Living on burrowed time: Aestivating fishes in south-western Australia face extinction due to climate change

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ABSTRACT

Climate change poses a severe global threat to freshwater ecosystems. Aestivating freshwater fauna (i.e., those that undergo periods of dominancy during dry periods) are likely to be highly susceptible to climate change due to their unique life-cycles, although empirical data that demonstrates this are lacking. This study utilised two aestivating fishes as model species, the salamanderfish, *Lepidogalaxias salamandroides*, and the black-stripe minnow, *Galaxiella nigrostriata*; both of which are endemic to Australia's Southwestern Province, an area undergoing a severe drying trend that global climate models project will continue. We found that the distributions of both species have undergone large reductions in extent of occurrence (EOO) and area of occupancy (AOO), coinciding with an extensive period of severe drying of the region. *L. salamandroides* was more likely to be found in deeper pools, while *G. nigrostriata* was found more often in pools with greater connectivity, and the probability of occurrence of both species was significantly associated with lower water temperature and pH, and higher oxidation–reduction potential. The majority of those factors are strongly influenced by the local climate, as stable isotope analyses found the hydrology of the pools in which they occurred was influenced by winter rainfall, either through direct input and/or expression from a superficial aquifer. This suggests that increasing water temperature and declining rainfall, features of the climatic shift in south-western Australia over the last 40 years, have been the major drivers of the observed declines in range. Given that climate models project both continued rainfall reductions within the species' distributional ranges and water table decreases by up to 4 m by 2030, further population losses are expected. Moreover, a review of the distribution of aestivating fish species throughout the world revealed that 75% are found in regions projected to dry due to climate change. This has considerable management implications, and active intervention, such as the protection of their existing environments and the creation of new suitable habitats, is required to prevent species declines. Fortunately, at least for the two aestivating species found in Western Australia, artificial ponds are utilised and therefore the construction of new waterbodies could help halt their decline, although further research is required to develop the optimal design for species persistence.

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1. Introduction

Freshwater environments comprise <1% of global water and these ecosystems are experiencing declines in biodiversity much greater than those seen in either terrestrial or marine ecosystems (Dudgeon et al., 2006). Several anthropogenic stressors have been identified as the causal factors of these declines, including water pollution, flow modification, habitat degradation or loss, damage to riparian vegetation, the introduction of invasive species and climate change (Morrongiello et al., 2011; Comte et al., 2013). Climate change is predicted to severely impact freshwater systems around the globe, through increased water temperature and altered flow regimes (Tisseuil et al., 2012). Changes in water availability are predicted to push numerous species of freshwater fishes towards extinction by 2070 (Kingsford, 2011), which will in turn impact on the functional diversity of freshwater ecosystems globally (Buisson et al., 2013).

Given their unusual life-cycles and habitat requirements, aestivating animals, those that undergo periods of dormancy through either summer or a dry period (Storey and Storey, 2012), are a group thought to be particularly susceptible to climate change (Saddlier et al., 2010). Aestivation occurs in a considerable range of animals, including the Australian land snail *Rhagada tescorum* (Withers et al., 1997), the green-striped burrowing frog *Cyclorana alboguttata* (Cramp et al., 2005), and the western-swamp tortoise *Pseudemydura umbrina* (Mitchell et al., 2012), however it is rare in fishes. To cope with challenges such as water retention and loss, and energy storage and utilisation, in these temporary environments, fishes that aestivate have unique adaptations, including metabolic suppression, unique morphology to aid in burrowing (e.g. elongated flexible bodies), and the secretion of mucous (Delaney et al., 1983; Allen and Berra, 1989; Gleeson et al., 1999; Koster, 2003; Chew et al., 2005).

Global climate circulation models project a mean global temperature rise of 1–7 °C by the end of the 21st century (IPCC, 2013). The resultant increase in water temperatures, decrease in dissolved oxygen, alterations to flow regimes, and changes in habitat quality and availability are likely to have a significant impact on freshwater fishes across the globe (Ficke et al., 2007; Morrongiello et al., 2011). A number of studies have considered the implications of rising water temperatures on the distribution of cold water fish species, such as salmonids, in northern hemisphere temperate systems (Meisner, 1990; Heino et al., 2009; Comte et al., 2013), while in Mediterranean, semi-arid and arid regions, hydrological shifts resulting from climate change are likely to have the major impact on fish communities (Jenkins et al., 2011; Morrongiello et al., 2011; Davis et al., 2013; Beatty et al., 2014). To date, however, there are no empirical data which might help predict how aestivating fishes will be affected.

This study considers the impact of climate change on two aestivating species, salamanderfish, *Lepidogalaxias salamandroides* Mees and black-stripe minnow, *Galaxiella nigrostriata* Shipway, from a drying region, Australia's Southwestern Province. Both *L. salamandroides* and *G. nigrostriata* are restricted to ephemeral, acidic wetlands within conservation reserves (i.e. National Parks, Nature Reserves, and State Forest) in the extreme south-western corner of the region (Berra and Allen, 1989a,b; Morgan et al., 1998). They are both small bodied, have relatively short life cycles and annually aestivate underground (Berra and Allen, 1989b; Pusey, 1989). *L. salamandroides* is unique in that it is the sole member of the Lepidogalaxiidae (Berra and Pusey, 1997), basally placed as the sister taxon of all Euteleosteomorpha (Li et al., 2010).

The Southwestern Province of Australia has experienced climatic drying since the 1970's, with the total rainfall declining by 10 to 50 mm per decade (~16% overall reduction) causing annual stream flows to decline by ~50% (Petrone et al., 2010). Mean annual air temperature has also increased between 0.10 and 0.30 °C per decade over that period, resulting in higher rates of evaporation from surface waters (Australian Bureau of Meteorology, 2014). Projections of climate change for the region indicate a further reduction (up to 20%) in mean annual rainfall and further warming (0.5 to 1.3 °C) by 2030 (Suppiah et al., 2007). Moreover, large reductions in fresh groundwater levels (>10 m in some areas) are also projected to occur (Barron et al., 2012). The severe decline in surface flows is negatively impacting potamodromous fishes in this region (Beatty et al., 2014); however, the potential impact of climate change on aestivating fishes is unknown. Although both *L. salamandroides* and *G. nigrostriata* are listed as near threatened under the IUCN Red List of Threatened Species (Wager, 1996a,b), we hypothesise that based on dire rainfall projections for the Southwestern Province, each of these species is likely to have suffered recent declines and would thus require urgent re-assessment.

This study aimed to investigate the resilience of the two aestivating fishes, *L. salamandroides* and *G. nigrostriata*, to climate change, by quantifying changes in their geographical range over the last few decades, identifying the factors that best explain their current distributions, and assessing the overall threats to the viability of remnant populations. We hypothesised that: (i) there has been a decline in the distribution of both species since previous surveys; and (ii) the environmental drivers of this distributional decline are directly linked to climate change. Finally, we review the distribution of aestivating fishes globally, to determine the degree to which their geographical ranges overlap with areas of projected drying due to climate change.

2. Materials and methods

2.1. Distribution

A historical (pre-2000) database of 39 site records for *L. salamandroides* and 28 (a subset of these 39) site records for *G. nigrostriata* was compiled from historical collections found in the Western Australian Museum (WAM), and studies conducted by Christensen (1982), Pusey and Edward (1990) and Morgan et al. (1998). A contemporary database was created of 53 sites examined in 2014 to 2015 for the presence or absence of these fishes (Fig. 1), which included a re-examination of all historical records. Additional sites were selected that had not previously been surveyed in order to identify new populations and increase the sample size for species distribution modelling. The sites were sampled with the use of seine nets, across two seasons, with all 53 sites being sampled in June and August (i.e. winter) and 34 re-sampled in either December or January (i.e. summer), just prior to the habitats drying. The area of the pool sampled varied from 12 to 300 m², with the aim of sampling a proportion (minimum of ~5%) of each habitat and minimising disturbance to the pool. Each species was recorded as present at a site if one or more fish were captured in either the winter or summer sample.

Current and historical presence data were mapped in ArcGIS Pro 1.0.2. Extent of occurrence (EOO) was determined by constructing minimum convex polygons around the perimeter of the sites as per IUCN (2014). Separate EOO polygons were constructed for historical sites and for current sites, with the difference in area between the two providing an estimate of the temporal change in EOO. Area of occupancy (AOO) was also determined as per IUCN (2014). A 1 km² grid was inserted as an overlay in ArcGIS with the number of squares overlaying sites summed and multiplied by the area of each square to determine the AOO. Again, the AOOs (one historical and one current) were determined for each species, with the difference between the two providing a measure of temporal change.

2.2. Environmental variables

The physical characteristics of all sites are available in the Supplementary Information (Table A1). A number of water quality and habitat variables were recorded at each site in winter and summer. Water quality variables commonly associated with freshwater fish distribution included water temperature, dissolved oxygen (ppm and percentage), conductivity ($\mu S \cdot cm^{-1}$), total dissolved solids (TDS), salinity (ppt), pH, and oxidation–reduction potential (ORP; mV) (Matthews, 1998). All variables were taken at a depth of approximately 30 cm using a water quality meter (YSI), with three replicates taken at various habitats within each site and a mean (± 1 S.E.) determined.

The habitat condition at each site was assessed by scoring for five variables: 'pool naturalness' (ranging from a score of 1 for no anthropogenic modifications to 5 for completely artificial); bank condition (ranging from a score of 1 for pristine to 5 for highly eroded); riparian vegetation naturalness (ranging from a score of 1 for all native to 5 for all introduced species); average bank angle; and proportion of shade cover (both based on visual observation). Other habitat variables measured included pool area (calculated from length and width dimensions), maximum pool depth (using a metre stick) and connectivity (ranging from a score of 1 for highly connected, such as a stream, to 5 for a completely isolated pool).

In 30 sites, selected at random across the sampling area, HOBO Pendant temperature/light data loggers were used to help identify date of drying. Each logger was installed at the deepest accessible point in winter. The loggers were programmed to record temperature (°C) and light (lum/ft²) at an interval of every two hours and were collected in the following autumn, with data being downloaded and temperature and light plotted over time for each site. Logged sites were deemed to have dried once a drastic diurnal fluctuation in light level was recorded. Drying date was recorded in days relative to the site which dried earliest in the year and varied from 0 (the earliest drying site) to 122 (the last site to dry).

2.3. Species distribution modelling

As environmental variables could not be measured at all sites in summer (due to drying of ponds), current occurrence data over the full range of 53 sites were related to environmental predictor variables