Threatened Species Recovery Hub
National Environmental Science Programme

Post-fire impact assessment for priority frogs: northern *Philoria*

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July 2021

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Queensland Government

Cite this publication as: Heard, G., Bolitho, L., Newell, D., Hines, H., McCall, H., Smith, J., and Scheele, B., 2021. Post-fire impact assessment for priority frogs: northern *Philoria*. NESP Threatened Species Recovery Hub Project 8.1.3 report, Brisbane.

Main cover image: *Philoria* habitat. Image: Liam Bolitho. Insert images (from top to bottom): *Philoria kundagungan*. Image: Harry Hines; *Philoria richmondensis*. Image: David Newell; *Philoria loveridgei*. Image: Harry Hines.

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Executive Summary

Bushfires in Australia are predicted to increase in frequency and severity with climate change. The large-scale 'Black Summer' fires of 2019/2020 validated these predictions, burning 1.8 M ha of forest and woodland across eastern Australia. The fires began during Australia's hottest and driest year on record and burned an unprecedented extent of rainforest. In northern New South Wales and southern Queensland, over half of the Gondwana Rainforests of Australian World Heritage Area was affected by the 2019/2020 fire event — habitats that have rarely, or never, been recorded to burn.

These rainforests support a number of endemic, range-restricted amphibians, including several species of Mountain Frog (*Philoria* spp.). These frogs primarily occur in rainforest or wet sclerophyll vegetation communities, where they breed in headwater drainage lines and bogs. Their dependence on wet areas, as well as their limited and fragmented ranges and 'slow' life-history traits make these frog species sensitive to increased drought and fire associated with climate change.

This project aimed to assess the impact of the 2019/2020 'Black Summer' fires on three species of *Philoria* that occur primarily in the Gondwana Rainforests of Australia World Heritage Area. The species are:

1. *Philoria kundagungan* (Mountain Frog), from the Mistake Mountains and Main Range in Queensland, and Mount Clunie, Acacia Plateau and Koreelah Range in northern New South Wales;
2. *Philoria richmondensis* (Richmond Mountain Frog), from the Richmond Range and adjoining ranges in northern New South Wales, and;
3. *Philoria loveridgei* (Loveridge's Mountain Frog), from the Border Ranges, Mount Warning, Nightcap and Mount Nothofagus National Parks in New South Wales and Lamington, Springbrook and Mount Barney National Parks in Queensland.

These three species were affected by the 2019/2020 fires but were not addressed by other projects on the impact of these fires on threatened frogs. Specific aims of this project were:

1. Estimate the portion of potential habitat for each species affected by fire during the 2019/2020 season.
2. Resurvey previously surveyed sites for *P. kundagungan* and *P. richmondensis*, as well as new areas in neighbouring habitat that were affected by fire in the 2019/2020 season.
3. Survey a selection of sites for *P. loveridgei* in the western portion of its range, including habitat burnt in the 2019/2020 fires.
4. Examine the relationships between fire severity and both site occupancy and calling male abundance for *P. kundagungan* and *P. richmondensis*.
5. Assess change in site occupancy and calling male abundance from monitoring of *P. kundagungan* and *P. richmondensis* prior to the 2019/2020 fires.
6. Estimate the area of occupancy of both *P. kundagungan* and *P. richmondensis* following the 2019/2020 fires.

Surveys were completed at 48 sites for *P. kundagungan* and 50 sites for *P. richmondensis* during the 2020/2021 breeding season, with a further 5 sites surveyed for *P. loveridgei* in the western portion of its range. Data on habitat attributes, climate and fire severity were collected to assess their effect on occupancy patterns post-fire. The area of habitat burnt for each species was calculated by defining the area of potential habitat and overlapping these areas with the mapped 2019/2020 fire extent. Previous survey data for *P. kundagungan* and *P. richmondensis* was compiled and used to assess change in occupancy rates and abundance of calling males post-fire.

All three species had extensive areas of habitat that were affected by the 2019/2020 wildfires. We estimated that 30% of suitable habitat was affected for *P. kundagungan*, 12% for *P. richmondensis* and 54% for the western populations of *P. loveridgei.*

Field surveys during the 2020/2021 breeding season suggested northern *Philoria* have some resilience to fire. Calling males of each of the focal species were observed at burnt sites, even those that were burnt at moderate-to-high severity. However, we also found evidence of impacts of the 2019/2020 fires, as well as the drought conditions prevailing before and after these fires. Frogs were detected at a higher proportion of unburnt sites than burnt sites. Occupancy analysis found burnt sites were significantly less likely to be occupied by *P. kundagungan* and supported fewer calling males. *P. richmondensis* showed a similar pattern, although the fire was restricted to areas of low habitat suitability and limited our ability to statistically detect any effect of fire. Stream saturation extent — a measure of local drought stress — was also a key predictor of occupancy and calling male abundance for both *P. kundagungan* and *P. richmondensis*.

Occupancy and calling activity in *P. kundagungan* were lower post-fire in 2020/2021 compared to before the fire in 2016/17. For *P. richmondensis*, occupancy rates and counts of calling males were significantly lower in 2019/2020 just prior to the 'Black Summer' fires and during a period of intense drought. Encouragingly, significant winter and spring rainfall led to an increase in occupancy and calling male abundance for *P. richmondensis* post-fire in 2020/2021; however, counts of calling males were lower overall compared to earlier (2012/2013) surveys.

Both *P. kundagungan* and *P. richmondensis* were predicted to have very small areas of occupancy based on occupancy rates in 2020/2021. For *P. kundagungan*, the area of occupancy including potential terrestrial habitat was estimated to be 46.35 km^2, with only 13.46 km^2 of in-stream breeding habitat. For *P. richmondensis*, area of occupancy including potential terrestrial habitat was 40.43 km^2, with just 6.80 km^2 of in-stream breeding habitat.

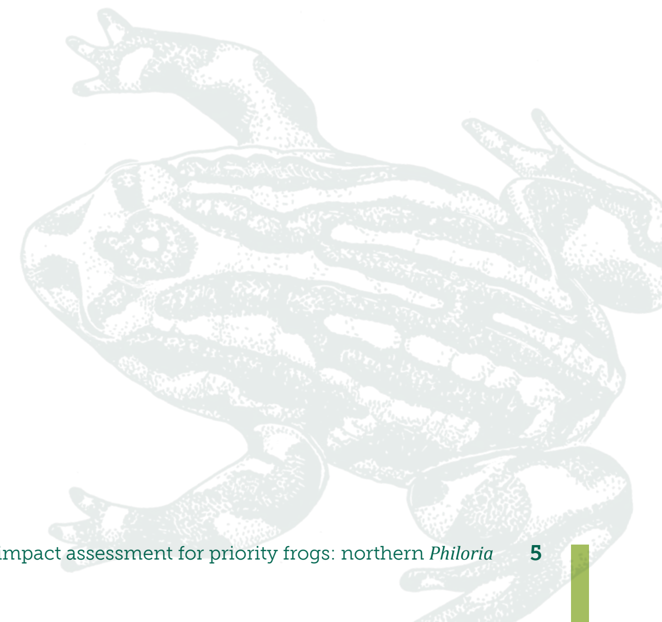
This study provides evidence that while these northern *Philoria* species have some resilience to drought and fire, the increasing frequency and severity of these perturbations under climate change represents a clear threat.

Proposed management actions arising from this study include:

1. Conservation status listing —This study supports the listing of *P. kundagungan* and *P. richmondensis* as Endangered under the EPBC Act, as recently recommended by the Threatened Species Scientific Committee.
2. Protection from fire — Recovery of each of northern *Philoria* from the 2019/2020 fires requires maintenance of unburnt refuges and recovery of burnt habitat. Fire suppression and careful planning of fuel reduction burning in adjoining fire-adapted communities will be crucial.
3. Protection from other threatening processes — Feral pig control, cattle exclusion and weed control is urgently required in a number of areas, particularly the Main Range, Mount Barney and Yabbra National Parks.
4. Captive breeding, head-starting and re-introduction — Establishing captive populations should be pursued as a matter of urgency. Head-starting and re-introduction may also support declining populations or re-establish those that have gone locally extinct.
5. Conduct ongoing monitoring — Ongoing monitoring of northern *Philoria* is vital to track population change and establish triggers for conservation action.

Priorities for future research include:

1. Complete spatial prioritisation analyses — To identify target habitat for post-fire management of northern *Philoria* and identify long-term climate-change refugia.
2. Complete detailed ecological studies — To fill gaps in our knowledge of these frogs, with the objective of completing population viability analyses.
3. Expand sampling for amphibian chytrid fungus — To clarify impacts of chytridiomycosis on northern *Philoria*, which is a known threat to other species in the genus.
4. Assess climate buffering — To examine whether rainforest environments can mitigate climate change impacts on these species.
5. Complete a meta-analysis of post-fire amphibian studies — To provide a broader understanding of the response of Australian frogs to bushfire.



Introduction

Background

The Black Summer fires of 2019/2020 burnt some 1.8 M hectares of forest and woodland across eastern Australia (Collins *et al.* 2021), beginning during Australia's hottest and driest year on record. The radiative power of these fires, the area burnt at high severity and the area of rainforest burnt were all outside the historical record (Abram *et al.* 2021; Collins *et al.* 2021). In New South Wales, around 37% of the state's rainforests were affected by fire, including 54% of the Gondwana Rainforests of Australia World Heritage Area (DPEI 2020). The equivalent figure for the Queensland component of this World Heritage Area was 36% (DAWE 2020b). In total, 53% of the Gondwana Rainforests of Australia World Heritage Area was affected by fire during the 2019/2020 event (some 196,296 ha; DAWE 2020b).

The Gondwanan rainforests of northern New South Wales and southern Queensland support several species of endemic, range-restricted amphibians. Notable among these are five of the six species of *Philoria* (formerly *Kyarranus*). These species are: (i) *P. kundagungan* (Ingram & Corben 1975) of the Mistake Mountains and Main Range in Queensland, and Mount Clunie, Acacia Plateau and Koreelah Range in northern New South Wales; (ii) *P. loveridgei* (Parker 1940) of central and eastern sections of the Border Ranges; (iii) *P. pughi* (Knowles *et al.* 2004) of the Gibraltar Range and Washpool National Parks; (iv) *P. richmondensis* (Knowles *et al.* 2004) of the Richmond Range and connected uplands, and; (v) *P. sphagnicolus* (Moore 1958), occurring from Guy Fawkes National Park south to the Bulga Plateau, north-east of Taree. These species are all allopatric, occurring as scattered mountain-top endemics (Anstis 2013). They primarily occupy rainforest or wet sclerophyll vegetation communities, and are known almost exclusively from headwater drainage lines and bogs in which breeding occurs (Knowles *et al.* 2004; Anstis 2013). Males construct and call from nests in muddy seepages within these headwater streams, usually under rocks or logs, amongst mosses or root masses, or within clumps of Rainforest Spinach (*Elatostema reticulatum*) (Anstis 2013). Tadpoles develop entirely within these nests and are reliant on constant moisture (Anstis 2013). Microhabitat use outside of the breeding season is almost totally unknown; however, radio-tracking studies on the southern member of the genus, the Baw Baw Frog (*P. frosti*), suggest that both males and females likely migrate small distances (< 85m) away from seepages after breeding to occupy the surrounding forest slightly upslope (Hollis 2003). During the non-breeding season, *P. frosti* shelter and forage under logs and in root masses, in soil cavities and amongst surface vegetation, including leaf litter (Hollis 2003). It is likely the northern *Philoria* show similar patterns of microhabitat use outside of the breeding season and there are occasional observations of animals away from potential breeding sites (H. Hines, pers. obs.).

These traits, and others, suggest that all *Philoria* species may be particularly sensitive to increased drought and fire intensity under climate change (Newell 2018). Severe rainfall deficits leading to drying of headwater seepages may lead to complete reproductive failure at some sites (due to skipped breeding or mortality of eggs and tadpoles) and desiccation of microhabitats could lead to mortality of juveniles and adults, either through effects on hydration or reductions in their fossorial invertebrate prey. Forest fires may directly kill juveniles and adults, with the microhabitats described above being combustible when dry (logs, roots masses, leaf litter). However, broader effects of fire are also probable, including sedimentation of headwater seepages, changes in soil structure and groundwater seepage, increased temperatures and evaporation in habitat where canopy loss is significant, and incursion by non-native species (including feral ungulates, Cane Toads and weeds, each of which are recognised threats to *Philoria*; Gillespie *et al.* 2020).

The small and fragmented ranges of *Philoria*, their small clutch sizes and slow growth rates, and their association with bogs and soaks, also suggest that members of the genus are inherently susceptible to increased perturbations under climate change. Small range size and low fecundity are key correlates of extinction risk for Australian frogs (Hero *et al.* 2005; Murray & Hose 2005; Murray *et al.* 2011), and slow rates of growth and maturation are consistently identified as factors in species decline generally (McKinney 1997). Small range size entails inherent risks, but these are exacerbated for species susceptible to megafires, for which the probability of impact to the entire range increases significantly as range size narrows. Relationships between extinction risk and both low fecundity and slow growth rates relate to an inability to rebound demographically from perturbations; populations of species with these traits remain suppressed for longer following perturbations, increasing the risk of stochastic extinction before recovery can take place (Pimm, Jones & Diamond 1988; Henle *et al.* 2004). With regard to association with bogs and soaks, Murray *et al.* (2011) demonstrated that Australian frogs associated with these habitats have had a substantially higher probability of decline over recent decades than other species. The reasons for this are various; however, the increasing frequency and severity of both drought and bushfire are almost certainly key reasons, resulting from ecological mechanisms such as those listed above.

This study sought to assess the impact of the 2019/2020 'Black Summer' fires on three species of *Philoria* from the Gondwana Rainforests of Australia World Heritage Area; namely, *P. kundagungan, P. richmondensis* and populations of *P. loveridgei* in Mount Barney and Mount Nothofagus National Parks at the western limit of this species' range. These three taxa were selected from the wider pool of threatened frog species affected by the 2019/2020 fires, being species for which extensive fires occurred within their known range and for which a clear gap was identified among the studies initiated after the 2019/2020 event (see 'Context' below).

Study species

The Mountain Frog (*P. kundagungan*) occurs across montane sections of Tooloom National Park, north through Mount Clunie and Koreelah National Parks in northern New South Wales to the Main Range National Park in Queensland, occurring as far north as the uplands of the Mistake Mountains (Bolitho *et al.* 2021). However, only high-elevation headwater streams are occupied, resulting in a recent estimate of the area of breeding habitat for *P. kundagungan* of only 11 km^2 (Bolitho *et al.* 2021). This species is listed as Endangered in New South Wales and Queensland, and by the International Union for the Conservation of Nature (IUCN 2021). It is currently not listed as threatened under the Commonwealth *Environment Protection and Biodiversity Conservation Act 1999* ('EPBC Act'), although this is currently under review following its assessment as 'Endangered' by the Commonwealth's Threatened Species Scientific Committee.

The Richmond Mountain Frog (*P. richmondensis*) is restricted to far north-eastern New South Wales, with a disjunct distribution along the montane uplands of Yabbra National Park in the west and the Richmond Range and Toonumbar National Parks in the east (Knowles *et al.* 2004; Anstis 2013). This small and fragmented distribution, and projected declines under climate change, have led to the species being listed as Endangered in New South Wales and by the IUCN. It is not currently listed as threatened under the EPBC Act, although it too is currently under review following its assessment as 'Endangered' by the Commonwealth's Threatened Species Scientific Committee.

Loveridge's Mountain Frog (*P. loveridgei*) is more extensively distributed than the previous two species. It occurs in montane uplands of the Border Ranges, Mount Warning, Nightcap and Mount Nothofagus National Parks in New South Wales and Lamington, Springbrook and Mount Barney National Parks in Queensland (Anstis 2013). The species is listed as Endangered in New South Wales and by the IUCN, but is not listed as threatened in Queensland, nor on the EPBC Act. The western populations in the Mount Barney and Mount Nothofagus National Parks were particularly impacted by the 2019/2020 fires.

Objectives

This study had six specific aims under the broader objective of assessing the impacts of the 2019/2020 fires on *P. kundagungan*, *P. richmondensis* and western populations of *P. loveridgei*:

1. Estimate the portion of potential habitat for each species affected by fire during the 2019/2020 season.
2. Resurvey all sites included in recent occupancy surveys for *P. kundagungan* and *P. richmondensis*, adding sites in habitat affected by fire in the 2019/2020 season.
3. Survey a selection of sites for *P. loveridgei* in the western portion of its range, including habitat burnt in the 2019/2020 fires.
4. Examine relationships between fire and both site occupancy and calling male abundance for *P. kundagungan* and *P. richmondensis*.
5. Assess change in site occupancy and calling male abundance from monitoring of *P. kundagungan* and *P. richmondensis* prior to the 2019/2020 fires.
6. Estimate the area of occupancy of both *P. kundagungan* and *P. richmondensis* following the 2019/2020 fires.

Context

The large-scale bushfires across Australia's eastern seaboard during the 2019/2020 season are likely to have had a major impact on biodiversity, yet the responses of Australian wildlife to such events is poorly understood. This is particularly the case for threatened frog species, which tend to occur in cooler, moist, upland areas where fire is less frequent or unknown prior to the 2019/2020 bushfires (Legge *et al.* 2020).

This project aimed to increase our understanding of how priority frog species were impacted by the 2019/2020 bushfires. This information will guide whether active conservation interventions are needed to assist species recovery, inform management priorities, and help provide the information required to undertake robust assessments of whether the fire impact has altered species extinction risk.

At the initiation of this project, the specific frog species to target for on-ground surveys had not been identified, as various funding applications for projects on priority frog species had been submitted to the 'Wildlife and Habitat Bushfire Recovery Program' open call. The outcome of these applications (announced in July 2020) was used to guide selection of the target species, in consultation with state and territory government managers and other research partners. Through this process, we identified three of the five northern species of *Philoria – P. kundagungan, P. richmondensis* and western populations of *P. loveridgei* – as representing key gaps in the suite of projects on threatened frogs affected by the 2019/2020 fires (Table 1).

Populations of *P. kundagungan, P. richmondensis* and western *P. loveridgei* had each been affected by the 2019/2020 event, with severe and/or unprecedented fires within the Gondwana Rainforests of Australia World Heritage Area. As above, these fires had the potential to have significant impacts on these populations, some of which occupy very small and isolated patches of rainforest that are intolerant of fire. However, three other factors were influential in the selection of these species:

1. Existence of baseline data on occupancy and abundance: Research by Dr David Newell's group at Southern Cross University and funded via the New South Wales 'Saving Our Species' program, provided solid baseline data on the occurrence and abundance of *P. kundagungan* and *P. richmondensis* pre-fire, including range-wide surveys for *P. kundagungan* in the 2016/2017 breeding season (Bolitho *et al.* 2021), and for *P. richmondensis* in the 2012/2013 season (Willacy 2014) and the 2019/2020 season (L. Bolitho and D. Newell unpublished data). These studies established survey protocols and sites across the range of these two species (including areas that burnt). Moreover, these surveys were designed using a robust methodology that provided a solid basis for estimating change in occupancy and abundance following the 2019/2020 fires.
2. Insights into responses of rainforest frogs to fire: Bushfire in rainforest is a very rare event historically, although the frequency and severity of fire impacts on rainforest in Australia are projected to increase this century under climate change. Focussing on *P. kundagungan, P. richmondensis* and western populations of *P. loveridgei* took advantage of a rare opportunity to assess fire-impacts on rainforest-affiliated threatened frogs.
3. Opportunities to collaborate with state agencies: Opportunities to collaborate with New South Wales Department of Planning, Industry and Environment were already established through the state's 'Saving Our Species' project on *P. kundagungan* and *P. richmondensis*. Collaboration with the Queensland Department of Science and Environment was also already in place for *P. kundagungan* and *P. loveridgei*, both priority taxa for post-fire assessments in that state.

Methodology

Study area and survey sites

The study area may be broadly defined as the northern section of the Gondwana Rainforests of Australia World Heritage Area, taking in the uplands of north-eastern New South Wales and south-eastern Queensland (Figure 1). In the east, surveys were completed for *P. richmondensis* in Yabbra National Park, Richmond Range National Park, Bungboozle Flora Reserve, Murray Scrub Flora Reserve and Toonumbar National Park. In the central-north of the study area, surveys were completed for western populations of *P. loveridgei* in Mount Barney National Park (Mount Ballow area). In the west and north-west of the study area, surveys were completed for *P. kundagungan* in Tooloom and Koreelah National Parks and Beaury State Forest in New South Wales and along the length of Main Range National Park in Queensland.

Previous studies by Willacy (2014) and Bolitho *et al.* (2021) established the core of the sites surveyed in this study. Sites were defined as 100 m transects along headwater streams (1^{st} or 2^{nd} order), beginning from the first expression of surface water at the time of original site inspections. Core sites for *P. richmondensis* (*n* = 37) were established by Willacy (2014) in spring 2012 with the aid of a Maxent model of the species' distribution (Willacy 2014). Headwater streams with a predicted habitat suitability of 40% or greater were identified, with only those within 500 m of a road or trail considered (allowing access). Sites were randomly located in 12 headwater stream catchments at varying elevations, spaced at least 200 m apart (Willacy 2014). Core sites for *P. kundagungan* (*n* = 32) were established by Bolitho *et al.* (2021) in spring 2016. Sites were randomly selected within the known range of this species under four constraints: (1) they contained a stream or creek > 100 m in length; (2) they were in rainforest or within 500 m of rainforest; (3) they were > 500 m from another site, and; (4) they were within 2 km of a road or operational fire trail. Sites were distributed across the known range to ensure adequate coverage.

To increase coverage of habitat burnt during the 2019/2020 fires, we added a further 16 sites for *P. kundagungan* and a further 13 sites for *P. richmondensis*. For *P. kundagungan*, sites were added further north along Main Range in the Mistake Mountains area, and in the south in the Emu and Teviot Creek catchments. Some of these sites could only be accessed from downstream and did not conform to the criterion of Bolitho *et al.* (2021) of beginning at the first expression of surface water. Selection of these sites was also affected by severe long-term rainfall deficits that greatly reduced surface water expression. For *P. richmondensis*, all new sites were added in the southern end of Yabbra National Park in New South Wales, which was the only area within this species range that was affected by the 2019/2020 fires. New sites were located in catchments with existing sites to ensure they remained within areas known to be occupied by *P. richmondensis* and with broadly congruent habitat conditions.

Given logistical and access constraints, only five sites were surveyed for *P. loveridgei* in the Mt Ballow area of Mount Barney National Park in Queensland. Sites were selected on the basis of the location of previous records and accessible similar habitat nearby. This population mostly occurs in remote areas limiting survey opportunities. Three of the sites were affected by fire (either along the 100 m transect or within their catchment), with two sites in the adjoining unburnt refuge area. Sites were a minimum of 300 m apart.

Table 1. Ranking of impacts of the 2019/2020 'Black Summer' fires on threatened frogs, as identified in the Commonwealth's rapid analysis of impacts (Legge *et al.* 2020). Coverage of post-fire surveys is detailed for each species, listed by contributing institutions. The three northern species of Mountain Frog (*Philoria* spp.) covered by this study are highlighted. Institutions are: UoN = University of Newcastle, NSW DPIE = Department of Planning, Industry and Environment, DAWE = Commonwealth Department of Agriculture, Water and Environment, ANU =- Australian National University, SCU = Southern Cross University, QLD DES = Department of Environment and Science, AM = Australian Museum, UNSW = University of New South Wales, UoM = University of Melbourne, VIC DELWP = Department of Environment, Land, Water and Planning.

Common name	Scientific Name	EPBC listing	IUCN listing	Score for fire overlap	Risk due 2019/2020 fires	Post fire surveys covered by
Giant Burrowing Frog (A)	*Heleioporus* sp.	Not assessed	Not assessed	4	8	UoN, NSW DPIE, DAWE
Northern Corroboree Frog	*Pseudophryne pengilleyi*	CR	EN	3	7	NSW DPIE, ANU
Mountain Frog	*Philoria kundagungan*	Not listed*	EN	3	6	ANU, SCU, NSW DPIE, QLD DES, DAWE (this study)
Pugh's Frog	*Philoria pughi*	Not listed	EN	3	6	UoN, NSW DPIE, DAWE
Sphagnum Frog	*Philoria sphagnicolus*	Not listed	EN	3	6	UoN, NSW DPIE, DAWE
Masked Mountain Frog	*Philoria loveridgei*	Not listed	EN	3	6	ANU, SCU, NSW DPIE, QLD DES, DAWE (this study)
Peppered Tree Frog	*Litoria piperata*	VU	CR	2	5	AM, UNSW
Kroombit Tinker Frog	*Taudactylus pleione*	CR	CR	1	5	QLD DES
Southern Corroboree Frog	*Pseudophryne corroboree*	CR	CR	1	5	NSW DPIE, ANU
Spotted Tree Frog	*Litoria spenceri*	EN	CR	1	5	UoM, DAWE
Giant Burrowing Frog (B)	*Heleioporus australiacus*	EN	VU	2	5	UoN, NSW DPIE, DAWE
New England Treefrog	*Litoria subglandulosa*	Not listed	VU	2	5	UoN, NSW DPIE, DAWE
Littlejohn's Tree Frog	*Litoria littlejohni*	EN	LC	3	5	UoN, NSW DPIE, VIC DELWP, DAWE
Richmond Range Mountain Frog	*Philoria richmondensis*	Not listed*	EN	1	4	ANU, SCU, NSW DPIE, DAWE (this study)
Davies' Tree Frog	*Litoria daviesae*	Not listed†	VU	2	4	UoN, NSW DPIE, DAWE
Stuttering Frog	*Mixophyes balbus*	VU	VU	3	4	UoN, NSW DPIE, DAWE
Fleay's Barred Frog	*Mixophyes fleayi*	EN	EN	1	3	SCU, GU, NSW DPIE, QLD DES
Giant Barred Frog	*Mixophyes iteratus*	EN	EN	2	4	UoN, NSW DPIE, QLD DSE, DAWE

*Under review, being recommended for uplisting to Endangered by the Commonwealth's Threatened Species Scientific Committee.
†Under review, being recommended for uplisting to Vulnerable in the recent review of Gillespie *et al.* (2020).

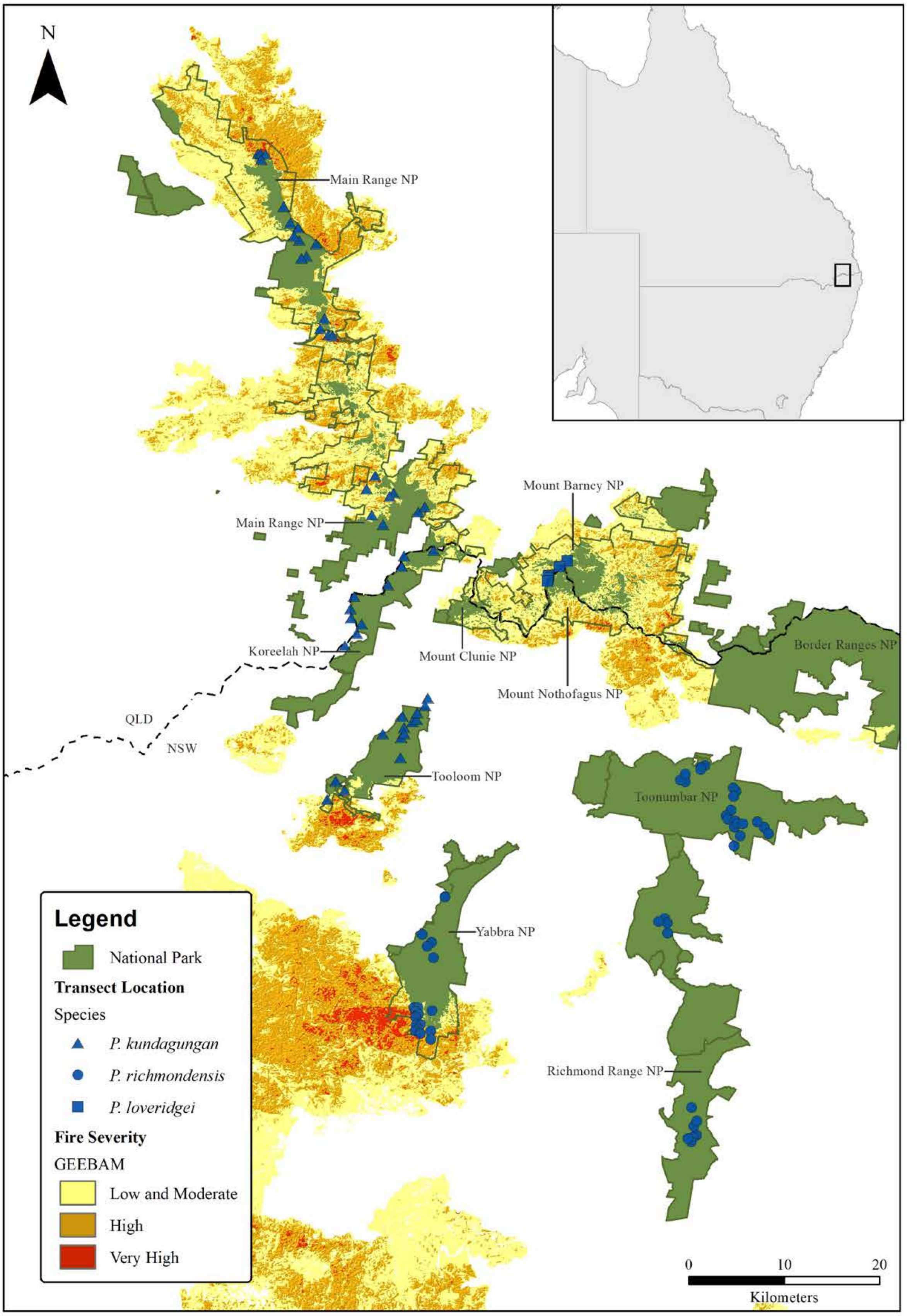


Figure 1. Map of the study area, showing locations of transects surveyed for *Philoria kundagungan, P. richmondensis* and *P. loveridgei*, with transects for the latter being restricted to the western portion of the range.

In total, 103 sites were surveyed across the study area during the 2020/2021 breeding season, encompassing 48 sites for *P. kundagungan*, 50 sites for *P. richmondensis* and five sites for *P. loveridgei*. Of these, 14 *P. kundagungan* sites were burnt in the 2019/2020 fires (29%), with 13 *P. richmondensis* sites burnt (26%) and two *P. loveridgei* sites burnt (40%).

Site surveys

Surveys were completed for calling males at each site between September 2020 and February 2021, with up to five repeat surveys completed at each site (mean = 3 surveys, total surveys = 290). Surveys were undertaken during daylight hours in all cases, as calling activity is minimal at night (Willacy *et al.* 2015; Bolitho *et al.* 2021). Surveys entailed quietly walking the transect for a minimum of 15 minutes, using either vocal call imitation or broadcasting of male advertisement calls using a hand-held speaker. Call elicitation using these methods was completed at 10 m increments along the transect. The total number of males detected calling and their position along the transect was recorded. Date of survey, start and end times and five variables thought or known to influence calling rate were measured during surveys, with the latter measured with the aid of a Kestrel weather meter and probe thermometer (various models in each case). These variables were ambient air temperature, water temperature, relative humidity, cloud cover and precipitation level.

Assessments of fire severity and habitat attributes

Fire severity at each site was scored during surveys, using a suite of measures that sought to capture the considerable variation in fire extent and severity within these headwater streams. The percentage of the stream bed that was burnt was estimated visually, as was the percentage of the bank to 10 m and 20 m that was burnt on both sides of the stream. Fire severity was also scored on an ordinal scale at these two distance classes on both sides of the stream, as either: (1) 'Unburnt'; (2) 'Low severity' (canopy and subcanopy unscorched, shrubs may be scorched, fire-sensitive low shrubs may be killed); (3) 'Moderate severity' (partial canopy scorch, subcanopy partially or completely scorched, and/or fire-sensitive tall shrub or small tree layer mostly killed); (4) 'High' (full canopy scorch to partial canopy consumption, subcanopy fully scorched or consumed), or; (5) 'Extreme' (full canopy, subcanopy and understorey consumption). The Appendix provides examples of sites burnt at varying levels of severity.

A suite of additional environmental variables was measured at each site, either as one-off assessments or during each survey. The percentage of the stream saturated was measured during each survey by pacing out saturated sections, as was the percentage of the stream with surface water and the percentage of the stream supporting clumps of Rainforest Spinach (an important microhabitat for *Philoria*; see 'Introduction'). Maximum water depth and average stream width were estimated visually during each survey, and pig damage scored as: (1) 'P0' (no recent pig damage); (2) 'P1' (recent pig damage, over a small area or very patchy); (3) 'P2' (light to moderate recent pig damage, scattered over less than 50% of the area censused), or; (4) 'P3' (moderate to heavy recent pig damage, continuously or scattered over the majority of the area censused). Water pH and total dissolved solids (TDS) was also measured at the majority of sites on each survey, dependent on availability of surface water and the relevant probes. Notes on fire impacts and other disturbances were also taken.

Remote-sensed and other spatial variables

To complement field-based measures of fire severity at each site, the Australian Google Earth Engine Burnt Area Map (GEEBAM; DAWE 2020a) was acquired as a 30 m resolution raster layer and the average burn severity (scale = 0 – 24) estimated for each site within a 20 m buffer surrounding the length of stream surveyed and within a 100 m buffer of the transect midpoint. Both were strongly correlated with burn severity measured during site assessments, with correlation coefficients of >0.9 when tested against the average burn severity (ordinal score) on the bank to 20 m.

Two remote-sensed measures of drought-severity were acquired, both of which are spectral indices from MODIS reflectance data that are known to be strongly correlated with water availability for vegetation (Caccamo *et al.* 2012). Visible Atmospherically Resistant Index (VARI; Gitelson *et al.* 2002) is sensitive to vegetation greenness, whereas the Normalised Difference Infrared Index centred on 1650 nm (NDIIb6) is sensitive to vegetation water content (Caccamo *et al.* 2012). Layers of both indices at a 30 m resolution were derived for August 2019 just prior to the 2019/2020 fires, and were intended to represent measures of drought stress just prior to that event. As for the fire severity layer, the average of each drought severity index was measured within a 20 m buffer surrounding the length of stream surveyed and within a 100 m buffer of the transect midpoint.

As an alternative measure of drought-stress, 5 km resolution gridded daily rainfall estimates for the study area from the Australian Bureau of Meteorology were collated from 01/01/2019 to 01/04/2021, encompassing the period prior to the 2019/2020 fires, plus the current survey season. Cumulative rainfall for each site was estimated for the period 01/09/2019–01/09/2020 (12 months prior to the current surveys) and 01/09/2020–01/03/2021 (current survey period). However, these measures displayed no relationship with water availability at the study sites during the survey period (correlation coefficients with average percent of stream saturated all < 0.06) and were not considered further.

Finally, a digital terrain model at 30 m resolution was acquired from Geosciences Australia. It was used to estimate the elevation of each transect (m asl) at the mid-point.

Data analysis

To assess the area of habitat that was burnt during the 2019/2020 fire season for *P. kundagungan, P. richmondensis* and western populations of *P. loveridgei*, it was first necessary to define potential habitat for each taxon. Potential habitat was defined using the approach of Bolitho *et al.* (2021). In the first instance, previous genetic data (Knowles *et al.* 2004; L. Bolitho, unpublished data) was used to delineate the mountain ranges occupied by each species. Within these broad geographic extents, the envelope of possible habitat was defined as mapped watercourses and their immediate surrounds in rainforest or within 500 m of rainforest. Spatial data defining watercourses at the 1:25,000 scale was obtained from the New South Wales and Queensland Governments. Rainforest distribution was taken from the CRAFTI Upper North East Floristics layer available from the New South Wales Government, and from Regional Ecosystem layers available from the Queensland Government. Potential habitat for each species was defined as sections of mapped streams meeting the above criteria, rasterised to 90 m × 90 m pixels to capture areas of adjoining forest likely to be utilised by *Philoria* (measured at up to 85 m from breeding sites for *P. frosti*; Hollis 2004). Overlap between fire extent and potential habitat for each species defined in this way was assessed based on the GEEBAM layer, using the 'sf' package (Pebesma 2018) to manipulate spatial layers in R version 4.0.2 (R Core Team 2021).

Analysis of site occupancy and male abundance data collected during the 2020/2021 season began with identifying variables most likely to influence these parameters. From the wider pool of variables listed above, a subset of 10 was selected based on previous studies of *P. kundagungan* (Bolitho *et al.* 2021) and *P. richmondensis* (Willacy 2014), and by inspection of the raw detection and count data. These 10 variables were elevation (m asl, the most important predictor of occupancy probability in the studies of Willacy 2014 and Bolitho *et al.* 2021), whether sites were burnt or not in 2019/2020 (binary variable), percentage of the stream bed burnt, average burn severity score (unburnt = 0, low = 1, moderate = 2, high = 3) for both stream banks, average GEEBAM burn severity within a 100 m buffer of the transect centre, percentage of stream with Rainforest Spinach, average and minimum percentage of the stream saturated, and average drought severity (NDIIb6 and VARI) within a 100 m buffer of the transect centre. An initial candidate model set was developed from these 10 variables, with elevation included in all models *a priori* (given its strong relationship with site occupancy for both species; Willacy 2014, Bolitho *et al.* 2021) and all additive combinations of the remaining variables included. This initial model set was then pruned to remove models that included variables that were correlated with each other, as defined by correlation coefficients of > 0.5. The final model set spanned 36 candidate models, with a maximum of five covariates for site occupancy and male abundance per model.

Due to the small sample of sites surveyed for western *P. loveridgei*, this taxon was excluded from the analysis. As such, we only report descriptive statistics from the five surveys completed for this taxon.

Single-season occupancy models (MacKenzie *et al.* 2006) were fitted to the survey data for *P. kundagungan* and *P. richmondensis* from the 2020/2021 season, with model structures for the probability of site occupancy (Ψ) defined by the candidate model set. The per-survey detection probability (p) was set as a function of air temperature at the time of survey for both species, with this being the key correlate of detection probability for *P. kundagungan* in the study of Bolitho *et al.* (2021). Effects of site-level variables on Ψ and air temperature on p were modelled using a linear equation and logistic-link function, following Mackenzie *et al.* (2006). Candidate models were initially fitted to the data for both species using maximum likelihood with the aid of the 'unmarked' package (Chandler *et al.* 2021) for R. The top-ranked model for each species, as judged by Akaike's Information Criteria (AIC), was subsequently refitted to the data in a Bayesian framework using Markov Chain Monte Carlo (MCMC) sampling with program JAGS (Plummer 2003) in R via the R2jags package (Su & Yajima 2020). Doing so was motivated by the desire to estimate the full posterior distribution of the parameters and to allow direct estimates of the number of occupied sites from the latent occupancy states (see below). Posterior distributions were derived from 20,000 MCMC samples in each case, after a burn-in of 20,000 samples. Convergence was assessed using the Brooks-Gelman-Rubin statistic (Brooks & Gelman 1998).

The candidate model set was fit to counts of calling males during each survey for *P. kundagungan* and *P. richmondensis* using generalised linear mixed-models (GLMMs) with a Poisson error structure. Specifically, counts during each survey were assumed to be a random variable from a Poisson distribution, with mean count for each species (λ) modelled as a function of site- and survey-level covariates using a log-linear equation. As for occupancy models, only air temperature was included as a predictor of detection rate at the time of survey. Offset parameters allowing for variation in survey effort were not included, as a standard transect length was used (100 m) and survey length varied little (91% being a standard 15 minutes). Candidate models were fitted sequentially for both species in R using the 'lme4' package (Bates *et al.* 2021), with all models including a site-based random effect to account for repeated counts at most sites. The top-ranked model for each species, as judged by AIC, was again refitted to the data using MCMC sampling in JAGS, called from R. Parameter estimates were derived from 20,000 MCMC samples after a burn-in of 20,000 samples.

Surveys conducted prior to the 2019/2020 fires at the focal sites were used to assess change in site occupancy and abundance of calling males. Change in abundance was assessed purely with the raw count data, with application of formal statistical approaches to estimate changes in abundance (specifically, N-mixture models; Royle 2004) giving grossly inflated estimates of the number of calling males per transect, due to steeply declining detection rates as the season progressed. Change in site occupancy was assessed by fitting single-season occupancy models to survey data for *P. kundagungan* obtained during the 2016/2017 breeding season (Bolitho *et al.* 2021), and to survey data obtained for *P. richmondensis* during the 2012/2013 and 2019/2020 breeding seasons (Willacy 2014; L. Bolitho and D. Newell, unpublished data). Surveys for *P. richmondensis* in 2019/2020 were completed just prior to fires affecting this species' range, as part of the DPIE SoS monitoring program. The probability of site occupancy was modelled as a function of elevation alone in the earlier seasons, with this proving the top model in the studies of Willacy (2014) and Bolitho *et al.* (2021) and measures of the additional covariates considered here being unavailable for those years. The probability of detection was modelled as a function of air temperature, as above. For each species and season, the number of occupied sites was estimated by summing the latent occupancy states following the approach of Royle and Kéry (2007). This allowed estimates of occupancy across the full set of sites surveyed in 2020/2021, with occupancy of sites not surveyed in the earlier years imputed from the relationships between occupancy and elevation in those years.

To assess the broader impacts of the 2019/2020 fires on *P. kundagungan* and *P. richmondensis*, top occupancy models fitted to the 2020/2021 survey data were used to estimate the area of occupancy for each species in this season. Estimates were produced at two spatial scales: first, across the full area of potential habitat (all 90×90 m pixels covering potential stream habitat, taking in breeding and non-breeding habitat), and second, for just the streams themselves within these pixels (representing the area of occupied breeding habitat, corresponding to the metric of Bolitho *et al.* 2021). Area of occupancy calculations were made using Monte Carlo simulations. For each of 5000 estimates of the parameters of the top occupancy models, occupancy or not of each pixel *i* of potential habitat was set as Bernoulli trial with probability Ψi, where Ψi was estimated from the values of the relevant covariates for pixel *i* and the estimated effects of these variables on Ψ. Estimates of the area of occupancy for each species were derived by summing the area of occupied pixels in each simulation (breeding and non-breeding habitat) or summing the area of stream within each occupied pixel (just breeding habitat). The mean and 95% credible intervals (95% CIs) were derived for each metric from across the 5000 simulations.

Philoria kundagungan, Main Range National Park. Image: Harry Hines

Findings

Area of habitat burnt

The area of potential habitat for *P. kundagungan* was estimated at 69.5 sq. km, of which 21.1 sq. km (30%) was burnt during the 2019/2020 fires. Of the burnt area, 71% was burnt at low–moderate severity, 27% was burnt at high severity and 2% was burnt at very high severity (based on GEEBAM classifications). For *P. richmondensis*, 11.9 sq. km of the 96.8 sq. km of potential habitat was burnt (12%), with 45% of the burnt area experiencing low–moderate severity fire, 42% experiencing high severity fire and 13% experiencing very high severity fire. The potential habitat for the western populations of *P. loveridgei* studied here was estimated at 50.1 sq. km, of which 54% was burnt in 2019/2020. The vast majority of the burnt area experienced low–moderate (78%) or high severity fire (21%). Only 1% of the burnt habitat of *P. loveridgei* experienced very high severity fire.

Outcomes of surveys

Of the 48 sites surveyed for *P. kundagungan* in the 2020/2021 season, 14 were burnt in the 2019/2020 fires. Fire severity was generally scored as low to moderate within these sites; however, five sites experienced >90% of the stream bed or banks burnt, with two of these experiencing high severity fire. Burnt sites were primarily located in Main Range National Park, with single sites burnt at low severity at each of Koreelah and Tooloom National Parks. Surveys detected *P. kundagungan* at 25 of 48 sites (52%), with the species detected at only two of the 14 burnt sites (14%) compared to 23 of the 34 unburnt sites (68%). Counts of calling males were generally low (62% of counts at occupied sites being $\leq$ 5 individuals); however, 10 or more individuals were detected on 11 occasions, with a maximum count of 22 in a single survey. Maximum counts at the two burnt sites at which *P. kundagungan* was detected were 4 and 10, while at unburnt sites counts averaged 7 individuals, with a range of 1–22 (Figure 2).

For *P. richmondensis*, 13 of the 50 sites were burnt, with all burnt sites located in lower elevation sections of Yabbra National Park. Twelve of these 13 sites were burnt at moderate to high severity, with an average of 98% of the stream bed burnt and 99.7% of the banks burnt. *Philoria richmondensis* was detected at 23 of 50 sites (46%), with detection at 3 of the 13 burnt sites (23%) compared with detection at 20 of 37 unburnt sites (54%). Counts of calling males were lower than for *P. kundagungan* overall. A higher proportion of counts at occupied sites were $\leq$ 5 individuals (82% for *P. richmondensis* vs. 62% for *P. kundagungan*), with counts of 10 or more individuals only occurring during three surveys. The maximum count during any one survey was 14 calling males (compared with 22 for *P. kundagungan*). At the three burnt sites at which *P. richmondensis* was detected, maximum counts of calling males were 1, 3 and 14, with the latter being at a site that was burnt with very low severity. Counts of *P. richmondensis* at unburnt sites averaged 5, with a range of 1–14 (Figure 2).

Two of the five sites surveyed for western populations of *P. loveridgei* were burnt. These sites burnt at low severity, although 100% of the stream bed was burnt at one of these sites. Only a single survey was completed at each site surveyed for this species, providing limited information about detections. The species was detected at four of five sites (80%), being detected at both burnt sites (100%) and two of three unburnt sites (66%). Counts of calling males at sites at which *P. loveridgei* was detected ranged from 1–6, with a mean of 3.5. At the two burnt sites, 1 and 3 calling males were detected, with 0–6 detected at the three unburnt sites (Figure 2).

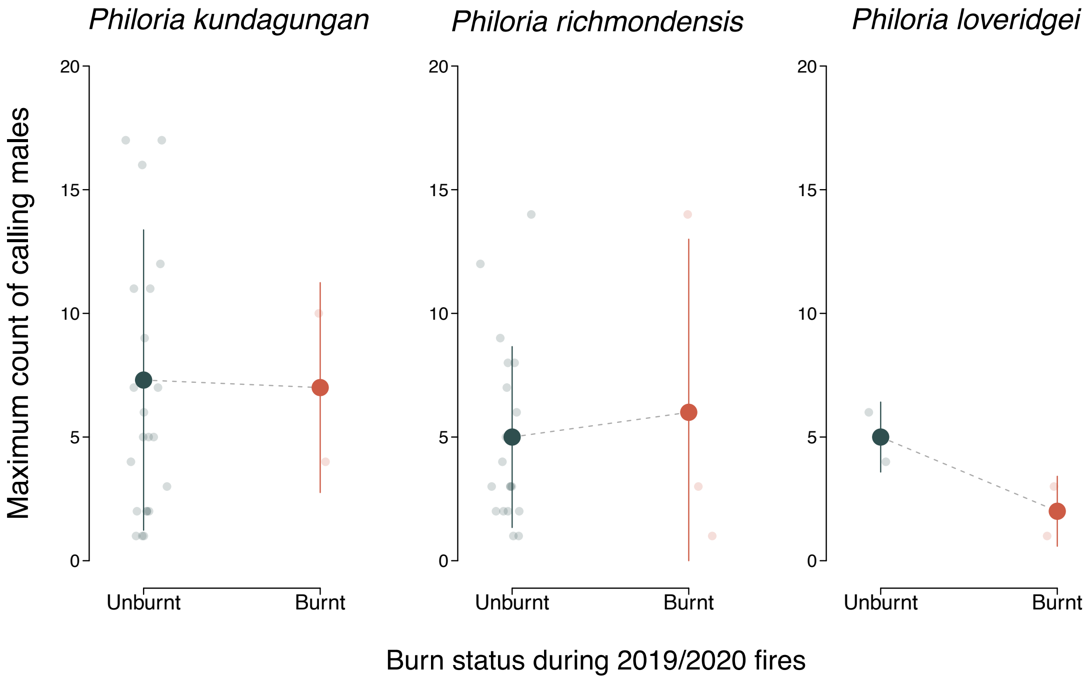


Figure 2. Maximum counts of *Philoria kundagungan, P. richmondensis* and *P. loveridgei* at burnt and unburnt sites at which these species were detected during the 2020/2021 breeding season. Raw data are shown as jittered points, with large dots showing the mean count and vertical lines ± 1 SD.

Plotting average fire severity scored in the field against maximum counts of calling males suggests counts during the 2020/2021 season declined with increasing fire severity (Figure 3). No individuals were detected at sites burnt at high severity on both banks (average score for both banks = 3), and the maximum count of calling males at sites with an average burn severity of 2 to 2.5 ('moderate' to 'moderate-high') was only four. However, sites burnt at high severity were often at low elevation (particularly the case for *P. richmondensis*) and were unlikely to be occupied for this reason.

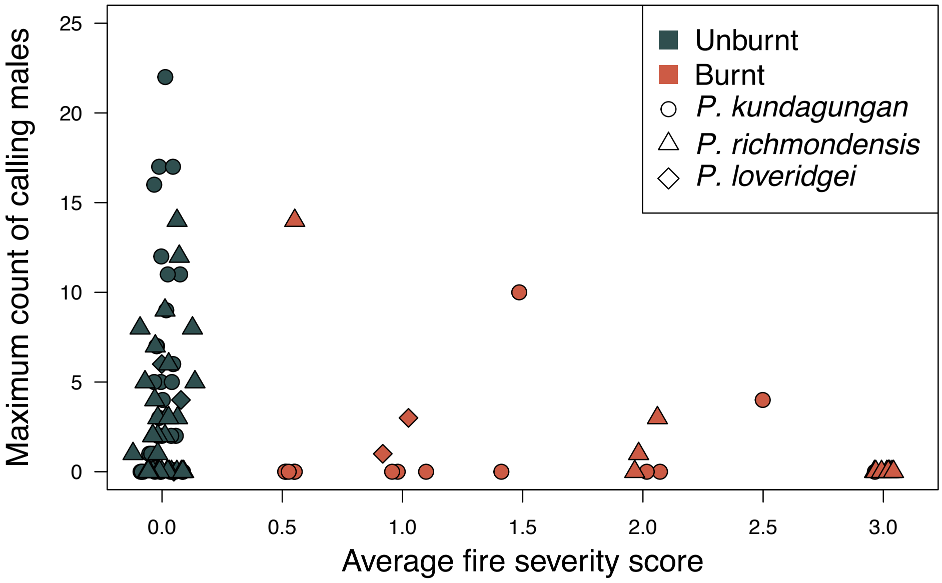


Figure 3. Maximum counts of *Philoria kundagungan, P. richmondensis* and *P. loveridgei* at sites surveyed in the 2020/2021 season plotted against average fire severity scored in the field (averaged across left and right banks, with scores being unburnt = 0, low = 1, moderate = 2, high = 3). Average fire severity scores are jittered slightly to improve readability.

Occupancy modelling

Model selection statistics for the 36 models fitted to the detection data for *P. kundagungan* and *P. richmondensis* are shown in Table 2. In addition to elevation (which was included in all models *a priori*), the top three models for *P. kundagungan* included fire effects on occupancy probability, being specifically whether a site was burnt or not during the 2019/2020 fires. Two of the three top models also included an effect of percent stream saturation during surveys on occupancy probability for *P. kundagungan*, with the second top model including an effect of remote-sensed drought severity (variable NDiib6) prior to the 2019/2020 fires. However, while drought severity displayed a negative relationship with occupancy probability as expected, the effect was weak. For *P. richmondensis*, the top three models included effects of percent stream saturation on occupancy probability in addition to elevation. Bank burn severity appeared in the third top model; however, the effect was weak and this model had substantially less support than the equivalent model excluding this effect.

Table 2. Model selection statistics for the top five models fitted to the detection data for *Philoria kundagungan* and *P. richmondensis*. Akaike's Information Criterion (AIC) is provided for each model, as is distance from the top model (ΔAIC). Elevation was included in all models *a priori*, with an effect of air temperature on detection probability also being common to all models.

Model	Log Likelihood	AIC	ΔAIC	Model Weight
P. kundagungan				
Elevation, Burn status, Percent stream saturated (average)	-50.84	115.72	0.00	0.26
Elevation, Burn status, Drought severity (NDIIb6)	-51.21	116.47	0.75	0.18
Elevation, Burn status, Cover of Rainforest Spinach, Percent stream saturated (average)	-50.64	118.07	2.35	0.08
Elevation, Bank burn severity, Percent stream saturated (average)	-52.31	118.67	2.95	0.06
Elevation, Burn status, Cover of Rainforest Spinach, Drought severity (NDIIb6)	-51.03	118.85	3.13	0.05
P. richmondensis				
Elevation, Percent stream saturated (average)	-52.07	115.51	0.00	0.29
Elevation, Percent stream saturated (minimum)	-52.88	117.13	1.62	0.13
Elevation, Bank burn severity, Percent stream saturated (average)	-51.95	117.85	2.34	0.09
Elevation, Burn status, Percent stream saturated (average)	-52.05	118.06	2.56	0.08
Elevation, Cover of Rainforest Spinach, Percent stream saturated (average)	-52.06	118.07	2.57	0.08

Estimated effects of the variables in the top models for *P. kundagungan* and *P. richmondensis* are provided in Table 3, with relationships between these variables and the probability of occupancy depicted in Figure 4. Occupancy probability rose steeply with elevation for *P. kundagungan*, with the effect moderated by fire, such that burnt sites at low and mid elevations had a substantially lower probability of occupancy than unburnt sites at the same elevations. The same was true for stream saturation extent, with substantially higher probabilities of occupancy over the range of this variable for unburnt habitat. Occupancy probabilities for *P. richmondensis* also rose steeply with elevation; however, the effect was weaker and more uncertain than for *P. kundagungan*. For *P. richmondensis* there was a clearer and stronger effect of stream saturation extent, which proved the strongest predictor of occupancy probability for this species during the 2020/2021 season.

Table 3. Coefficient estimates from the top models of site occupancy for *Philoria kundagungan* and *P. richmondensis*. Mean estimates are provided along with the 95% credible interval (95% CI).

Variable	Covariate of	*P. kundagungan*		*P. richmondensis*	
		Mean	95% CI	Mean	95% CI
Intercept	Occupancy	5.07	0.81, 11.54	0.92	-0.65, 3.33
Elevation	Occupancy	12.01	3.24, 25.48	3.68	0.13, 8.51
Burn status	Occupancy	-7.63	-16.24, -1.89	-	-
Percent of stream saturated	Occupancy	4.42	0.56, 9.55	6.76	3.10, 12.35
Intercept	Detection	-0.01	-0.19, 0.57	0.06	-0.55, 0.66
Air temperature	Detection	-2.24	-3.51, -1.13	-2.32	-3.48, -1.30

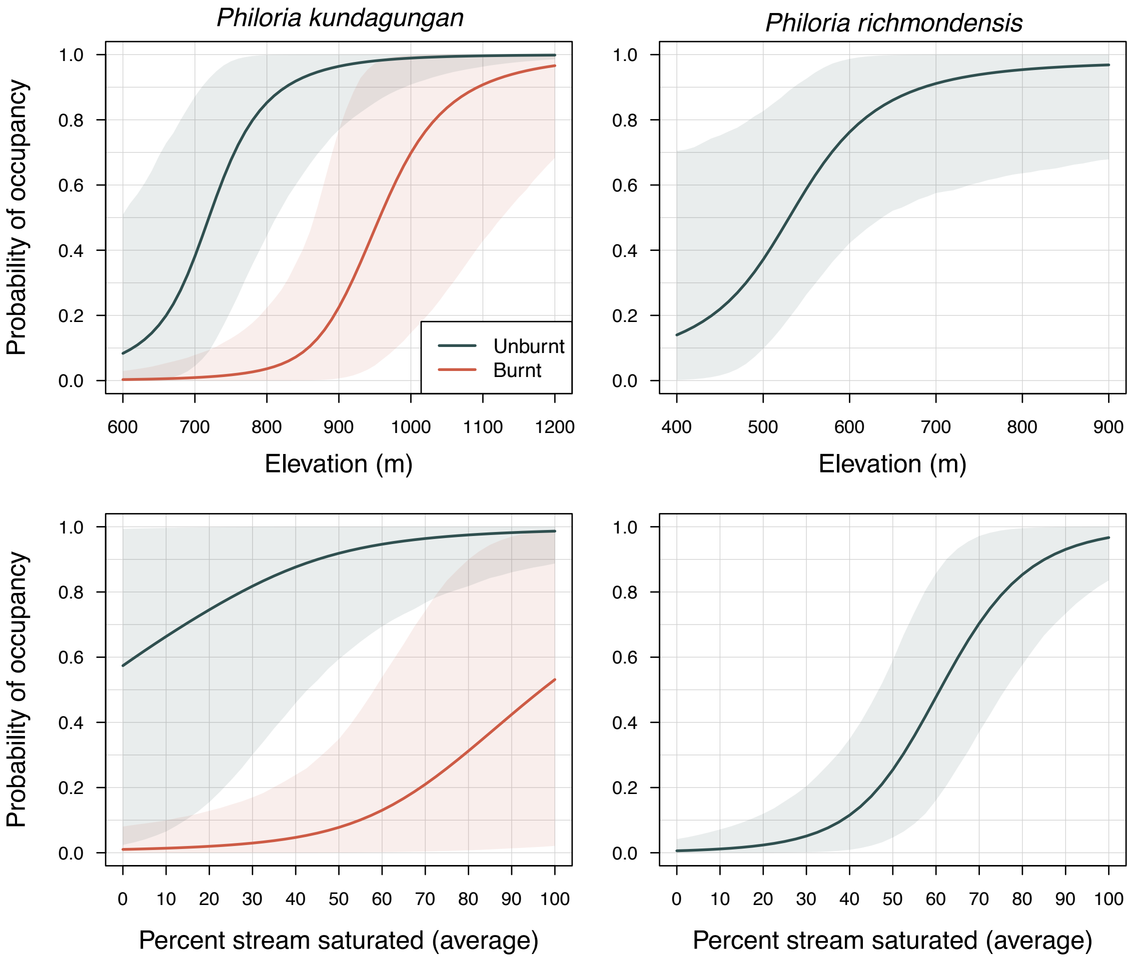


Figure 4. Relationships between the probability of site occupancy and both elevation and percentage of stream saturated for *Philoria kundagungan* and *P. richmondensis* during the 2020/2021 breeding season. Relationships are depicted for burnt and unburnt sites for *P. kundagungan*, with burn status being a key predictor of occupancy probability for this species. Relationships for elevation and percentage of stream saturated are shown with the other covariate held at its mean. Shaded areas are 95% credible intervals.

Abundance modelling

Fitting the 36 candidate models to the count data for *P. kundagungan* and *P. richmondensis* led to very similar model selection rankings as those obtained for the occupancy models for both species (Table 4). Burn status during the 2019/2020 fires again featured in the top model for *P. kundagungan*, along with elevation and percent stream saturated. However, the effect of burn status was weaker than that for the probability of occupancy, with the model including burn status being only slightly superior to the second ranked model which excluded this effect (Table 4), and the 95% credible interval (95% CI) for the effect of burn status on counts overlapped zero (Table 5). Burn status also featured in the top model for *P. richmondensis*; however, the effect was weakly positive (Table 5) as a result of an outlying count of 14 calling males along one very lightly burnt site (see Figure 2 & 3). Given this effect was weak, counter to the expected ecological relationship and the result of an outlier at a site that was effectively unburnt, this result is considered spurious. For both *P. kundagungan* and *P. richmondensis*, counts of calling males increased with elevation and percent stream saturated, and declined with increasing air temperature at the time of survey (Table 5; Figure 5). Air temperature was included in all models given its known effects on calling activity for these species.

Table 4. Model selection statistics for the top five models fitted to the count data for *Philoria kundagungan* and *P. richmondensis*. Akaike's Information Criterion (AIC) is provided for each model, as is distance from the top model (ΔAIC). Air temperature (not shown) was included in all models given its effect on calling rate.

Model	Log Likelihood	AIC	ΔAIC	Model Weight
P. kundagungan				
Elevation, Burn status, Percent stream saturated (average)	-209.70	432.03	0.00	0.29
Elevation, Percent stream saturated (average)	-211.02	432.50	0.47	0.23
Elevation, Cover of Rainforest Spinach, Percent stream saturated (average)	-210.66	433.95	1.93	0.11
Elevation, Bank burn severity, Percent stream saturated (average)	-210.66	433.95	1.93	0.11
Elevation, Burn status, Cover of Rainforest Spinach, Percent stream saturated (average)	-209.62	434.10	2.07	0.10
P. richmondensis				
Elevation, Burn status, Percent stream saturated (average)	-156.21	325.02	0.00	0.26
Elevation, Percent stream saturated (average)	-157.42	325.28	0.26	0.23
Elevation, Burn status, Cover of Rainforest Spinach, Percent stream saturated (average)	-155.92	326.64	1.63	0.12
Elevation, Bank burn severity, Percent stream saturated (average)	-157.36	327.32	2.30	0.08
Elevation, Cover of Rainforest Spinach, Percent stream saturated (average)	-157.36	327.33	2.31	0.08

Table 5. Coefficient estimates from the top models of counts of *Philoria kundagungan* and *P. richmondensis*. Mean estimates are provided along with the 95% credible interval (95% CI).

Variable	*P. kundagungan*		*P. richmondensis*	
	Mean	95% CI	Mean	95% CI
Intercept	-1.22	-2.14, -0.52	-2.03	-2.88, -1.33
Elevation	2.09	0.64, 3.90	1.23	0.46, 2.11
Burn status	-1.27	-3.02, 0.15	0.78	-0.48, 1.93
Percent of stream saturated	2.49	1.22, 3.94	3.42	0.72, 4.94
Air temperature	-1.68	-2.09, -1.30	-2.01	-2.61, -1.46

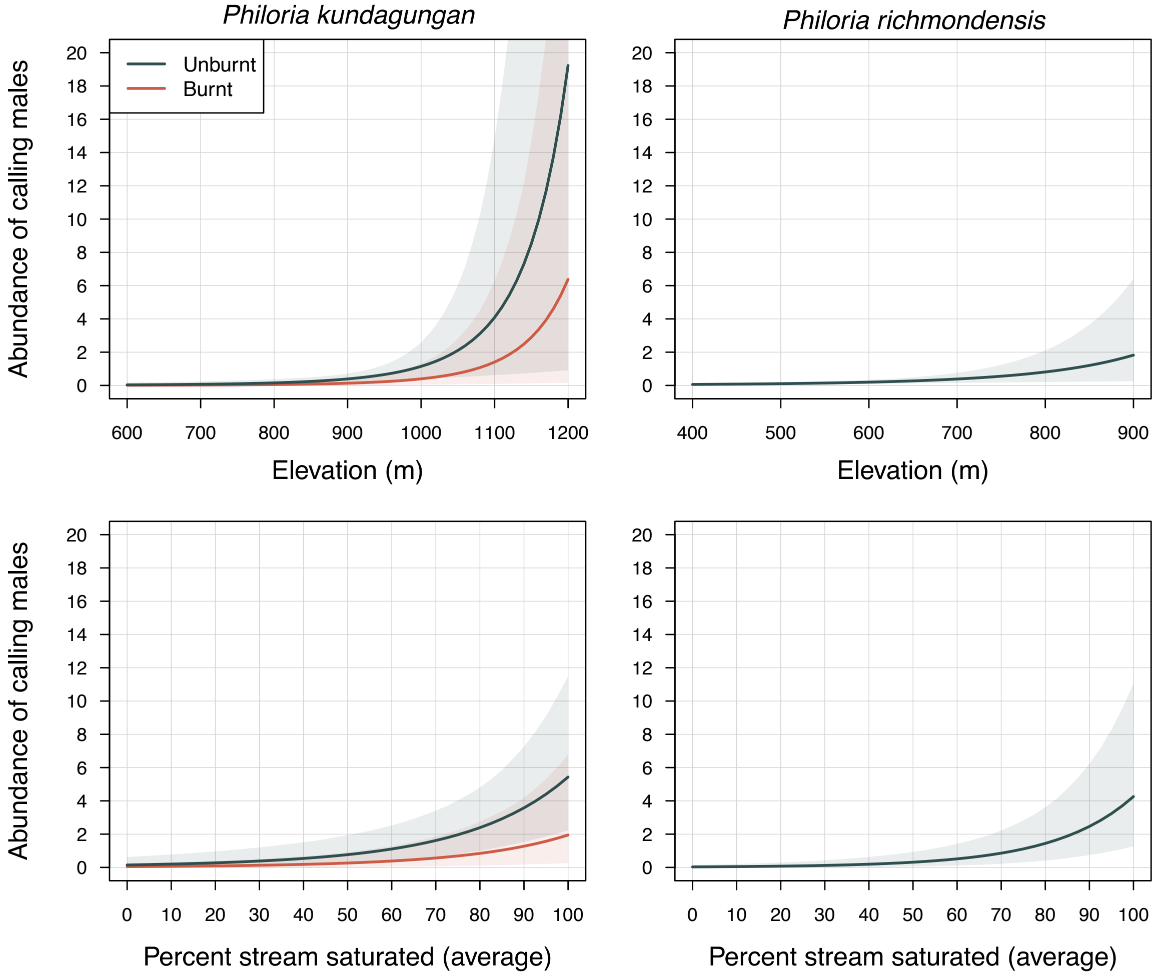


Figure 5. Relationships between counts of calling males and both elevation and percentage of stream saturated for *Philoria kundagungan* and *P. richmondensis* during the 2020/2021 breeding season. Relationships are depicted for burnt and unburnt sites for *P. kundagungan*, with burn status being an additional predictor of counts for this species. Relationships for elevation are shown with percent of stream saturated held at its mean, whereas relationships for percent stream saturated are shown when elevation is held at 1000 m for *P. kundagungan* and 800 m for *P. richmondensis*. Shaded areas are 95% credible intervals.

Comparisons with surveys prior to the 2019/2020 fires

Figure 6 compares the results of surveys prior to the 2019/2020 fires with those undertaken in 2020/2021. When considering all 48 sites surveyed for *P. kundagungan* in the 2020/2021 season, the number of occupied sites was estimated to have declined from 31 in 2016/2017 to 27 in 2020/2021 (a decline of ~13%; Figure 6). However, when considering only the subset of sites surveyed in both seasons (35), occupancy rates were similar, indicating that the larger decline predicted for all 48 sites stems from imputed occupancy during 2016/2017 for many of the sites added in 2020/2021. Nevertheless, maximum counts of calling males at occupied sites were lower on average in 2020/2021 (mean = 11 in 2016/2017 vs. 7 in 2020/2021). Likewise, the summed maximum counts of calling males for the 35 sites surveyed in both seasons – a standardised measure of total detections – was substantially lower in 2020/2021 than 2016/2017 (149 vs. 247).

For *P. richmondensis*, site occupancy and counts of calling males were substantially lower during surveys completed just prior to the 2019/2020 fires than in either the 2012/2013 or 2020/2021 seasons (Figure 6). Estimated site occupancy among all 50 sites surveyed for this species was 28 in 2012/2013 and 2020/2021, but dropped to 21 in the intervening season (a decline and subsequent recovery of 28%). This pattern was essentially identical when only considering the subset of 37 sites surveyed in all three seasons. Maximum counts of calling males at occupied sites were largely consistent between the three seasons for *P. richmondensis*, although the average was slightly lower in 2019/2020 and 2020/2021 than 2012/2013, and the highest counts of 15 and 18 males along a transect in 2012/2013 was not matched in either of the subsequent seasons. Summed maximum counts of calling males for the 37 sites surveyed in all seasons was significantly depressed in 2019/2020 relative to 2012/2013, and although recovering in 2020/2021, remained 14% lower than in 2012/2013 (at 100 vs 116; Figure 6).

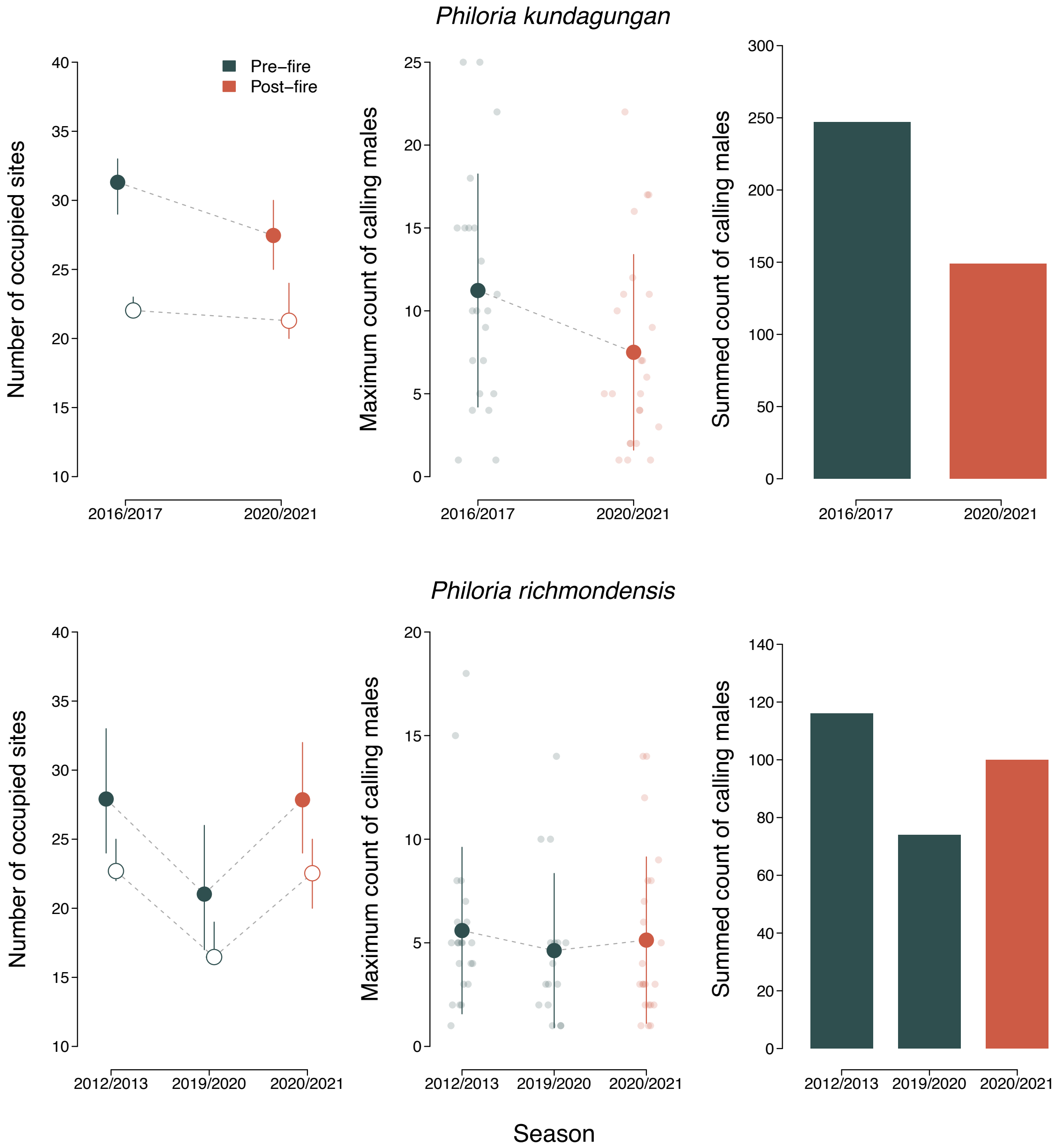


Figure 6. Comparison of survey data for *Philoria kundagungan* and *P. richmondensis* from the 2020/2021 season with that available from prior to the 2019/2020 fires. The left-hand plots show estimates of the number of occupied sites in each season, among all sites surveyed in 2020/2021 (filled dots) and among the subset of sites surveyed in all seasons (open dots). Occupancy estimates for sites not surveyed in the earlier seasons are imputed from the relationship between occupancy probability and elevation in those years for each species. Vertical lines are 95% credible intervals. The middle plot shows maximum counts of calling males at occupied sites in each season. Raw data are shown as jittered points, with large dots showing the average count and vertical lines ± 1 standard deviation. The right-hand plots show the summed maximum counts from across the subset of sites surveyed in each season, giving a standard measure of overall detections in each year.

Area of occupancy estimates after the 2019/2020 fires

Area of occupancy for both *P. kundagungan* and *P. richmondensis* in 2020/2021 was estimated both including and excluding potential non-breeding habitat. For *P. kundagungan*, the area of occupancy including non-breeding habitat was estimated to be 46.35 km^2 (95% CI = 37.49–53.95 km^2), with the equivalent figure for stream breeding habitat being 13.46 km^2 (95% CI = 10.83–15.61 km^2). For *P. richmondensis*, area of occupancy including potential terrestrial habitat was 40.43 km^2 (95% CI = 22.70–63.61 km^2), with that for stream breeding habitat being just 6.80 km^2 (3.93–10.42 km^2).

Discussion

Impact of the 2019/2020 fires

This study provides critical insights into the impact of the 2019/2020 'Black Summer' fires on three of the five northern *Philoria* species: *P. kundagungan, P. richmondensis* and western populations of *P. loveridgei*. All are restricted to rainforests and their ecotones; environments that are highly sensitive to fire, and which very rarely experience it. However, this study found that extensive areas of the habitat of these three taxa were affected by 2019/2020 fires. Some 30% of potential habitat was burnt for *P. kundagungan*, 12% for *P. richmondensis* and 54% for the western populations of *P. loveridgei* studied here. Remote-sensed mapping suggests the majority of affected areas burnt at low to moderate severity; however, 29% of burnt habitat for *P. kundagungan* experienced high or very high severity fire according to this mapping, with the equivalent figures for *P. richmondensis* and *P. loveridgei* being 55% and 22%, respectively.

Field surveys during the 2020/2021 breeding season suggest northern *Philoria* have some resilience to fire, with persistence observed for each of the focal species at burnt sites. Two of 14 burnt sites were occupied for *P. kundagungan*, 3 of 13 burnt sites for *P. richmondensis* and 2 of 2 burnt sites for *P. loveridgei*. Relatively high counts of calling males at burnt sites were also observed for *P. kundagungan* (maximum = 10) and *P. richmondensis* (maximum = 14), and calling males of both these species were detected at sites that experienced moderate fire severity. It is clear, therefore, that northern *Philoria* can survive fire events, and will attempt to breed in burnt habitat when suitable conditions return (although this does not discount longer-term effects of fire - see below).

Nevertheless, this study does provide evidence of important effects of fire on northern *Philoria*. Occupancy probabilities and counts of *P. kundagungan* were lower in habitat burnt during the 2019/2020 season. There was also some indication of lower counts of calling males at burnt sites for *P. loveridgei*, although sample sizes were very low for this species (see Figure 2). The lack of an effect of fire for *P. richmondensis* either for occupancy rates or counts could be interpreted as greater resilience to fire than the other two species; however, we caution that this conclusion cannot be reliably drawn from the survey data reported here. Only a small portion of suitable habitat for *P. richmondensis* experienced fire in 2019/2020 (12%), the majority of which was in low elevation areas that display low occupancy rates and abundance. It is likely, therefore, that the absence of an effect of fire on occupancy rates or counts for *P. richmondensis* is the result of low statistical power to detect these effects, in turn a result of the specific nature and extent of the fire that impacted this species' habitat in 2019/2020.

The role of drought

Differentiating the respective roles of fire and drought in the patterns of occurrence and abundance of northern *Philoria* during the 2020/2021 season is difficult, with rainfall deficits being an important driver of fire within *Philoria* habitat. Nevertheless, it is clear that the extent and severity of drought are important determinants of the occurrence and abundance of the species studied here. While remote-sensed drought variables measured just prior to the 2019/2020 fires did not prove useful predictors of the probability of occupancy or counts during the 2020/2021 season, field estimates of stream saturation extent were strongly positively correlated with both parameters for *P. kundagungan* and *P. richmondensis*. It can be concluded from these relationships that both *P. kundagungan* and *P. richmondensis* were much more likely to reproduce in streams with higher saturation in 2020/2021. In turn, it may be inferred that the capacity of both species to recover from the drought and fires experienced during 2019/2020 was moderated across the range by water availability within their breeding habitat (headwater seepages), controlled ultimately by winter and spring precipitation.

The importance of drought in moderating breeding success (and ultimately persistence) for northern *Philoria* is further supported by the monitoring data compiled for *P. kundagungan* and *P. richmondensis* in recent years. Relative to surveys in the 2012/2013 season (Willacy 2014), occupancy rates and counts of calling males were significantly depressed for *P. richmondensis* in 2019/2020 just prior to the 'Black Summer' fires and during a period of intense drought. Rainfall deficits across the Gondwana Rainforests of Australian World Heritage Area were severe in 2019, leading to extensive drying of headwater streams across the region (L. Bolitho, H. Hines, D. Newell, pers. obs.). However, La Niña conditions during 2020 brought significant winter and spring rainfall across the range of *P. richmondensis*, with a concomitant recovery in occupancy rates and counts of this species in 2020/2021 (Figure 6). For *P. kundagungan*, significant rainfall deficits extended into the 2020/2021 season across much of the range, with extensive rainfall not falling until December or January. Based upon data for *P. richmondensis*, the reduced occupancy rate and counts of calling males during 2020/2021 relative to those in 2016/2017 (Figure 6) are likely a result of continued drought conditions, in addition to the 2019/2020 fires. The recovery of *P. richmondensis* during 2020/2021 from the previous season suggests northern *Philoria* may skip breeding during drought, with the lower occupancy and abundance observed in 2020/2021 for *P. kundagungan* potentially explained by this phenomenon.

Drought and fire are key threats under climate change

This study confirms that the increasing frequency and severity of drought and wildfires projected under climate change (Lucas *et al.* 2007; Sharples *et al.* 2016; Abram *et al.* 2021) represent important threats to northern *Philoria*. Primary evidence gathered during this study is three-fold: (i) the negative effect of fire on the occupancy rate and counts of *P. kundagungan* following the extensive wildfires of 2019/2020; (ii) the clear positive effect of stream saturation on occupancy and calling male abundance for *P. kundagungan* and *P. richmondensis* in 2020/2021, and; (iii) the correspondence between fluctuations in occupancy and calling activity for *P. kundagungan* and *P. richmondensis* and rainfall deficits in recent years. Fire and drought can suppress the extent and success of breeding activity in these frogs, with obvious implications for persistence long-term with these two perturbations likely to increase in frequency over the remainder of this century.

Additional impacts are like to arise from the impacts of drought and fire on *Philoria* habitat. Ecosystem state change from rainforest to sclerophyll forest is possible from these two processes, particularly where rainforest canopy trees are consumed or suffer high mortality rates. Canopy loss will increase insolation and temperatures, leading to higher rates of evaporation and hastening stream and substrate desiccation. Moreover, as calling activity by northern *Philoria* declines sharply with increasing temperatures (Willacy 2014; Willacy *et al.* 2015; Bolitho *et al.* 2021; this study), canopy loss could shorten the breeding season of these frogs. Numerous other ecological mechanisms linking habitat change driven by drought and fire to the decline of northern *Philoria* are plausible, including loss of microhabitats such as saturated leaf-litter, rotting fallen timber and deep humus layers, as well as breakdown of humus-associated food chains.

Other threatening processes

Surveys over recent years have identified several additional threatening processes for northern *Philoria*. The first is the fundamental rarity of the species. Small populations are inherently susceptible to stochastic extinction, either through perturbations or demographic stochasticity (Soulé 1987). Monitoring of *P. kundagungan* and *P. richmondensis* has consistently identified small choruses of calling males during surveys, with maximum counts of 25 males over a 100 m transect for *P. kundagungan* and 22 males over this distance for *P. richmondensis* (and, notably, these transects often cover the entire area of suitable habitat in headwater streams). Surveys this season, although limited, detected low numbers of calling males of the western populations of *P. loveridgei* studied here, with a maximum of 6 during a single survey. The highly fragmented nature of the upland rainforest habitat of these species suggests that their range is characterised by numerous small, largely discrete populations. As such, stochastic extinction of local populations represents a significant threat to the persistence of all northern *Philoria*.

Feral pigs were recognised as a key threat to northern *Philoria* by Gillespie *et al.* (2020). During the 2020/2021 season, damage to stream beds from feral pigs was detected at 8 sites, all within the range of *P. kundagungan*, and specifically, all within the Main Range National Park in Queensland. Feral pigs are also known from Yabbra National Park within the range of *P. richmondensis* (L .Bolitho and D. Newell, unpublished) and from Mount Barney National Park (H. Hines, H. McCall, pers. obs.) in the range of *P. loveridgei*, although impacts on sites occupied by these species were not detected during the 2020/2021 season. Pig activity may be particularly detrimental to *Philoria*. Adults and nests may be preyed on directly, and the uprooting feeding behaviour of pigs can cause significant damage to rainforest streams, destroying the subterranean structures and seepages on which *Philoria* rely for reproduction (Anstis 2013).

Feral cattle also pose a threat to northern *Philoria*, as does unauthorised cattle grazing inside conservation reserves. Unauthorised cattle grazing was observed during the 2020/2021 season in Mount Barney National Park in close proximity to sites occupied by *P. loveridgei* (H. Hines, H. McCall, pers. obs.). Evidence of cattle well inside Main Range National Park was also observed, including in unburnt rainforest at an elevation of 950 m which was occupied by *P. kundagungan* (H. Hines, H. McCall, pers. obs.).

Post-fire weed invasion represents a threat for northern *Philoria* in burnt habitat. Various weed species were observed within or near survey sites, with high-threat weeds including Lantana (*Lantana camara*), Mist Flower (*Ageratina riparia*) and Crofton Weed (*Ageratina adenophora*). In addition to altering habitat structure and possibly subterranean hydrology and water availability, these weeds may also exacerbate future fire threat (particularly the case for Lantana, which also promotes Bell Miner associated dieback of Eucalypts).

Application of research

This study has important implications for the conservation response to the 2019/2020 fires for *P. kundagungan, P. richmondensis* and western populations of *P. loveridgei*, as well as medium-term conservation priorities for these species.

Conservation status

This study supports the listing of *P. kundagungan* and *P. richmondensis* as Endangered under the Commonwealth EPBC Act 1999. Western populations of *P. loveridgei* should be the focus of a dedicated monitoring program using the methods outlined in the current study to ascertain their status after being significantly affected by the 2019/2020 fires.

Protection from fire in the near term

Recovery of each of the northern *Philoria* species from the 2019/2020 fires is dependent on the maintenance of unburnt refuges and recovery of burnt habitat. Fire suppression is a high priority in the short-to-medium term to allow demographic recovery of *Philoria* within occupied sites, to stabilise exposed soils in burnt habitat and allow re-establishment of rainforest flora, and to facilitate recolonisation of sites that suffered local extinction during the 2019/2020 fires and drought event. Careful consideration of fuel reduction burning adjacent to habitat of northern *Philoria* is required to protect unburnt refuges and reduce the likelihood of further impacts on burnt habitat, either from wildfire or fuel reduction burning itself. Direct risks from fuel reduction burning could arise from poorly selected fuel reduction zones or escape of fuel reduction burns in sclerophyll forest adjoining *Philoria* habitat. Hence, it is imperative that fire management planning within the range of northern *Philoria* consider these species. Occupancy models developed during this study could be directly applied to these planning exercises, providing spatial layers of the probability of occurrence across the study region that can be used to identify key unburnt refuges and burnt habitat with highest probability of recolonisation. Short-term priorities for protection from fire are:

1. *P. kundagungan*: Main Range National Park, including high-elevation portions of the Mistake Mountains and Goomburra section, high-elevation areas in the vicinity of Cunningham's and Spicer's Gaps and large unburnt remnants in the Mount Roberts area; Mount Clunie National Park, including unburnt and adjoining burnt habitat at high-elevations; Tooloom National Park, particularly extensive unburnt refuge habitat in the northern block.
2. *P. richmondensis*: Yabbra National Park, particularly higher-elevation areas in the southern and central sections (including burnt and adjoining unburnt refuge habitat); Toonumbar National Park, covering the extensive unburnt refuge habitat in the central and eastern sections; Richmond Range National Park, particularly unburnt, high elevation sites in the central and southern sections.
3. *P. loveridgei* (western population): Mount Barney National Park, particularly unburnt refuges and adjoining burnt habitat at high elevation in the Mount Ballow area.

Protection from ungulates and weeds

Effective feral pig control is urgently required in the northern sections of Main Range National Park (Goomburra and Mistake Mountains) where incursions into *P. kundagungan* habitat were detected during this study, and which suffered relatively high impacts from the 2019/2020 fires. Similarly, feral pig control is needed at Yabbra National Park for *P. richmondensis* and the Burnett Creek catchment of Mount Barney National Park for western *P. loveridgei*. Each of these areas were significantly affected by the 2019/2020 fires. Pig surveillance is required across the range of northern *Philoria* given the expanding distribution of these pests, and their capacity to cause significant damage to in-stream breeding habitat.

Protection of key refuge habitat from feral or domestic cattle is also urgently required in some areas. Priorities highlighted by this study are establishing exclusion fencing in Main Range National Park in the Emu Creek area and the Burnett Creek catchment in Mount Barney National Park.

Post-fire weed control in and adjacent to burnt *Philoria* habitat should be pursued. Lantana control is a particular priority in this regard, as it threatens rainforest regeneration, promotes the likelihood of Bell Miner Associated Die Back (in Eucalypt-dominated forests on rainforest margins) and exacerbates further fire risk.

Captive populations, head-starting and re-introduction

While this study demonstrates some resilience to drought and fire among northern *Philoria*, particularly evidenced by the recovery occupancy rates and counts of calling males of *P. richmondensis* in 2020/2021, it is clear that northern *Philoria* are sensitive to these events. Ongoing and cumulative impacts are likely under climate change, with resulting population declines and range contractions.

Establishing captive populations of northern *Philoria* is advisable as part of the suite of management initiatives needed to conserve these species. Captive husbandry protocols for these frogs are likely to be highly specialised, in which case considerable experimentation may be required to establish viable captive populations. However, recent advances in the captive breeding program for *P. frosti* in Victoria (McFadden *et al.* 2018) provides a framework for husbandry protocols for northern *Philoria*. Establishing captive populations prior to further declines provides adequate lead time to refine husbandry protocols and allows the widest possible array of genetic diversity to be included in the founding stock. As such, establishing captive populations should be pursued as a matter of urgency.

Similarly, consideration should be given in future to 'head-starting' and re-introduction to bolster declining populations or re-establish those that have gone locally extinct. Head-starting entails collecting egg masses, raising tadpoles to metamorphosis and re-releasing these individuals. It is being practised for *P. frosti* and represents a key component of the conservation strategy for that species (McFadden *et al.* 2018). Re-introduction may be trialled in locations where natural recolonisation is considered unlikely, due to dispersal barriers or distance to remnant populations.

Ongoing monitoring

This study, along with the previous studies of Willacy (2014) and Bolitho *et al.* (2021), has demonstrated protocols for monitoring northern *Philoria* that deliver robust estimates of population change. Ongoing monitoring will be vital for *P. kundagungan, P. richmondensis* and *P. loveridgei*, with each species facing multiple threatening processes. Conservation initiatives such as those listed above require careful planning underpinned by up-to-date knowledge of occupancy and abundance. The core set of sites established through this and earlier studies should continue to be monitored, with sites added to increase the probability of detecting a significant change in occupancy between seasons (particularly for western populations of *P. loveridgei*, for which the current set of sites is inadequate). In combination with clear trigger points for specific conservation actions, ongoing monitoring following the protocols described here provides a foundation for recovery planning for northern *Philoria*.

Philoria richmondensis, Toonumbar National Park. Image: David Newell

Impact of the research

Greatly expanded knowledge of distribution

Prior to 2017, *P. kundagungan* was known from only 8 locations and considered Data Deficient in New South Wales. Surveys over recent years, including those in 2020/2021, have greatly expanded our knowledge of the occurrence of this species. It is now known to be one of the rarest frog species in New South Wales, and has been moved to the site-management stream of the New South Wales Government's 'Saving our Species' program (indicative of the fact that all remaining sites in New South Wales are critical to its persistence). Surveys for *P. kundagungan* and *P. richmondensis* have also enabled occupancy models to be developed that provide spatial layers of habitat occupancy probabilities for both species, which are available to managers for the purposes of identifying core habitat and habitat linkages.

Development of a monitoring framework

The monitoring approach applied here provides a robust framework under which the occupancy rates of these narrow-range endemics can be monitored in response to climate change impacts. The framework has been developed in partnership with the New South Wales 'Saving our Species' and has been crucial in assessing the conservation status of *P. kundagungan* and *P. richmondensis* both at State and Commonwealth levels.

Informing review of EPBC listing and development of conservation advice

This project, including the surveys in 2012/2013, 2016/2017 and 2019/2020, have been used by DAWE to guide review of the listing of *P. kundagungan* and *P. richmondensis* under the EPBC Act and have informed development of conservation advice to the Minister for both species.

Informing assessment of fire impacts on biodiversity

The surveys completed in 2020/2021 have been a crucial component of fire impact assessments within the Gondwana Rainforests of Australian World Heritage Area completed by the New South Wales Department of Planning, Industry and Environment and the Queensland Department of Science and Environment. The project also represents an important collaboration between these State agencies, the University sector and the Commonwealth Department of Agriculture, Water and Environment in assessing fire impacts and post-fire recovery.

Identifying feral ungulate incursions

Our surveys provide evidence of pig incursions in key *Philoria* habitat. Past surveys have been used by NSW NPWS to inform pig control in Yabbra National Park. Those completed during 2020/2021 will support QPWS in pig control efforts in the Main Range and Mount Barney National Parks.

Outreach

Dr David Newell contributed to the 'Evidence based planning for resilient World Heritage Areas' webinar as part of the 'Climate Change, Fire, and Biodiversity' series run by the NESP Threatened Species Recovery Hub. This webinar allowed dissemination of recent findings on the conservation requirements of northern *Philoria* to a multi-disciplinary audience of stakeholders in the management of the Gondwana Rainforests of Australia World Heritage Area.

Broader implications

This study is one of very few to assess bushfire impacts on Australian frogs. It suggests northern *Philoria* have some resilience to fire, with persistence in burnt sites, including those burnt at moderate to high severity. However, important impacts of fire were demonstrated, including a lower rate of occupancy and calling male abundance at burnt sites.

It is imperative that the results of this study are combined with those of the suite of projects initiated following the 2019/2020 fires (those listed in Table 1 and others) to provide a synthetic view of the impacts of these fires on Australia's threatened frogs, and to realise the opportunity these studies provide to complete a meta-analysis of the impacts of bushfire on Australian frogs. This understanding will be crucial to conservation planning for Australia's frogs under climate change, with the increasing frequency and intensity of bushfire being a key mechanism predicted to drive future population declines (Gillespie *et al.* 2020).

Future research priorities

Continue long term monitoring, with an expanded set of sites

Continuing long-term monitoring for northern *Philoria* is vital to track population trajectories and to guide conservation interventions for these species. Monitoring should continue using the protocols applied here, allowing robust estimates of change in site occupancy and counts of calling males through time. The approach should be extended to *P. loveridgei*, particularly in fire-affected sections of its range. Power analysis should be pursued to identify the number of additional sites required to detect change in occupancy, following the approach of Bolitho *et al.* (2021).

Complete spatial prioritisation analyses

The occupancy models developed for *P. kundagungan* and *P. richmondensis* could be used to complete spatial prioritisation analyses that identify core habitat for these species following the 2019/2020 fires. This habitat could represent targets for post-fire management, including fire suppression, control or exclusion of ungulates and weed control (as above). Potential to project changes in habitat suitability under climate change could also be pursued, with spatial prioritisation analyses in this context identifying likely refugia for longer-term targeting of conservation investment in northern *Philoria*.

Complete detailed autecological studies

The ecology of each species of northern *Philoria* remains very poorly known. Detailed autecological studies are required on each species, with focus on compiling the demographic data needed to build population viability analyses. Studies that determine clutch sizes and tadpole, metamorph and adult survival rates are of particular importance. Intensive monitoring of nests will be required, with studies of skeletochronology (from toe-clipping adults in nests) a means of estimating adult survival rates (as has been completed for *P. frosti* in Victoria; Hollis 2003).

Clarify impacts of chytridiomycosis

Chytridiomycosis, caused by the amphibian chytrid fungus, has been an important driver of the decline of Australia's montane frogs, including *P. frosti* in Victoria (Hollis 2011; Hunter *et al.* 2018). However, the impact of the pathogen on northern *Philoria* is uncertain. Initial field studies on the prevalence of chytrid infections has been undertaken (L. Bolitho, D. Newell, unpublished). Expanded field sampling should be pursued, with laboratory studies on susceptibility of the species required.

Assess climate buffering

Projections of climate change impacts on northern *Philoria* rely crucially on understanding how changes in the macroclimate (which may be predicted from climate models) alter the microclimate experienced by the frogs. Research on the relationship between macroclimate and microclimates within the rainforest habitat of northern *Philoria* has begun (L. Bolitho, D. Newell, unpublished), and will provide important insights into the capacity for climate buffering to mitigate the impacts of climate change on these species.

Meta-analysis of post-fire amphibian studies

The various studies initiated on the impacts of the 2019/2020 fires on frogs across Victoria, New South Wales and Queensland provide a unique opportunity to fill a key knowledge gap: the response of Australian frogs to bushfire. It is strongly recommended that these data are compiled for the purposes of meta-analyses that seek to both estimate the impacts of the 2020/2021 fires on targeted species and provide a synthetic understanding of the response of Australian frogs to bushfire.

Data sets

Survey datasets, including counts of calling males, habitat assessments and fire severity assessments, were compiled for 103 sites across the range of *P. kundagungan, P. richmondensis* and *P. loveridgei*. These data have been combined with survey data from previous seasons as the foundation of a long-term monitoring program, and form the basis of Liam Bolitho's PhD thesis (in preparation). The data are held securely by Dr Ben Scheele at the Australian National University and Dr David Newell's group at Southern Cross University, in both cases on network drives that are password protected and regularly backed up. Location data have been made available to State agencies through which the surveys were licensed. These data include sensitive location information and have therefore not been made publicly available (such as through the Atlas of Living Australia). Data access may be requested through Dr David Newell (David.Newell@scu.edu.au). Survey data collected by QPWS staff have been entered into the state's wildlife information system WildNet.

Recommendations

Recommendations for management and future research on northern *Philoria* arising from this study are detailed below.

Management

1. Conservation status: This study supports the listing of *P. kundagungan* and *P. richmondensis* as Endangered under the Commonwealth EPBC Act 1999.
2. Protection from fire: Recovery of each of the northern *Philoria* species from the 2019/2020 fires is dependent on the maintenance of unburnt refuges and recovery of burnt habitat. Fire suppression is a high priority in the short-to-medium term, with careful consideration of fuel reduction burning to protect unburnt refuges and reduce the likelihood of further impacts on burnt habitat. Occupancy models developed during this study could be directly applied to these planning exercises.
3. Protection from ungulates and weeds: Effective feral pig control is urgently required in northern sections of Main Range National Park (Goomburra and Mistake Mountains) and the Burnett Creek catchment of Mount Barney National Park. Feral pigs are impacting unburnt refuge habitat for *Philoria* in these areas, both of which were significantly affected by the 2019/2020 fires. Similarly, protection of key refuge habitat from feral or domestic cattle is urgently required in the Main Range National Park in the Emu Creek area, and the Burnett Creek catchment in Mount Barney National Park. Weed control in burnt *Philoria* habitat should be pursued, particularly Lantana, which threatens rainforest regeneration, promotes the likelihood of Bell Miner Associated Die Back (in Eucalypt-dominated forests on rainforest margins) and exacerbates further fire risk.
4. Captive populations, head-starting and re-introduction: Establishing captive populations of northern *Philoria* is advisable as part of the suite of management initiatives needed to conserve these species. Captive husbandry protocols need to be established as a matter of urgency and could be guided by recent advances in the captive breeding program for *P. frosti* in Victoria. Establishing captive populations should be pursued immediately to provide adequate lead time to refine husbandry protocols and allow the widest possible array of genetic diversity to be included in the founding stock. Consideration should be given in the future to 'head-starting' and re-introduction to bolster declining populations or re-establish those that have gone locally extinct (where 'head-starting' entails collecting egg masses, raising tadpoles to metamorphosis and re-releasing these individuals). Re-introduction may be trialled in locations where natural recolonisation is considered unlikely, due to dispersal barriers or distance to remnant populations.
5. Ongoing monitoring, with expanded set of sites: This and previous studies have demonstrated protocols for monitoring northern *Philoria* that deliver robust estimates of population change. Conservation initiatives for these species require careful planning underpinned by up-to-date knowledge of occupancy and abundance. The core set of sites established through this and earlier studies should continue to be monitored, with sites added to increase the probability of detecting a significant change in occupancy between seasons. In combination with clear trigger points for specific conservation actions, ongoing monitoring following the protocols described here provides a foundation for recovery planning for northern *Philoria*.

Research

1. Spatial prioritisation analyses: The occupancy models developed here should be used to complete spatial prioritisation analyses that identify target habitat for post-fire management of northern *Philoria*, including fire suppression, control or exclusion of ungulates and weed control (as above). Potential to project changes in habitat suitability under climate change must also be pursued, with spatial prioritisation analyses in this context identifying likely refugia for longer-term targeting of conservation investment.
2. Complete detailed autecological studies: The ecology of each species of northern *Philoria* remains very poorly known. Detailed autecological studies are required on each species, with focus on compiling the demographic data needed to build population viability analyses for these frogs.
3. Clarify genetic subdivisions: The highly fragmented distribution of northern *Philoria* and restriction to high elevation rainforest suggests considerable genetic subdivision occurs in these taxa, with the potential for cryptic species in poorly surveyed areas. Comprehensive genetic sampling is recommended to resolve species boundaries and identify genetic subdivisions.
4. Clarify impacts of chytridiomycosis: The impacts of chytridiomycosis on northern *Philoria* is unknown, but could represent a key threat to these species, as it does for *P. frosti* in Victoria. Expanded field sampling to understand prevalence of infections should be pursued, with potential for laboratory studies on susceptibility to the pathogen.
5. Assess climate buffering: Projections of climate change impacts on northern *Philoria* rely crucially on understanding how changes in the macroclimate alter the microclimate experienced by the frogs. Research on this knowledge gaps has begun and will provide important insights into the capacity for climate buffering to mitigate the impacts of climate change on these species.
6. Meta-analysis of post-fire frog studies: The various studies initiated on the impacts of the 2019/2020 fires on frogs across eastern Australian provide a unique opportunity to fill a key knowledge gap: the response of Australian frogs to bushfire. It is strongly recommended that these data are compiled for the purposes of meta-analyses that seek to provide a synthetic understanding of the response of Australian frogs to bushfire.

Conclusion

This study sought to assess the impacts of the 2019/2020 wildfires on northern *Philoria* species – *P. kundagunga*n, *P. richmondensis* and western *P. loveridgei* – within the Gondwana Rainforests of Australia World Heritage Area. We estimate that 30% of suitable habitat was affected for *P. kundagungan*, 12% for *P. richmondensis* and 54% for western *P. loveridgei*. Surveys completed during the 2020/2021 breeding season provide evidence of some resilience to fire among these three species, with persistence and breeding choruses detected in burnt sites. However, impacts of the 2019/2020 fires and drought conditions before and after these fires were evident for the two species with adequate survey data, *P. kundagungan* and *P. richmondensis*:

1. Burnt sites were less likely to be occupied by *P. kundagungan* and supported fewer calling males. Similar effects were apparent for *P. richmondensis*, although restriction of fire to areas of low habitat suitability limited statistical power to detect these effects.
2. Stream saturation extent – a measure of local drought stress – was a key predictor of occupancy for both *P. kundagungan* and *P. richmondensis*.
3. Rates of occupancy and counts of calling males were lower post-fire for *P. kundagungan*, and significantly lower in 2019/2020 for *P. richmondensis* at the height of rainfall deficits across the range of this species. Encouragingly, significant winter and spring rainfall led to a recovery in occupancy and calling male abundance for *P. richmondensis* in 2020/2021.

This study provides evidence that increasing drought and fire frequency under climate change threatens northern *Philoria* and supports the listing of *P. kundagungan* and *P. richmondensis* as Endangered under the Commonwealth EPBC Act 1999. In the short term, fire suppression, control of feral pigs and cattle and weed control is needed to facilitate the recovery of these frogs from the 2019/2020 fires. Strategies are required to secure these species under a changing climate, with pursuit of captive populations a matter of urgency.

Acknowledgements

This study was funded through the Threatened Species Recovery Hub under the National Environmental Science Program of the Commonwealth Department of Agriculture, Water and Environment. Co-funding was provided by the 'Saving Our Species' program of the New South Wales Department of Planning, Industry and Environment and through Commonwealth bushfire recovery funds to the Queensland Department of Environment and Science. Assistance of the New South Wales and Queensland government agencies is gratefully acknowledged. In New South Wales, Mick Andren and Dave Hunter have provided vital input, with access to field sites and logistical support provided by Stephen King and Allan Goodwin at the Kyogle National Park office. In Queensland, Department of Science and Environment staff Andrew Meiklejohn, Sophia Levy and Diana Hughes are thanked for their contributions to fieldwork. Queensland Parks and Wildlife Service Rangers-in-Charge Stephen Finlayson and Justin O'Connell facilitated access to the Main Range National Park and Mount Barney National Park, respectively. Rachel Rouse, Matthijs Hollanders (both of Southern Cross University) and Desmond Bolitho are thanked for their contributions to fieldwork. Patrick Norman of the Griffith Climate Change Response Program at Griffith University provided vital support with sourcing and manipulating spatial layers for the analysis, with additional assistance with obtaining spatial layers provided by Rhonda Melzer and Michael Harte of the Queensland Department of Environment and Science. Finally, we acknowledge the important contribution of Rosalie Willacy to this study, whose Honours research on *P. richmondensis* established the core field sites for this species and delivered data on occupancy rates and abundance in the 2012/2013 season.

Ethics statement

Surveys in New South Wales were completed under scientific license SL102444 and Southern Cross University animal ethics permit 20/036. Surveys in Queensland were under authority from the Queensland Department of Agriculture and Fisheries Community Animal Ethics Committee, reference number SA 2019/08/700.

Philoria loveridgei, Mount Barney National Park. Image: Harry Hines

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Appendix

Sites at which surveys for northern *Philoria* were completed during the 2020/2021 season, showing examples of unburnt habitat, habitat burnt at different levels of severity and drought affected habitat.

1. Unburnt and no longer drought affected (Photo: Liam Bolitho)

2. Burnt at low-moderate severity and drought affected (Photo: Liam Bolitho).

3. Burnt at moderate severity and drought affected (Photo: H. Hines).

4. Burnt at moderate to high severity, no longer drought affected (Photo: L. Bolitho)

5. Burnt at high severity on left bank and drought affected (Photo: L. Bolitho).

6. Burnt at high severity and drought affected (Photo: L. Bolitho)

Further information:

http://www.nespthreatenedspecies.edu.au

This project is supported through funding from the
Australian Government's National Environmental Science Program.

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