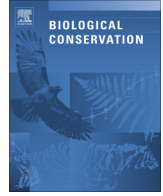




Contents lists available at ScienceDirect

## Biological Conservation

journal homepage: [www.elsevier.com/locate/biocon](http://www.elsevier.com/locate/biocon)

# Interactive effects of climate change and fire on metapopulation viability of a forest-dependent frog in south-eastern Australia



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## ARTICLE INFO

## Article history:

Received 14 November 2014

Received in revised form 20 May 2015

Accepted 29 May 2015

## Keywords:

Amphibian

Fire regime

Ecological niche model

Matrix population model

Niche-population model

Maxent

Population viability analysis

RAMAS

Habitat suitability model

## ABSTRACT

Climate change directly affects the suitability of habitats for species, but also indirectly alters natural disturbances such as fire, which can negatively impact species' persistence. Developing accurate predictions of climate change impacts requires estimates of the interactive effects of climate and disturbance regimes at both population and landscape scales. Here we couple a habitat suitability model with a population viability model to examine the interactive effects of climate change and altered fire regimes on a fire-responsive frog species across its geographic range in south-eastern Australia. By 2100, we predict expected minimum abundances (EMA) to decline by 66% (under GFDL-CM2 A1FI climate projections) or 87% (CSIRO Mk3.5 A1FI) in the absence of fire. Increased frequency of low-intensity fires reduced EMA by less than 5%, whereas increased frequency of high-intensity fires reduced EMA by up to 40% compared with the no-fire scenario. While shifts in fire regimes are predicted to impact metapopulation viability, these indirect effects of fire are far less severe than the direct impact of climate change on habitat suitability. Exploring the interactive impacts of climate change and altered disturbance regimes can help managers prioritize threats across space and time.

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## 1. Introduction

Climate change is one of the most significant challenges for the conservation of biodiversity in the 21st century (Heller and Zavaleta, 2009). Even small shifts in climate can directly affect survival (Chamaille-Jammes et al., 2006), reproductive output (Forchhammer et al., 1998; Crick and Sparks, 1999; Winkler et al., 2002), resource availability (Visser and Both, 2005) and habitat use (Telemeco et al., 2009). Climate change may also alter biotic interactions that affect the suitability of habitat for interacting species (Meynecke, 2004; Cahill et al., 2013; Moritz and Agudo, 2013). A relatively small number of native species are predicted to benefit by extending their distributions into previously unoccupied areas (Thuiller et al., 2005; Hamann and Wang, 2006; Van der Putten

et al., 2010). However, for most taxa, currently suitable habitat is predicted to decrease in spatial extent, and undergo considerable fragmentation (Meynecke, 2004; Thuiller et al., 2005; Penman et al., 2010). Under such circumstances, persistence will depend on an ability to maintain viable populations in remaining pockets of habitat, or an ability to colonise and persist in newly available habitat.

Climate change is also likely to impact species indirectly through altered disturbance regimes. The severity, frequency and seasonality of drought, flood and wildfire are all predicted to change under future climate scenarios (Palmer et al., 2008; Allen et al., 2010; Bradstock et al., 2012). Species are adapted to specific disturbance regimes, and the alteration of these regimes may affect habitat suitability or key population processes (Whelan, 1995; Keith, 1996; Wardell-Johnson, 2000). However, integrating shifts in disturbance regimes into projections of climate change impacts is difficult due to a poor understanding of the link between climate and most forms of disturbance (Bradstock et al., 2014).

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Predicted changes in fire regimes, for example, are extremely uncertain due to the interactive effects of the key drivers of fire in different landscapes. Most studies have predicted that wildfires will increase in frequency and intensity under climate change, but most of these predictions are based solely on predicted increases in the frequency and severity of fire weather (Bergeron and Flannigan, 1995; Fried et al., 2004; Hennessy et al., 2005; Pitman et al., 2007; Westerling and Bryant, 2008; Clarke et al., 2011). Annual area burnt is one spatial expression of fire regimes, and is a function of four switches – weather, biomass, fuel moisture, and ignitions (Archibald et al., 2009; Bradstock, 2010). Several opposing forces influence fire regimes under future climates. Firstly, severity of fire weather has been predicted to increase globally (Westerling et al., 2006; Matthews et al., 2012; Clarke et al., 2013), although average wind speed, a key predictor of fire behaviour, is predicted to decrease (McVicar et al., 2008). Secondly, leaf litter, a major component of biomass, is predicted to decrease in many forests and woodlands due to decreased moisture and increased temperature (Penman and York, 2010; Matthews et al., 2012; Thomas et al., 2014). Decreases in biomass may in turn result in a decrease in fire intensity and spread. However, these changes may be offset by increases in the concentration of carbon dioxide. Elevated carbon dioxide concentration increases the structural complexity of fuels by thickening woody vegetation and increasing foliage cover (Eamus and Palmer, 2007; Donohue et al., 2013), thereby increasing fuel loads. Thirdly, fuel moisture is likely to decrease with increasing temperature and seasonal reductions in rainfall (Matthews et al., 2012). Fourthly, ignitions are expected to increase under future climate scenarios (Price and Rind, 1994; Syphard et al., 2007; Penman et al., 2013; Romps et al., 2014), potentially increasing fire frequency and extent. However, the relative importance of each these variables, as well as their feedbacks and the influence of humans on ignition patterns and frequencies, are extremely uncertain, making definitive statements about future fire regimes difficult.

Accurate predictions of climate change impacts on species require estimates of the interactive effects of climate and disturbance regimes at population and landscape scales. Correlative habitat suitability models (HSMs) (Elith and Leathwick, 2009) are commonly used to infer the impact of climate change (Meynecke, 2004; Araújo and New, 2007). However, these models typically use coarse estimates of dispersal to estimate realised distributions (e.g., no dispersal vs. unlimited dispersal), and do not explicitly incorporate demographic parameters that influence extinction risk (Pearson and Dawson, 2003; Fordham et al., 2012). Recently, these limitations have been addressed by coupling HSMs with population viability analyses (PVAs) (Keith et al., 2008; Brook et al., 2009; Dullinger et al., 2012; Fordham et al., 2012). PVAs use key demographic variables to estimate a population's risk of decline, often simulating various alternative management scenarios. Coupling HSMs with PVAs allows for spatially explicit predictions of metapopulation extinction risk while accounting for dispersal between the temporally dynamic locations of suitable habitat patches (Keith et al., 2008; Conlisk et al., 2012; Fordham et al., 2012; Swab et al., 2012; Bonebrake et al., 2014). Few studies, however, have examined the interactive effects of climate change and disturbance regimes on metapopulation viability (but see Keith et al., 2008; Swab et al., 2012).

Frogs are considered extinction-prone globally (Houlahan et al., 2000; Stuart et al., 2004) and are likely to be extremely susceptible to future climatic changes (Pounds et al., 2006; Lemckert and Penman, 2012). Species that occur predominantly in fire-prone vegetation will be further impacted by climate-driven shifts in fire regimes. The impacts of different fire regimes on frog species are poorly understood (Penman et al., 2006b) but are likely to vary with fire intensity, species' life history traits, and the exposed life

stages (Bamford, 1992; Driscoll and Roberts, 1997; Pilliod et al., 2003; Woinarski et al., 2004; Penman et al., 2006b; Penman and Towerton, 2007).

Given the uncertainties about the impacts of climate change on fire regimes, it is important to examine the response of extinction-prone biota across the potential spectrum of fire frequency and intensity combinations that may occur in the future. Results of such studies will help identify fire regimes associated with high extinction risk and clarify whether these vary depending on future climate (Keith et al., 2008; Swab et al., 2012). Once high-risk fire regimes are identified, we may be able to develop management strategies that reduce the probability of such regimes arising under future climates. In this study, we couple a HSM and a PVA to examine the interactive effects of climate change and fire regime on a fire-responsive frog species. Many studies have examined shifts in predicted suitable habitat under climate change without explicitly considering dispersal or impacts of other threats such as fire. Here we overcome these limitations by explicitly considering dispersal and shifts in habitat suitability under changing climates and a range of fire regimes. This coupled modelling approach brings several advantages. First, it allows us to explicitly test effects of fire regimes on the species under changing climates, and the interaction between these threats on the viability of the species over a 100-year period. Second, this approach estimates the viability of the species directly by integrating demographic data, rather than inferring viability based on habitat suitability alone. As a result, we are able to estimate the relative threat to persistence due to fire and to climate change, which we would not have been able to do if we did not have a coupled model. Finally, the landscape-level impacts of fire on metapopulation viability would not be captured in either a standalone HSM approach or a static PVA. In undertaking this analysis, we specifically aimed to assess whether the fire regimes that are most detrimental to this species under current climates are also detrimental under future climates, and explore whether we can develop management strategies to mitigate the risk of extinction under such regimes.

## 2. Materials and methods

### 2.1. Study species

The study focused on the Giant Burrowing Frog, *Heleioporus australiacus*, as this species is restricted to fire-prone native vegetation throughout its range, primarily dry forest and heathland (Penman et al., 2004). *H. australiacus* is a large myobatrachid species found in native vegetation along the coast and adjacent ranges of south-eastern Australia (Fig. 1). Climate strongly influences the species' current distribution (Penman et al., 2005b, 2007b) and behaviour (Penman et al., 2006a). Longer-term climatic variation has also shaped the species' distributional extent. There are currently six species within the *Heleioporus* genus, with all other species occurring in south-western Australia (Cogger, 2000). Molecular evidence indicates that *H. australiacus* evolved from a west to east migration five million years ago and has subsequently been isolated due to changing climates that caused loss of forest cover in southern Australia (Maxson and Roberts, 1984; Roberts and Watson, 1993).

The species spends most of its time at depths of 1–30 cm below the soil surface (Lemckert and Brassil, 2003; Penman et al., 2008c). It is most active after rainfall, when it forages above ground (Penman et al., 2006a). After summer or autumn rains, the species moves to breeding sites such as hanging swamps, small creeks and wet heath (Littlejohn and Martin, 1967; Daly, 1996; Penman et al., 2006c). *H. australiacus* is currently listed as *vulnerable* on the IUCN Red List, and under Australian Commonwealth and State legislation.

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