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Original Articles

Take the long way home: Minimal recovery in a K-selected freshwater crayfish impacted by significant population loss

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A B S T R A C T

Extreme disturbance can cause catastrophic mortality, population collapse and localised extinction in animal species. The ability of species to resist and recover from such disturbance is paramount to the persistent of populations thus regulating species distribution and diversity. The present study assessed the status of a slow-growing, long-lived and recreationally harvested freshwater crayfish, the Murray crayfish *Euastacus armatus*, which experienced significant population loss imposed by an extreme hypoxic blackwater disturbance in the Murray River in the southern Murray-Darling Basin, Australia. Specifically, before-after-control-impact monitoring, which accounted for imperfect and variable detection, was employed to assess indicators of recovery (occupancy, abundance, sex ratio and length structure) at affected and non-affected sites over 3–5 years following the hypoxic blackwater. A stochastic population model was further utilised as an indicator of longer term trajectories of recovery under a range of management scenarios. The indicators employed in the study suggested minimal recovery as there was no significant improvement in occupancy or abundance and length structures emphasising the continued underrepresentation of juveniles across the affected populations. Modelling simulations reinforce these findings with lengthy recovery trajectories (e.g. 50 years to reach pre-disturbance population sizes) forecast under natural recovery scenarios and any scenario involving harvest pressure predicted to delay this recovery timeframe. The findings emphasise the need to acknowledge realistic recovery timeframes for K-selected species impacted by extreme disturbance. It is now a critical time for concerted conservation and fisheries management to facilitate the recovery of the species across its range.

1. Introduction

Extreme disturbance events – natural or human-induced – can cause catastrophic mortality, population collapse and localised extinction of animal species (see Fey et al., 2015; Pickett and White, 2013). These extreme disturbances can be abrupt or gradual in nature, and have local and global implications, with the impacts often at disparity with the duration of the event (Smith, 2011). Prominent examples include the mortality of 250,000 seabirds following the *Exxon Valdez* oil spill in Prince William Sound (Piatt and Ford, 1996), widespread deaths of common seals *Phoca vitulina* around the coast of Europe in 1989 (Harwood and Hall, 1990), and even human populations (Kelly, 2005; Morgan et al., 2006). The ability of animal species to resist and recover from extreme disturbances is paramount to the persistence of populations thus regulating species distribution and diversity (Pickett and White, 2013; Reice et al., 1990). Understanding the impact of, and recovery from, disturbance is necessary as extreme climatic events and anthropogenic impacts are predicted to increase in the future (Bailey and Pol, 2016; Smith, 2011).

Animal populations naturally recover following extreme disturbance through reproduction of surviving individuals and/or recolonisation (Hughes, 2007; Parkyn and Smith, 2011). These recovery mechanisms are governed by the: (1) pre-disturbance population size and connectivity; (2) extent and severity of the disturbance; (3) return of suitable habitat and resources; and (4) life-history traits, phenotypic plasticity, genetic diversity and dispersal ability of the species (Beever et al., 2016; Parkyn and Smith, 2011). Species with short generation times, high fecundity and rapid growth (i.e. r-selected) are considered to have high recovery potential, whereas gradual recovery is anticipated for large, long-lived, late maturing and dispersal limited species (K-selected: Hutchings et al., 2012). Impaired recovery is likely if population declines are rapid and large (e.g. in excess of 50%) to the

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extent that Allee effects are expressed, or if threats are not abated and effective conservation is lacking (Hutchings, 2015; Hutchings et al., 2012). For fisheries species, the release from harvest pressure is considered a major driver of recovery (Hilborn et al., 2014; Lotze et al., 2011).

Whilst the study of recovery has a long history (see Duarte et al., 2015; Niemi et al., 1990), universal definition and assessment indicators remain elusive, and often vary amongst fisheries and conservation settings (Lotze et al., 2011; Westwood et al., 2014). Typically, recovery is defined as the temporal process in which populations return to pre-impact levels and the disturbance is no longer posing a threat (Parker and Wiens, 2005). Presently, Before-After-Control-Impact (BACI) designs are considered the most robust to assess environmental disturbance (Underwood, 1994; Verdonschot et al., 2013). It is important for these designs to account for the spatial extent of the disturbance and be of sufficient temporal scale to adequately reflect prevailing dynamics (i.e. acknowledge shifting baselines) and track indicators over time (i.e. full recovery: Verdonschot et al., 2013). It is also acknowledged that predetermined endpoints are necessary as are multiple indicators to robustly assess recovery (Keeley et al., 2014; Verdonschot et al., 2013). As abundance is most commonly used as the indicator of recovery (Lotze et al., 2011), it is imperative to account for imperfect detection as not to make incorrect conclusions (cf. Gwinn et al., 2016; Kellner and Swihart, 2014).

Approximately one-third of freshwater crayfish across the world are at risk of extinction (Richman et al., 2015); with members of the Australian *Euastacus* genus considered among the most threatened (Furse and Coughran, 2011b; Furse et al., 2013). *Euastacus* species appear particularly vulnerable to environmental disturbance and anthropogenic change (Furse and Coughran, 2011c), with flash flooding (Furse et al., 2012), severe blackwater (McCarthy et al., 2014), wildfire (Johnston et al., 2014) and habitat degradation (Noble and Fulton, 2017) shown to contribute to population declines. The capacity of *Euastacus* species to recover through population growth or recolonisation is expected to be constrained (Furse and Coughran, 2011a) by life history traits (i.e. slow-growth, late maturity and low fecundity); restricted movement and dispersal ability; and limited gene flow and low levels of genetic diversity (Honan and Mitchell, 1995; Miller et al., 2014; Ryan et al., 2008). Additionally, many *Euastacus* species have already experienced declines in distribution and abundance (Furse and Coughran, 2011a), increasing the likelihood of smaller population sizes and population fragmentation, which will further act to limit capacity to resist and recover from disturbance (Allendorf et al., 2013; Frankham et al., 2010).

The Murray crayfish *Euastacus armatus* (von Martens, 1866) is a recreationally harvested freshwater crayfish occurring across the southern Murray-Darling Basin (MDB), Australia (Morgan, 1997). This long-lived (~28 years), slow-growing (K = 0.0933), late-maturing (~8–9 years) and low fecundity (up to 2000 eggs per mature female) has experienced substantial decline in distribution and abundance over the past 50 years attributed to river regulation, pesticides and pollutants, habitat degradation and harvest pressure and blackwater events (Furse and Coughran, 2011a,b; Horwitz, 1995; Walker and Thoms, 1993). Most recently over 2010–11, *E. armatus* populations, along with other freshwater crayfish (common yabby *Cherax* spp., freshwater shrimp *Paratya* spp. and freshwater prawns *Macrobrachium* spp) and freshwater fish, were impacted by an extreme hypoxic blackwater disturbance (King et al., 2012; Leigh and Zampatti, 2013; McCarthy et al., 2014); antecedent drought conditions and unseasonal inundation led to the accumulation and breakdown of large quantities of organic matter, which resulted in hypoxia, persisting for nearly six months, across much of the present range of the species (Whitworth et al., 2012).

During this extreme disturbance, *E. armatus* were observed emerging from the water in the affected areas, a behavioural adaptation to resist short-term periods of adverse water quality (King et al., 2012), but indicators (at the scale of 1–2 years) revealed a significant (81%) population loss across affected areas, with all size classes and both sexes equally impacted (McCarthy et al., 2014). The impacts of the 2010–11 blackwater disturbance, and other recent research (Zukowski et al., 2011, 2012), led to amendments of the recreational fishery regulations, including closure of affected areas; shift from a minimum length limit (MLL = 90-mm) to a harvestable slot length limit (HSLL = 100–120-mm); the reduction of bag (five to two crayfish) and possession (10 to four crayfish) limits; and the contraction of the open fishing season (four to three months: NSW DPI, 2014). The legacy of this extreme disturbance could to be profound, with restricted movement (Ryan, 2005) and gene flow suggesting natural recovery of affected populations could occur across a decadal timescale (Whiterod et al., 2017).

This study aimed to assess the present status of *E. armatus* populations affected by the 2010–11 blackwater event as well as forecasting long-term indicators of recovery. Specifically, using a BACI design we assess key indicators, including abundance (adjusted for detection probability: Gwinn et al., 2016), sex ratio and length structure at hypoxic blackwater affected sites and non-affected sites before (2010: Zukowski, 2012) and shortly after (2012: McCarthy et al., 2014) with now three to five years (2014, 2015 and 2016). To achieve broader insight, we forecast long-term recovery indicators under potential management scenarios using a recently developed population model (Todd et al., in press). It was hypothesised that the abundance of the species will have increased little and affected populations will remain patchy and modelled simulations will reveal slow population growth trajectories indicating that recovery will be a gradual process.

2. Methods

2.1. Sampling region and protocol

The Murray River flows 2530-km from the south-eastern highlands of Australia, through the southern MDB, to the sea at Goolwa (Eastburn, 1990). The Murray River is highly-regulated by upland impoundments, low-level weirs and irrigation diversions along much of its length (Walker, 2006). The present study focused on a 1100-km section (i.e. 1094–2194 river-km upstream of the Murray mouth) of the Murray River previously sampled in 2010 (Zukowski, 2012) and 2012 (McCarthy et al., 2014) (Fig. 1). This section encompassed the lower sections of the headwater tract of the river along with the low-gradient and meandering river channel of the gently undulating riverine plains and Mallee trench tracts (Eastburn, 1990). Consistent with 2010 and 2012, a total of 16 sites were sampled across this section, which conformed to the Before-After-Control-Impact (BACI) design: six control sites upstream of the large and significant river red gum floodplain wetland system, Barmah-Millewa Forest that were "non-affected" by hypoxic blackwater in 2010–11 whereas 10 treatments sites within and downstream of Barmah-Millewa Forest were affected.

At all sites, twenty standard hoop nets (single 800-mm steel hoop diameter, 13-mm stretch mesh size, 0.3-m drop baited with ox liver) were first deployed by boat during daylight hours (0800–1700) and retrieved hourly over two deployments (maximum of 40 net lifts). Nets were typically set 3–10-m from the river bank at least 40-m apart over a 2-km reach and re-deployed approximately 10-m from its initial set position after the first lift. Nets where bait had been lost or were snagged upon lifting were excluded from the abundance analysis. As with 2010 and 2012, the 16 sites were sampled in random order during the austral winter (1–11 July 2014; 15–26 June 2015; 18–28 July 2016) when *E. armatus* catches are highest (Zukowski et al., 2012).

Sampled *E. armatus* were sexed and occipital carapace length (OCL, in mm: from eye-socket to rear of carapace) measured with vernier calipers (Kinchrome). Additionally, sampled individuals were marked (see Ramalho et al., 2010) to identify potential recaptures (none were obtained: *unpublished data*) before being returned to the water. Water temperature (°C) was measured during sampling at each site using a multiprobe system (556 MPS, Yellow Springs Institute, YSI) and mean