW were negatively impacted by high fire severity, whereas an effect was not detected in the northern region. Generalist species may respond positively to fire (e.g. *Anaxyrus spp.*; dos Anjos *et al.* (2021), compared with specialist species (e.g. *Phyllodytes luteolus*; (Papp and Papp 2000; dos Anjos *et al.* 2021). Our generalist/common species (*Crinia signifera*, *Litoria peronii*, *Limnodynastes peronii*) had the most records compared with more specialist species with few records (for example, our threatened species). Persistence in post-fire landscapes reflects evolutionary history (see Beranek *et al.* 2023), ecological interactions, time since last burn, availability of local waterbodies and available shelter (dos Anjos *et al.* 2021). Some frogs can hide in burrows, ant-mounds and even bromeliads during fires (Moreira *et al.* 2009; Nomura *et al.* 2009; dos Anjos *et al.* 2021). Australian frogs take shelter in hollows, under bark, in dense vegetation, within deep soil



**Fig. 3.** Change in species richness per DEA/NIAFI site in southeast Australia. Species richness increased in some sites and decreased in others post-fire. Some sites did not change species richness. Shaded areas indicate fire zones.

cracks or within reeds such as *Lomandra spp.* (Mahony et al. 2023). Changes in diversity are likely to be considerably less sensitive than changes in abundance, and fire may impact frog population size, which may not be captured in our dataset (Hillebrand et al. 2018; Antão et al. 2020). It would also be useful to examine what factors drive adaptiveness to fire regimes in different frog species and what makes some species more susceptible. Our analysis was not able to necessarily differentiate between persistence and recolonisation, because some species presence post-fire could be due to quickly recolonising burnt areas rather than persistence through the fire period. Fine-scaled *in-situ* research, however, could contribute to better understanding these differences after future fire events.

Although our surveys were widespread, they are unlikely to include all frog species present. Time of year and weather, in particular, influence frog calling activity (Thompson et al. 2022), with some species only detectable under specific conditions (e.g. *Litoria brevipalmata* are rarely detected

**Table 1.** Statistical output of generalised additive model investigating change in species richness ~ fire severity (low, moderate, high, extreme) + vegetation type (cleared, forest, scrub/heath, woodland) + number of submissions per site (N) + non-linear smoother (longitude, latitude). No significant effects were identified. Note that change in species richness in each vegetation category is compared with change in species richness in cleared sites.

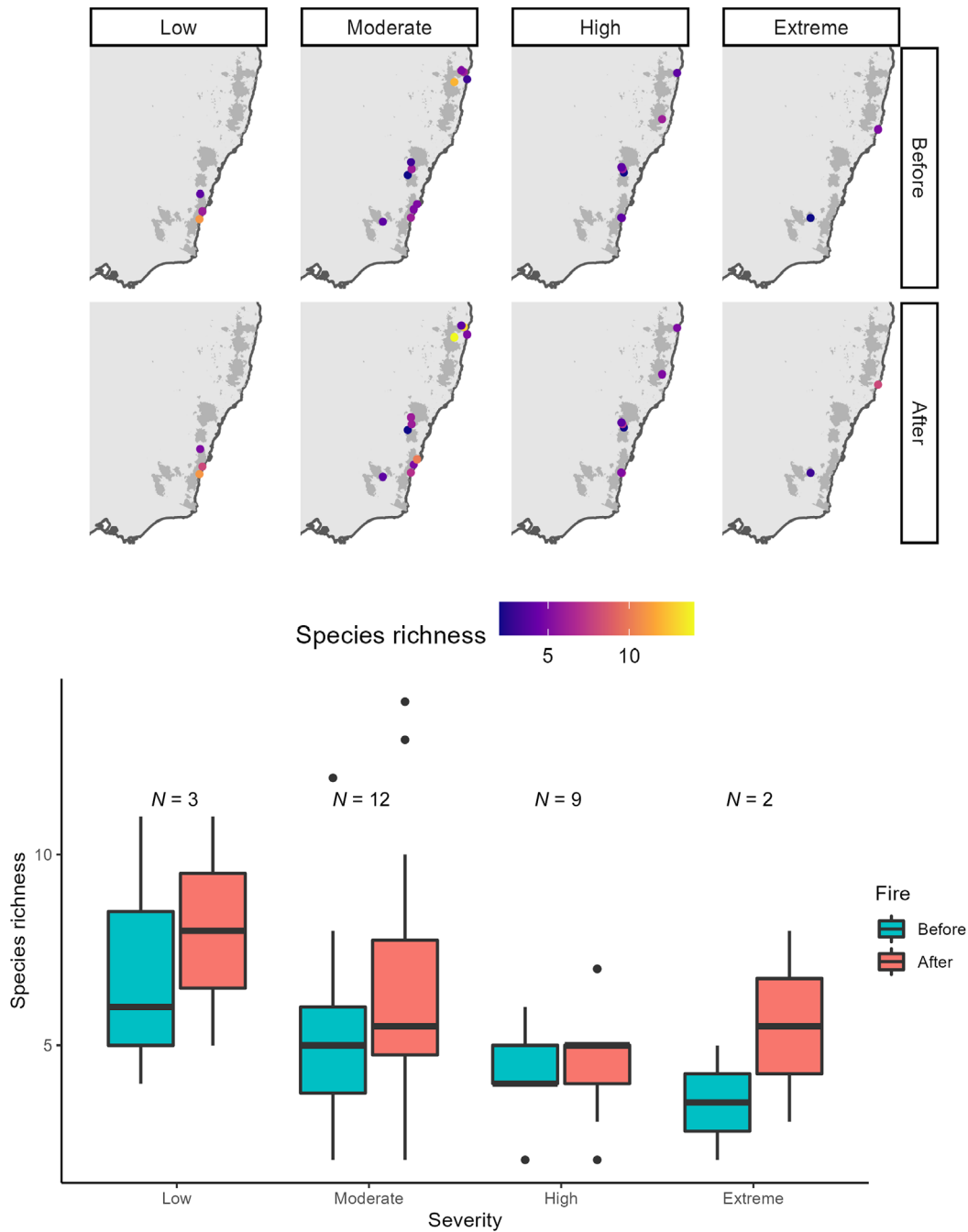
	Estimate	t-value	P
Extreme (intercept)	1.959	0.806	0.432
High	-1.701	-0.653	0.523
Moderate	-0.294	-0.116	0.909
Low	-0.773	-0.261	0.797
Forest	0.017	0.018	0.986
Scrub/Heath	-0.188	-0.051	0.960
Woodland	1.613	0.633	0.536
N	-0.014	-0.717	0.483

	d.f.	F-value	P
s(lng, lat)	2	1.048	0.374

except after recent heavy rainfall). In addition, range-restricted species that occur in remote areas are less well sampled via the FrogID project and other citizen science projects. Other survey methods, such as targeted scientific surveys or passive acoustic monitoring (Beranek et al. 2023), may be necessary to elucidate the effects of fire on these species. This may explain why, despite considerable data collection, there were still relatively few sites with high fire impact ( $n = 11$ ; Fig. 4). Other species may remain undetected if surveys were not conducted at the right time of year (e.g. *Pseudophryne bibronii* are most active in autumn). Moreover, we had relatively few records for many threatened species (in some cases none), limiting our ability to test their responses after the 2019/20 bushfires. This may be due to low survey effort in the habitat of many of the threatened frog species in the fire zone (i.e. relatively remote rainforest habitats). Thus, our analysis is complementary to targeted, ‘on the ground’ surveys for threatened species to quantify their responses post-fire (Beranek et al. 2023). Nonetheless, the high levels of sampling overall by citizen scientists post-fire increases confidence in areas where we see sites with decreased species richness. For example, less than half of the species were detected at a site in the Blue Mountains (NSW) (seven before to only three species post-fire), despite over three times more sampling effort post-fire. The site-specific changes our data reveal will inform identification of priority zones for future surveys to determine the full effects of fire on frog populations. For instance, these field-based surveys will be necessary to detect breeding success and to assess less readily detectable impacts from ecological data, such as genetic erosion (Potvin et al. 2017).

Moreover, unlike most Australian bushfire seasons, the subsequent onset of La Niña (September 2020) provided additional



**Fig. 4.** Species richness per site under four different fire severity categories (low, moderate, high and extreme) in New South Wales. Species richness overall did not significantly change before and after fire between the different levels of fire severity. Shaded areas indicate fire zones. N represents the number of sites within each fire severity category.

rainfall to the burnt landscape (Bureau of Meteorology 2020) – offering conditions likely more conducive for frog breeding activity and persistence. Thus, we emphasise that our results should be interpreted with caution: future results may differ if we see similar bushfire activity under more erratic El Niño Southern Oscillation (ENSO)/Indian Ocean Dipole (IOD) cycles due to climate change (Cai *et al.* 2009,

2013; Fasullo *et al.* 2018). For instance, an overall decrease in species richness may occur under El Niño conditions because necessary requirements for breeding recovery (such as water availability) may not be met. Furthermore, this study does not consider the interactions with other stressors and extreme events predicted under climate change, such as drought (King *et al.* 2017) or disease (Hossack *et al.* 2013).

Thus, future research integrating these additional stressors with fire would be of great interest.

In conservation biology, we require extensive yet timely data to make informed decisions (Kirchhoff et al. 2021). After the severe bushfires, the global pandemic made collection by professional biologists problematic (Corlett et al. 2020). As a result, species priority lists for post-fire monitoring were estimated based on expert advice in the absence of the most up-to-date data. However, our study identifies the importance of citizen science in providing critical data for conservation, especially in response to unprecedented fires, and highlights how citizen science may also be relevant for other disturbances. It provided useful data on the effects of fire severity and extent, which did not significantly affect species richness of frogs overall in either the short- or long-term after fire. We emphasise the need for ongoing citizen science data collection and targeted scientific monitoring, especially for species less well-sampled, via citizen science and for threatened species.

## Supplementary material

Supplementary material is available [online](#).

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**Data availability.** Not all data can be made Open Access due to data sensitivity/privacy of the underlying recordings and localities of threatened or otherwise sensitive species (Rowley and Callaghan 2020), but most data (i.e. with exact localities of sensitive species buffered) are made available to the public on an annual basis (Rowley and Callaghan 2020). Code used within our study will be made available in a Zenodo repository. Further data can also be requested from the Australian Museum.

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