Using population models to estimate the expected benefit of management actions

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Acknowledgment

We acknowledge and respect Victorian Traditional Owners as the original custodians of Victoria's land and waters, and their unique ability to care for Country and deep spiritual connection to it. We honour Elders past and present, whose knowledge and wisdom has ensured the continuation of culture and traditional practices.

We are committed to genuinely partnering, and meaningfully engaging, with Victoria's Traditional Owners and Aboriginal communities to support the protection of Country, the maintenance of spiritual and cultural practices, and their broader aspirations in the 21st century and beyond.

Arthur Rylah Institute for Environmental Research
Department of Environment, Land, Water and Planning
PO Box 137
Heidelberg, Victoria 3084
Phone (03) 9450 8600
Website: www.ari.vic.gov.au

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Case studies of six aquatic species

Jian D.L. Yen[1], Zeb Tonkin[1], Charles Todd[1], Daniel Stoessel[1], Tarmo A. Raadik[1] and Jarod Lyon[1]

[1]Applied Aquatic Ecology, Arthur Rylah Institute for Environmental Research
123 Brown Street, Heidelberg, Victoria 3084

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Arthur Rylah Institute for Environmental Research
Department of Environment, Land, Water and Planning
Heidelberg, Victoria

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Summary

***Context and aims*:**

Given the widespread threats to many species and limited conservation funding, biodiversity managers often need to prioritise management actions based on their likely effectiveness, given current and future threats. However, estimating the effectiveness or expected benefit of management actions is challenging, especially when knowledge of biodiversity responses to threats and actions is limited. Expert elicitation is commonly undertaken when collating the current knowledge for many species and locations in the absence of detailed data. When considering fewer species for which more data exist, population models are a complementary approach that can be used to predict how populations and species might respond to proposed management actions.

This report presents a population-modelling framework for estimating the expected benefit of management actions over a 50-year period under five scenarios of climate change. The framework was applied for six aquatic species in Victorian waterways: Macquarie perch (*Macquaria australasica*), Murray cod (*Maccullochella peelii*), platypus (*Ornithorhynchus anatinus*), estuary perch (*Percalates colonorum*), Yarra pygmy perch (*Nannoperca obscura*) and barred galaxias (*Galaxias fuscus*). The six species range from the narrowly distributed barred galaxias to the widespread Murray cod and platypus, and provide illustrative case studies with diverse threats and management actions.

***Methods*:**

We developed models of population dynamics that incorporate the effects of species-specific management actions (e.g. genetic mixing, stocking, fishing regulations, instream and riparian habitat rehabilitation, protection of natural flows, and environmental water delivery). We used these models to simulate population trajectories under all relevant combinations of management actions, in multiple locations representing the broad habitat types occupied by self-sustaining populations of each species. We calculated the expected benefit (defined as the probability of population persistence over 50 years relative to a do-nothing management scenario) of each management action in isolation and in combination with other actions from simulated population trajectories. Calculations of expected benefit were undertaken at the level of individual populations and across multiple populations of each species.

***Results*:**

Ranking management scenarios by expected benefit highlighted broad differences among species, and three distinct patterns emerged across many management scenarios (gradual declines in benefit, rapid declines in benefit, or constant benefit). These patterns have consequences for management decisions. The first pattern is associated with incremental effects resulting from managers having access to many actions and substantial flexibility in management decisions (e.g. based on cost), the second pattern is associated with managers being limited to aa highly constrained set of beneficial decisions focused on a few essential actions, and the third pattern is associated with managers having access to many highly effective combinations of actions and consequent flexibility in decisions. Differences among climate change scenarios did not substantially alter the rank order of management scenarios or the most-beneficial management actions. The exceptions to this pattern were most apparent in the estimates of cost-effectiveness, with smaller (i.e. cheaper) sets of actions sometimes being more cost-effective under less-extreme climates. Despite similarities in the rank order of the management scenarios across climate change scenarios, the absolute value of the expected benefit differed among climate change scenarios, suggesting that the degree of need for management intervention may depend on climate.

***Conclusions and implications*:**

Estimating the expected benefit of management interventions requires insight into species biology, underlying threats, and the feasibility and effectiveness of management actions. Expert elicitation is an efficient and useful method of expanding the current ecological knowledge of species in the absence of published data, but often results in implicit or unspecified mechanisms and assumptions. Population models are a complementary approach that captures the links between threats, actions, and species' responses in explicit, quantitative statements based on measurable ecological processes. A focus on processes has benefits and limitations—improving the interpretability and testability of expected benefit estimates, but simultaneously increasing the complexity of the estimation method, thereby reducing its generality and scalability.

The case studies presented here demonstrate that population models can be developed from different levels of existing knowledge, with model outputs likely to be most informative when based on detailed, species-specific knowledge. A certain amount of detailed knowledge exists for some of the species of our case

studies, due to substantial coinvestment in (i) the development of population models and (ii) targeted research into population processes and responses to management interventions. The case studies developed primarily within this project illustrated that model outputs were informative even when the existing knowledge of population processes was limited; however, the less-developed models will benefit from expert assessments of the assumptions about species biology and of the feasibility and effectiveness of management interventions. A critical next step will be testing and comparing management actions in field settings to identify gaps in the basic knowledge of species biology, constraints in the implementation of management actions, and interactions between threats and management actions not captured in existing population models. Iteratively testing and updating population models in this way will determine where and when model predictions are reliable and reduce the risk of adverse outcomes from management interventions.

1 Introduction

1.1 Background

Decisions around biodiversity management often depend on the expected benefits of proposed management actions (Walsh et al. 2012). However, expected benefits can be challenging to estimate, especially when data on biodiversity responses to management actions are limited in extent or detail. For many species and actions, expert elicitation is an effective method for generating quantitative estimates of benefit in the absence of detailed data, especially when coupled with systematic conservation planning methods (e.g. McIntosh et al. 2017; Thomson et al. 2020). When considering a smaller set of species for which some data exist, a complementary approach is to estimate expected benefits with quantitative models based on the underlying ecological processes (Briscoe et al. 2019). Models based on ecological processes ('process-explicit models') provide a structured, quantitative method for using existing knowledge and data to predict biodiversity responses to proposed management actions. These include matrix population models (Caswell 2001; Yen et al. 2013; Todd and Lintermans 2015), which are the focus of this report, but also other process-explicit models, such as individual-based models or biophysical models (Briscoe et al. 2019; Hradsky et al. 2019).

A focus on ecological processes has multiple benefits. First, estimating the effects of management actions requires clear statements about how and why a particular action affects a population or species. This need for clarity connects modellers with relevant experts and stakeholders through a common currency (ecological processes). Second, process-explicit model structures can be transferred to new situations whenever the underlying processes are assumed to be the same. Examples include the transfer of flow-ecology knowledge to previously unstudied rivers, and known biodiversity responses to modelled future climates. Notwithstanding the risks associated with extrapolation, this transferability can support extensive comparisons of counterfactual scenarios (or 'causal modelling', Hernán et al. 2019), which is particularly useful when estimating the effects of threats or actions in complex systems where replicated experiments are infeasible. Third, process-explicit models generate predictions at multiple levels and time scales, which allows testing of model assumptions or predictions over short time scales (e.g. one or a few years), even when the primary model output is defined over long time scales (e.g. decades). Last, process-explicit models can be readily updated when circumstances change. Such updates might be required when new data become available, when models are found to perform poorly, or when models are applied to a new decision context.

1.2 Aims

This report presents a population-modelling framework for estimating the expected benefits of a suite of management actions over a 50-year period under multiple scenarios of climate change (Figure 2). This framework uses existing data and research outputs, where available, together with outputs from expert elicitation, to identify threats to and relevant management actions for individual species. The estimates of expected benefit were derived from a matrix population model connecting population vital rates (typically, survival and reproduction) with changes in population abundances through time. Management actions exert their expected benefits through their effects on vital rates, so the primary modelling assumption (that a particular action will result in an expected benefit) can be captured in a measurable and testable process. Expected benefit is defined as the probability of population persistence over 50 years relative to a do-nothing management scenario, which is consistent with Biodiversity Division's decision-support tools Strategic Management Prospects and Specific Needs.

The population-modelling framework was applied to case studies of six aquatic species in Victorian waterways: Macquarie perch (*Macquaria australasica*), Murray cod (*Maccullochella peelii*), platypus (*Ornithorhynchus anatinus*), estuary perch (*Percalates colonorum*), Yarra pygmy perch (*Nannoperca obscura*) and barred galaxias (*Galaxias fuscus*). These species differ in their life history, distribution, conservation status, and threatening processes, providing illustrative case studies that span a range of ecological, management and social contexts.

This report addresses the following aims:

1. to develop and demonstrate the application of new, open-source software for simulating population dynamics

2. for each species of the study, to refine existing population models or develop new population models to estimate the effects of current and future threats (e.g. climate change) and of implemented or proposed management actions
3. to simulate the population trajectories at locations representing the broad habitat types occupied by self-sustaining populations of each species, under multiple climate change scenarios and management actions
4. to present estimates of the expected benefit at the level of individual populations and across multiple populations.

This report is the final output for the project *Using population models to estimate the expected benefit of management actions*. Details of each case study are provided in Appendices A–F. The Macquarie perch case study is in preparation for submission to the peer-reviewed journal *Biological Conservation*.

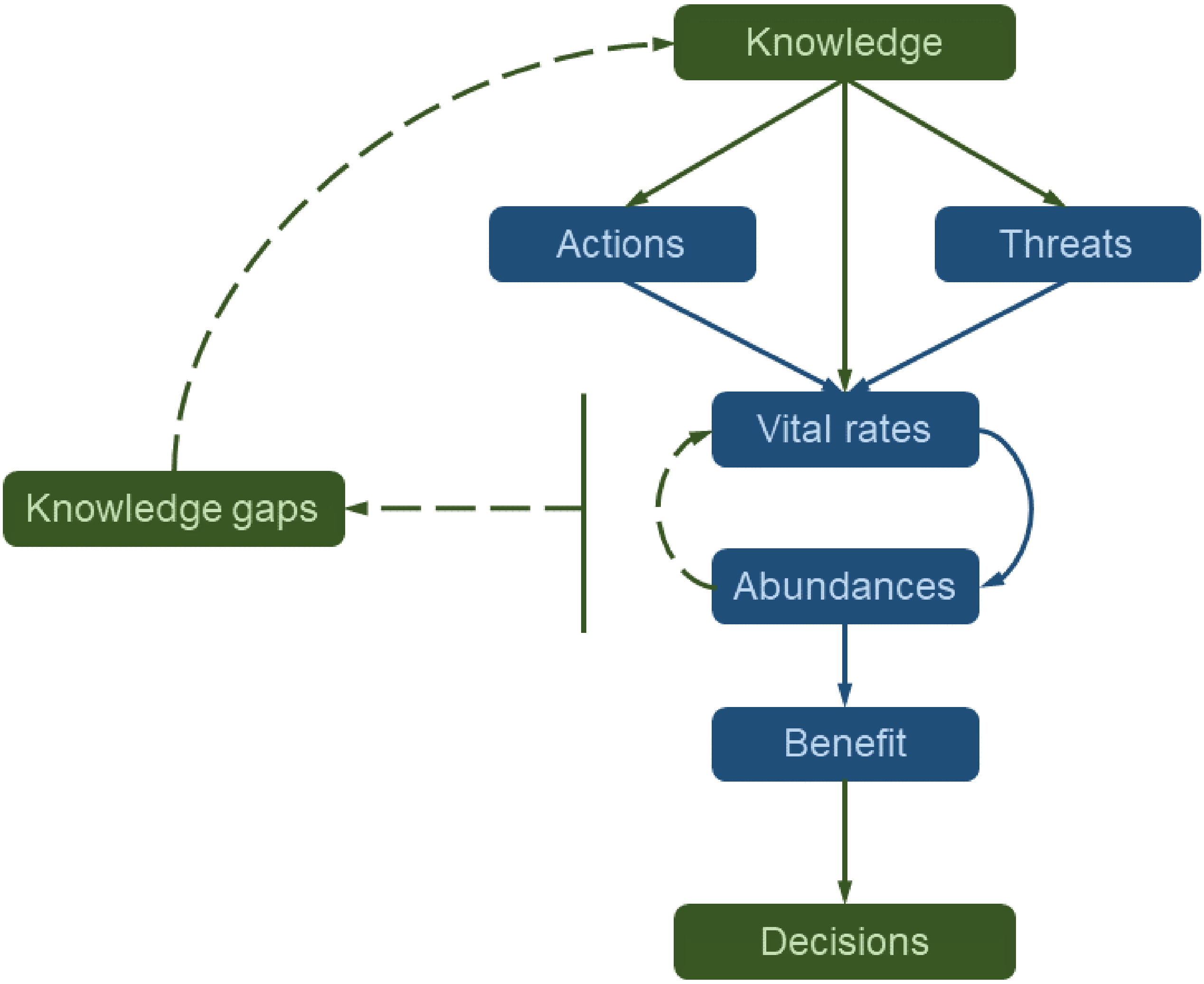

Figure 1. Overview of the population-modelling framework. Current knowledge, including existing data and expert knowledge, informs population vital rates, threats, and relevant management actions. Vital rates can be used to project population abundances resulting from particular threats and actions, and thus support the estimates of expected benefit under various management scenarios. Expected benefit can be defined in many ways, depending on the management context, but is defined here as change in the probability of population persistence over 50 years relative to a do-nothing management scenario. The blue boxes and arrows highlight key components of the population-modelling framework, with current knowledge and decisions (green boxes and arrows) being determined independently of the population models (e.g. through a structured decision-making framework). Dashed lines represent iterative model validation and updating, which can identify and address critical gaps in knowledge. Iterative model updates are mentioned but not described in detail in this report.

2 Methods

2.1 Study species and systems

Case studies are presented for river populations of six species, ranging from widespread species (Murray cod, platypus and estuary perch) to those with narrow or fragmented distributions (barred galaxias, Macquarie perch and Yarra pygmy perch) (Table 1). All species are dependent on sufficient river flows in order to complete their life cycle, and are susceptible to a range of threatening processes, including climate change, habitat loss, overexploitation, and competition with or predation from introduced fish species. Importantly, all six study species are the focus of specific management actions designed to mitigate key threats. Key actions include management of riverine flows (particularly environmental water delivery), fisheries regulations, instream and riparian habitat rehabilitation, genetic mixing, and active stocking or translocations. All six species have high conservation significance, with four species listed under the Victorian *Flora and Fauna Guarantee Act 1988* and the remaining two species subject to multiple conservation management interventions.

2.1.1 Macquarie perch

Macquarie perch is a freshwater fish species that was historically abundant in south-eastern Australia (Koehn et al. 2020). Macquarie perch supported an important recreational fishery, particularly in Victoria, until the 1980s, but the species has since undergone a dramatic decline in range and abundance. Extant Macquarie perch populations face a range of threats, such as habitat loss, overexploitation, water extraction, and loss of genetic diversity in the small, isolated populations (Koehn et al. 2020). Management actions attempting to reverse the decline of Macquarie perch have included recreational fishing regulations (Hunt et al. 2011), instream and riparian habitat repair (Saddlier et al. 2002; Sharley et al. 2019), the delivery of environmental water (Tonkin et al. 2014, 2017), and water harvesting regulations (HARC 2020) (Table 1). Recent management interventions have included the reintroduction of Macquarie perch into areas where it has become locally extinct and the reintroduction of genetic variation into depauperate populations (Pavlova et al. 2017; Lutz et al. 2021) (Table 1). The Macquarie perch case study is based on a highly developed population model (Todd and Lintermans 2015) and studies of Macquarie perch population processes and responses to management interventions (Table 1) supported by substantial coinvestment from management and research agencies [the Arthur Rylah Institute for Environmental Research (ARI), the Victorian Fisheries Authority (VFA) and the Goulburn Broken Catchment Management Authority (GBCMA)].

Three extant Macquarie perch populations were selected for this study (Table 1). The locations of these populations were: Seven Creeks, an unregulated creek in central Victoria; the Yarra River, a regulated river that flows through greater Melbourne; and Lake Dartmouth, a constructed water storage on the Mitta Mitta River in north-eastern Victoria.

2.1.2 Murray cod

Murray cod is one of the largest freshwater fish species in the world and is widely distributed across south-eastern Australia (Koehn et al. 2020). Murray cod are economically and culturally important and are highly valued by recreational anglers. Although Murray cod remain widespread, the species has experienced consistent declines in abundance over recent decades (Koehn et al. 2020). The key threats to Murray cod populations are water extraction, altered flow seasonality, overexploitation, and loss of instream woody habitat (Koehn et al. 2020). Actions to mitigate threats to Murray cod populations include the delivery of environmental water, restrictions on water releases during summer, rehabilitation of instream woody habitat, stocking of young-of-year fish into rivers, and regulations on recreational fishing (Koehn and Harrington 2006; Stuart et al. 2019; Tonkin et al. 2020, 2021) (Table 1). The Murray cod case study builds on a highly developed population model (Todd et al. 2004, 2005; Sherman et al. 2007) and reflects two decades of research into Murray cod population dynamics and targeted management of the species (Table 1).

Three extant Murray cod populations were selected for this study (Table 1). The locations of these were: the lower Goulburn River, a regulated river in central Victoria; the lower Campaspe River, a regulated river in central Victoria; and the heavily regulated Murray River between Yarrawonga and Barmah.

2.1.3 Platypus

The platypus is a freshwater monotreme widespread across eastern and south-eastern Australia. Platypus populations have declined in abundance over recent decades, with these declines predicted to continue unless current threats to the species are addressed (Bino et al. 2019, 2020). Threats include loss of riparian and instream habitat, low-flow conditions due to water extraction or drought conditions, and physical barriers to dispersal (Bino et al. 2020). These threats are exacerbated by climate change, in particular, the extended drought conditions and more frequent and severe bushfires observed in south-eastern Australia in recent decades (Klamt et al. 2011; Bino et al. 2020). Several actions have been proposed to mitigate the threats to platypus populations, including instream and riparian habitat rehabilitation, the delivery of environmental water, the provision of drought refuges or relief (e.g. targeted watering), and the removal of barriers to dispersal (Table 1). The platypus population model was based on a previously developed metapopulation model and associated research (Bino et al. 2015, 2020) and has not been refined for or tested in the rivers studied here.

Four extant platypus populations were selected for this study (Table 1). The locations of these were: the upper Campaspe River, a regulated river in central Victoria; the Mackenzie River, a small river in the Wimmera catchment in north-western Victoria; the Ovens River, a largely unregulated river in north-east Victoria; and the Little Yarra River, an unregulated tributary of the Yarra River east of Melbourne.

2.1.4 Estuary perch

Estuary perch is a catadromous fish species that inhabits estuaries across south-eastern Australia. Estuary perch are long-lived, top-order predators and are highly valued by recreational anglers. Estuary perch remain relatively widespread, but shifts in age distributions towards older fish in southern latitudes suggest that these populations are at risk of collapse due to repeated recruitment failure (Stoessel et al. 2018, 2020). Threats to estuary perch include water extraction, reduced inflows, and historical losses of instream woody habitat (Stoessel et al. 2020). Although the species is at risk of overexploitation, recreational angling is not a major threat to the Snowy River population discussed here. Relevant management actions for estuary perch include stocking of young-of-year fish into rivers and the delivery of environmental water to reintroduce spawning cues and enhance survival of early life stages (Table 1). Rehabilitation of instream habitat was included as a potential management action but is not currently under consideration in the study location (Table 1). The estuary perch case study is based on an existing population model (Stoessel et al. 2018, 2020) and is supported by coinvestment from several management and research agencies [ARI and the East Gippsland Catchment Management Authority (EGCMA)] (Table 1).

The Snowy River estuary perch population was selected for this study (Table 1). The Snowy River is a large, heavily regulated river in south-east Victoria that has experienced large reductions in discharge following the construction of the Snowy Mountains Hydroelectric Scheme from 1949 to 1974.

2.1.5 Yarra pygmy perch

Yarra pygmy perch is a freshwater fish species endemic to south-eastern Australia. Although widely distributed across Victoria and South Australia, Yarra pygmy perch populations are highly fragmented, with many populations declining in abundance in recent decades (Saddlier and Hammer 2010). At least three known populations have not been recorded since they were first discovered in the 1970s and 1980s (Saddlier and Hammer 2010). Yarra pygmy perch populations face many threats, including loss of critical wetland habitat and connectivity of this habitat to river channels, loss of persistent water during drought, predation by introduced species, and loss of genetic diversity in the small, isolated populations (Saddlier and Hammer 2010). Several actions have been proposed to address threats to Yarra pygmy perch populations, including rehabilitation or maintenance of critical habitat, provision of drought refuges, detection and removal of introduced fish species, establishment of new populations, and reintroduction of genetic variation into depauperate populations (Table 1). The Yarra pygmy perch case study is based on an existing population model for the related southern pygmy perch (Todd et al. 2017) and is supported by coinvestment from several management and research agencies (ARI, Murray Local Land Services) (Table 1).

Two extant Yarra pygmy perch populations were selected for this study (Table 1). The locations of these were: Deep Creek, a major tributary of the Maribyrnong River in central Victoria; and Armstrong Creek, a small tributary of the Barwon River near Geelong, Victoria.

2.1.6 Barred galaxias

The barred galaxias is a freshwater fish species endemic to a small upland area on the northern side of the Great Dividing Range in Victoria. Barred galaxias are present in 10 river systems, with 11 known population centres likely resulting from the fragmentation of a previously larger population (Raadik et al. 2010; Raadik 2015). Extant populations are isolated from one another, with most populations being small and restricted to short (1–2 km) headwater sections of upland streams (Raadik et al. 2010). The primary threat to barred galaxias populations is predation by and competition with introduced fish species, particularly rainbow trout and brown trout (Raadik et al. 2010). Secondary threats are the loss of persistent water during drought, direct and indirect effects of bushfires on water temperature and water quality, and loss of genetic diversity in the small, isolated populations (Raadik et al. 2010). Actions implemented to address these threats include the detection and removal of introduced fish species, establishment of new populations, *ex situ* care of individuals during or following extreme events (e.g. drought, bushfire), and the reintroduction of genetic variation into extant populations (Table 1). The barred galaxias population model was developed from the life history characteristics of the species (and congeners) and has not been empirically validated (Table 1).

Four barred galaxias populations were selected for this study (Table 1). The locations of these were: the Rubicon River, a small river and tributary with headwaters on the slopes of Lake Mountain; the Howqua River, a small river with headwaters on the slopes of Mt Stirling; the Delatite River, a small river that rises north of Mt Buller; and Perkins Creek, a small tributary of the Goulburn River with headwaters near Woods Point. The relevant sites in the Rubicon River, Howqua Rivers and Delatite River are fed by snowmelt, so receive more reliable inflows than the lower-elevation Perkins Creek site.

Table 1. List of study species and systems

This table summarises the information on the study species and populations, relevant threats and actions, and the level of population model development. Actions are assumed to be relevant to all populations unless a subset of populations is listed in the Actions column. Population model development ranges from ‘preliminary’ (in which model structure and parameters have been based on life history characteristics reported in the literature) to ‘extensive’ (in which model structure and parameters have been based on existing research programs into population processes and their consequences for management interventions). For the latter, the population models presented here reflect substantial coinvestment from management agencies [e.g. the Department of Environment, Lands, Water and Planning (DELWP), VFA, and Catchment Management Authorities (CMAs)]. Relevant published works are listed in the right-hand column.

Species	Populations	Threats	Actions	Population model development
Macquarie perch (*Macquaria australasica*)	Seven Creeks Yarra River Lake Dartmouth (Mitta Mitta River)	Water extraction Habitat loss Overexploitation Loss of genetic diversity	Environmental water (Yarra River) Flow protection (Seven Creeks) Instream habitat rehabilitation (Seven Creeks) Stocking (Lake Dartmouth and Seven Creeks) Fishing regulations (Lake Dartmouth and Yarra River) Genetic mixing	Extensive (Tonkin et al. 2014, 2017, 2019; Todd and Lintermans 2015; Pavlova et al. 2017; Sharley et al. 2019; Lutz et al. 2021)
Murray cod (*Maccullochella peelii*)	Goulburn River Campaspe River Murray River	Low winter and spring flows Elevated summer flows Habitat loss Overexploitation	Environmental water Instream habitat rehabilitation (Goulburn River only) Stocking (Goulburn and Campaspe rivers) Fishing regulations	Extensive (Todd et al. 2004, 2005; Sherman et al. 2007; Lyon et al. 2019; Tonkin et al. 2021)
Platypus (*Ornithorhynchus anatinus*)	Upper Campaspe River Mackenzie River Ovens River Little Yarra River	Cease-to-flow events Habitat loss Barriers to dispersal Loss of genetic diversity	Environmental water Permanent drought refuges (Mackenzie and Upper Campaspe rivers) Temporary drought relief Riparian revegetation Instream habitat rehabilitation	Preliminary (Bino et al. 2015, 2020)

			Barrier removal Genetic mixing (Mackenzie River) Translocations (Mackenzie River)	
Estuary perch (*Percalates colonorum*)	Snowy River	Water extraction Loss of spawning cues Habitat loss	Environmental water Instream habitat rehabilitation Stocking	Moderate (Stoessel et al. 2018, 2020)
Yarra pygmy perch (*Nannoperca obscura*)	Deep Creek Armstrong Creek	Loss of wetland habitat or connectivity Loss of persistent water Predation by introduced fish species Loss of genetic diversity	Wetland habitat rehabilitation Drought refuges Introduced fish species detection and removal Genetic mixing Population establishment	Moderate (Todd et al. 2017)
Barred galaxias (*Galaxias fuscus*)	Rubicon River Howqua River Delatite River Perkins Creek	Loss of persistent water Predation by introduced fish species Bushfires Loss of genetic diversity	Introduced fish species detection and removal Genetic mixing *Ex situ* care Population establishment	Preliminary (Raadik et al. 2010)

2.2 Population-modelling framework

The population-modelling framework links threats and management actions to the expected benefit with matrix population models (Figure 2). Matrix population models use information on population vital rates, typically survival and reproduction, to project population abundances through time (Caswell 2001). Relating vital rates to threats and actions enables projections of population abundances over long time scales (e.g. 50 years) and under various threat or management scenarios. These projected abundances allow calculations of expected benefit as an absolute (e.g. abundance or persistence probability) or relative value (e.g. change in persistence relative to a do-nothing scenario).

The population-modelling framework has seven key steps:

1. Identify and characterise relevant threats for a given population.
2. Identify and characterise relevant management actions.
3. Specify climate change scenarios.
4. Construct a population model.
5. Link vital rates to threats and actions.
6. Project population abundances.
7. Summarise model outputs and calculate expected benefit.

The following sections describe these key steps as they apply to all species. Details of specific threats, actions, and vital rates are included in Appendices A–F.

2.2.1 Threats

For each species, threats relevant to each population were determined from the published literature and through expert elicitation (Appendices A–F). Most threats related to dynamic environmental conditions, particularly river discharge, habitat condition, and the presence of introduced fish species. Dynamic threats were included as 50-year sequences that directly altered vital rates and often differed among climate change scenarios. For example, daily measures of river discharge were compiled into a series of annual metrics capturing processes critical for each species, such as spawning cues or persistent flows during summer months. Several threats were modelled as static rather than as dynamic variables, including the effects of low genetic diversity and overexploitation. These threats were assumed to modify vital rates directly, typically reducing survival of relevant life stages in all years unless mitigated by an appropriate management intervention.

Data on river discharge, river height, and water temperature were downloaded from the Victorian Water Measurement Information System (https://data.water.vic.gov.au). Where available, daily gauge data were downloaded for 1970–2020—a 50-year sequence of data providing realistic estimates of daily, seasonal and annual variation for each site. These values were rescaled to reflect likely reductions in runoff and increases in temperature under future climates (see Section 2.2.3 Climate change scenarios, below). Missing data were common and were imputed in two steps. First, where data were missing for more than 300 days in a year, data for that year were replaced with data from a randomly selected year. This situation occurred in 0–19 years for each population, with an average of 5 years missing per population. This approach filled large gaps in gauge data while retaining seasonality in daily values. The remaining gaps in gauge data tended to span short time periods (5–20 days) and were filled with a 5-day rolling mean, calculated iteratively where gaps were longer than 5 days. These shorter gaps totalled 0–100 missing days (average = 25 days) spread over the 50-year time series in each population.

2.2.2 Management actions

Relevant management actions were identified from the published literature and expert elicitation (Appendices A–F). Management actions were assumed to modify vital rates either directly or indirectly by mitigating threats. Actions were included either as static changes in vital rates (or threats) over all years or as dynamic sequences that differed among years. Unless otherwise specified, dynamic actions were applied in all years. Many scenarios were compared for each study species: no actions, each action independently, and every feasible combination of actions. Combinations were restricted to a maximum of three actions per population for platypus because including all possible sets of actions would require over 4 billion scenarios. All actions were assumed to be implemented according to best practice, and we did not explicitly account for the feasibility of specific interventions. Therefore, outputs represent upper bounds on the expected benefit in each scenario, and the outcomes for different actions may not be comparable due to differences in their modelled effectiveness. Illustrative costs were included for each action (see Appendices A–F) to enable calculation of benefit–cost ratios (see 2.2.7 below). Cost estimates are illustrative only and are not based on a formal assessment of the costs associated with each management intervention.

Many management actions focused on water, either environmental water deliveries or the mitigation of drought impacts. Although the underlying strategies differ substantially within interventions (e.g. protection of groundwater versus releases of water from a storage), water-focused actions generally modified the underlying threats (e.g. discharge sequences) rather than altering vital rates directly. Where required, water-focused actions were separated into multiple actions, and they differed among sites. For example, environmental water deliveries were not applied to unregulated rivers and, in the case of platypus, drought refuges were considered distinct from temporary drought relief. Water-focused actions differed in their modelled effectiveness, with environmental water deliveries based on the highly constrained deliveries observed in recent decades, and drought refuges and relief based on optimistic scenarios in which drought impacts can be entirely mitigated.

Genetic-mixing and habitat-rehabilitation interventions were assumed to cause a static change in vital rates, either through changes in the survival of specific life stages or in the total population carrying capacity (the maximum number of individuals a site can support). Although the consequences of these actions are likely to change through time, either through continued changes in population genetic diversity or as habitat elements develop through time (e.g. establishment and growth of vegetation), modelling these actions dynamically would introduce substantial complexity and associated uncertainty. The impacts of genetic mixing were supported by a single study of Macquarie perch (Lutz et al. 2021), which reported early outcomes from a genetic-mixing trial but did not include data on the long-term impacts of genetic mixing (positive or negative). The likely impacts of genetic mixing in other species are largely unknown, with modelled interventions assuming a successful breeding program and consistently positive impacts of genetic mixing. Habitat rehabilitation was assumed to restore habitat to natural levels, which may be infeasible in some cases and not necessarily comparable with more-constrained actions (e.g. environmental water).

Stocking and translocations are common management tools in aquatic systems. For the purposes of this report, stocking refers to the introduction of young-of-year individuals (‘fingerlings’), whereas translocation refers to the introduction of juvenile or adult individuals. Typically, stocking introduces many individuals (tens of thousands) that are highly susceptible to mortality, and translocations introduce a smaller number of individuals (tens) with much lower expected mortality. These actions were modelled dynamically, and the number of individuals added to a population annually was drawn randomly from a distribution with known mean and variance. Survival rates post-introduction were assumed to follow those of the wild population. Stocking rates were based on observed stocking of each species in recent years. However, post-stocking survival is not known for many species, and the assumption that survival matches that of the wild population may be optimistic (Barrow et al. 2020).

Population establishment is related to translocations and involves the introduction of individuals to a new (or previously occupied) site. Population establishment is a complex process, requiring careful site selection, sourcing of individuals to be translocated, and continued monitoring and management of the new population. This process was highly simplified in the models presented here, with new populations assumed to be established successfully without removing individuals from existing populations. Importantly, new populations were assumed to be connected to an existing population, either directly (e.g. through the extension of a population’s range) or indirectly through continued translocations. These simplifications are reasonable for highly mobile species with successful hatchery or aquarium breeding programs, but may be less suitable for the two target species in this report (Yarra pygmy perch and barred galaxias). In general, assumptions of successful establishment and connectivity with existing populations are likely to be optimistic for most species.

2.2.3 Climate change scenarios

For each threat and management scenario, population abundances were projected under five different climate change scenarios. These five scenarios represented three plausible climate futures:

1. Future climate similar to observed climate from 1975–2020.
2. Future climate similar to observed climate from 1997–2020.
3. Future climate following Representative Concentration Pathway 8.5 (RCP8.5) emissions scenario projected to 2065 (three scenarios).

The first climate future assumes that future conditions will be similar to those observed during 1975–2020, which would result in wetter conditions than predicted under most global climate models (GCMs). The second future represents a step-change in climate from 1997 onwards, with future climates matching the generally drier and more volatile climatic conditions observed from 1997–2020. This future is drier than predicted in many GCMs, largely due to the influence of the Millennium drought. The third future climate is based on aggregated predictions from multiple GCMs under the RCP8.5 emissions scenario. Outputs from GCMs were used to derive catchment-specific predictions for changes in rainfall, runoff, and air temperature in the year 2065 (DELWP 2020). This approach may overestimate changes in rainfall and runoff due to climate change in the near future (prior to 2065). but is a consistent method for linking historical observations to outputs from GCMs (DELWP 2020). Given the large variation among GCMs, the RCP8.5 future scenario was separated into low-, medium- and high-change scenarios, which reflect bounds on the climatic conditions predicted to occur under an RCP8.5 emissions scenario. Low-, medium-, and high-change scenarios are the 10th, 50th and 90th percentiles of projected changes in GCM outputs, respectively, but no individual scenario is inherently more or less likely to occur than any other (DELWP 2020).

2.2.4 Constructing population models

Matrix population models are a structured method for projecting population abundances from vital rates, typically survival and reproduction (Caswell 2001). A feature of matrix population models is that they allow vital rates to differ among classes in a population. Most commonly, classes are based on age or life stage, but alternative approaches are possible, including size classes, sex, or combinations of these categories (e.g. age and sex). Regardless of how a population is categorised, matrix population models require estimates of vital rates for each class. The case studies presented here used models based on age or life stage, with vital rates determined from past research and the published literature where available (five case studies) and from life-history characteristics where vital rates were not known (one case study, barred galaxias). Population models based on past research differed in their level of development and past investment, with some models reflecting decades of continued investment and research into population processes and management interventions (Table 1). All population models tracked females only, which introduces an assumption that the number of males is not limiting reproductive output. Details of the model structures for the study species and individual populations are provided in Appendices A–F.

Alongside estimates of vital rates, matrix population models can include many other demographic processes (Caswell 2001). The six case studies included components to account for density dependence, demographic and environmental stochasticity, and associations between vital rates and environmental conditions. Density dependence is included in most matrix population models. In its simplest form, density dependence sets an upper bound on population abundances, avoiding uncontrolled population growth. More complex implementations can include details such as size- or age-dependent interactions and Allee effects (positive density dependence). Demographic and environmental stochasticity introduce random variation into matrix population models and can represent natural variation or parameter uncertainty. Demographic stochasticity is random variation in the outcomes of demographic events, such as birth and death. Environmental stochasticity is random variation in vital rates due to unpredictable or unknown changes in environmental conditions. Associations between vital rates and environmental conditions are included in population models less often than the other processes listed here, but are central to the inclusion of threats and actions in the population-modelling framework. In addition to vital rates, the parameters describing demographic processes were based on the published literature (or life-history characteristics in the absence of published estimates).

2.2.5 Linking vital rates to threats and actions

The population-modelling framework depends critically on how threats and actions affect vital rates. It is changes in the vital rates under the various threats or management action scenarios that determine changes in the expected benefit. Links between threats or actions and vital rates were based on published studies and data (where available) or, more commonly, estimated by species experts. In several cases, little was known about the effects of a particular threat or management action, in which case these effects were set at plausible but unverified values. A key benefit of focusing on ecological processes (vital rates) is that hypothesised effects of threats or actions can be tested and updated rapidly with new data.

The effects of flows management, stocking, and fishing regulations are well characterised for the large-bodied fish species [Murray cod (Todd et al. 2005; Sherman et al. 2007; Tonkin et al. 2021), Macquarie perch (Hunt et al. 2011; Tonkin et al. 2014, 2017, 2019; Todd and Lintermans 2015) and estuary perch (Stoessel et al. 2018, 2020)]. The difference between current and natural levels of instream habitat is also quantified for these species (Kitchingman et al. 2020; Tonkin et al. 2020), although the likely effects of instream habitat on fish population dynamics are less clear (but see Lyon et al. 2019). Some evidence is available to support the effects of genetic mixing on Macquarie perch (Lutz et al. 2021). The effects of the remaining actions were subject to greater uncertainty and at this stage should be viewed as hypotheses to be tested. Importantly, feasibility and effectiveness differed between management actions, so the modelled effects of some actions may differ from their realised implementations or outcomes.

2.2.6 Projecting population abundances

A set of vital rates for a population can be projected (or simulated) forward in time from an initial estimate of abundance in each age class or life stage. In this study, each case study used annual generations (one breeding event per year) projected over 50 years. Given natural variation and uncertainty in both vital rates and initial abundances, it is common to use stochastic projections of matrix population models, in which the transition from one generation to the next includes random variation (Caswell 2001). This variation results in population trajectories that differ among simulations depending on both the initial conditions and the random variation introduced in each generation. To characterise this uncertainty, population trajectories can be simulated many times, resulting in replicate trajectories. All projections in this study used 1000 replicate trajectories, each with its own set of initial abundances. Initial abundances were kept constant among climate and management action scenarios within each species to ensure that variation in initial conditions did not introduce spurious differences in outcomes between scenarios.

2.2.7 Summarising population model outputs

The primary output from a population model is an abundance trajectory (one per replicate) with a value for each age class or life stage and generation (year, in this study). Replicate abundance trajectories can be summarised in many ways (Caswell 2001). A common summary measure is the probability a population will decline below an abundance threshold beyond which extinction is inevitable, often called the quasi-extinction risk. The quasi-extinction threshold can be difficult to determine, but values larger than zero are typically used to account for the negative effects of small population sizes (e.g. inbreeding) and subsequent high extinction risk even if some individuals persist. Here, quasi-extinction thresholds were defined as 25% of population carrying capacity, to account for differences in genetic diversity among populations and species. Although this value is high relative to many population viability analyses, such values are likely necessary in order to retain sufficient genetic diversity in many populations, especially in threatened species for which population carrying capacities often are low relative to widespread, common species. The expected benefit was defined as the quasi-extinction risk for a given scenario minus that of a do-nothing scenario (i.e. population persistence probability without any management interventions). The cost-effectiveness of a scenario was estimated with benefit–cost ratios defined as expected benefit divided by the total cost of all actions included in a management action scenario.

Expected benefit was calculated at the level of individual populations and combined among populations to approximate expected benefit at the species level. Species-level estimates of expected benefit were defined in four ways. First, the geometric mean expected benefit of all populations was used to characterise the average benefit while emphasising low values (i.e. the loss of any population is a bad outcome). Second, the probability that at least one population will persist was used to represent a situation in which retaining any one population is equivalent to retaining all populations. Third, the probability that all populations will persist was used to represent a situation in which losing any population is equivalent to extinction. Last, estimates of expected benefit were combined with estimates of effective population sizes (Waples et al. 2011) to calculate a measure of expected benefit that emphasises genetic diversity over presence or absolute abundances.

2.3 Software and computational details

A critical component of this project was the development of open-source software to construct, simulate and summarise matrix population models. The purpose of this software is to support fast, flexible and reproducible population models. The developed software is included in the aae.pop package for the R programming language (R Core Team 2020). Details and demonstrations of this package are available at https://aae-stats.github.io/aae.pop, with source code available at https://github.com/aae-stats/aae.pop.

The aae.pop package uses a generic structure for population models (Figure 2). The core of this structure is a matrix of population vital rates, which is combined with any of several demographic processes to form a population dynamics object. Demographic processes are highly flexible, but the basic constructs support demographic and environmental stochasticity, two forms of density dependence (pre- and post-breeding), and associations between vital rates and covariates (e.g. environmental conditions). Dynamics objects can be used to simulate population dynamics based on specified (or randomly generated) initial conditions and any required parameters, which might include covariates or variables that affect density dependence or stochasticity. It is these parameters that specify the threat and action scenarios in the case studies presented here. The simulation step is specified by several settings, such as the number of replicate trajectories. Updates from one time step to the next use a fast algorithm for vectorised matrix multiplication, with users being able to define their own algorithms if required. The simulation step returns an array of population abundances (one value for each replicate, age class or life stage, and generation), which can be plotted directly or summarised in any way required.

A major benefit of the structure of aae.pop is the use of a single dynamics object with parameters providing relevant context (e.g. covariates, carrying capacity). This structure allows a single dynamics object to support many simulations. At present, this structure has been used to specify templates for each study species, which are then used with parameters to specify the threats and actions to be implemented in a given simulation. Population dynamics templates for each study species are included in the aae.pop.templates R package, with source code available at https://github.com/aae-stats/aae.pop.templates.

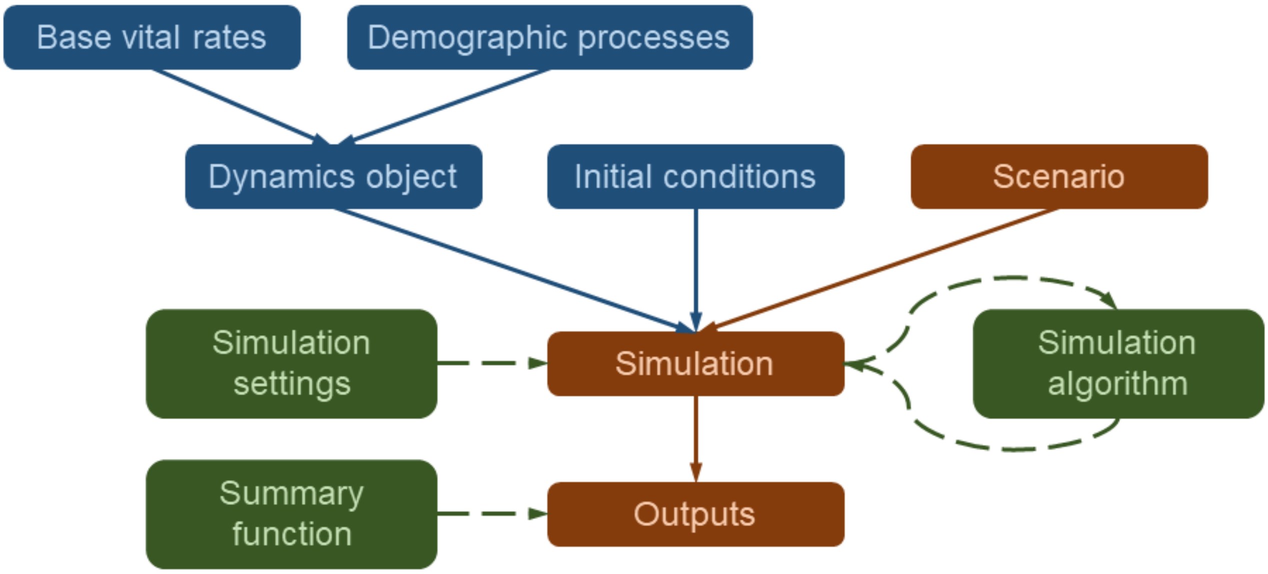

Figure 2. Overview of population-modelling software. A population dynamics object combines vital rates with specified demographic processes, such as density dependence and demographic or environmental stochasticity. Dynamics objects can be projected forward from some initial conditions, with specific details of simulation and model outputs determined by the user. Scenarios specify parameters that modify dynamics objects. Blue objects are defined once for each species, with this object being used repeatedly to simulate population dynamics under many scenarios (brown boxes). Green boxes are settings or functions that can be changed or completely redefined by a user.

3 Results

3.1 Patterns in expected benefit

Estimates of expected benefit at the level of species (aggregated over populations) highlighted some broad differences between species. When based on average persistence probability, expected benefit declined gradually across all scenarios in four species (Macquarie perch, platypus, estuary perch and Yarra pygmy perch), declined steeply in Murray cod, and was constant across the top-ranked 1000 scenarios in barred galaxias (Figure 3). These patterns suggest that there are relatively few beneficial management options for Murray cod, whereas most other species have a much larger set of beneficial management options (Figure 3). Estuary perch is difficult to characterise in this way, given the focus on a single population and the small number of total action scenarios (Figure 3). For all species except barred galaxias, expected benefit declined to values near zero in the displayed subset of scenarios, suggesting that lower-ranked management scenarios are unlikely to be effective in mitigating extinction risk (Figure 3). The consistently high estimates of expected benefit in barred galaxias reflect a single, highly effective action, which was included in a variety of combinations with other actions (see Section 3.2.6 below).

The absolute values of the expected benefit differed for the various climate change scenarios for Macquarie perch, Murray cod, platypus and estuary perch, but not for Yarra pygmy perch or barred galaxias, which had similar expected benefits for all of the climate change scenarios (Figure 3). The effects of the climate change scenarios were clearest in the top-ranked management action scenarios in Macquarie perch and Murray cod, and in lower-ranked management action scenarios in estuary perch, which suggests that estuary perch is less vulnerable to threat from climate change under the top-ranked management action scenarios (Figure 3). Differences in the responses of the species to the various climate change scenarios suggest that the expected benefit was greater in drier climate scenarios in Macquarie perch and Murray cod and in wetter scenarios in estuary perch (Figure 3). Greater benefit under more-extreme climate change scenarios may reflect the fact that benefit is defined relative to a do-nothing scenario, so the potential benefit is higher in dry scenarios due to worse outcomes under do-nothing scenarios. Differences in the absolute values of expected benefit did not substantially alter the rank order of the management action scenarios or the highest-ranked management actions (see Section 3.2 below).

The most-beneficial actions were often not the most cost-effective actions (Figure 4). This was clearest in Macquarie perch and Murray cod, for which the most cost-effective sets of management actions were ranked as being of relatively low benefit under some climate change scenarios (Figure 4). Cost-effectiveness increased with expected benefit in platypus and barred galaxias, although the cause of this pattern differed between the two species (Figure 4). In platypus, the expected benefit resulted from a combination of multiple actions, each with minor to moderate impact, so that removing any given action caused a decrease in both benefit and cost that was of similar magnitude (see Section 3.2.3 below). By contrast, the expected benefit for barred galaxias populations was dependent on a single, relatively expensive management action that was included in all of the top-ranked 1000 scenarios, so benefit and cost remained relatively constant among these scenarios (see Section 3.2.6 below). With two climate change scenario exceptions (the post-1975 and RCP8.5 low climate change scenarios), the most cost-effective actions for estuary perch were those with the highest expected benefit (Figure 4). Similar to barred galaxias, estuary perch are dependent on a single, relatively expensive action (environmental water), particularly in drying climates. For Yarra pygmy perch, the cost-effectiveness of management actions broadly matched the patterns in the expected benefit, especially under drier climates; note, the top-ranked 300 management action scenarios for this species all had equal expected benefit and are presented in no particular order in Figure 4. In both estuary perch and Yarra pygmy perch, the cost-effectiveness of the relatively low-ranked management action scenarios in wetter climates reflects the potential to achieve good outcomes with fewer actions in less-extreme climates.

‘Average persistence probability’ is one of several possible measures of expected benefit at the species level. The use of alternative measures of expected benefit can substantially alter patterns in expected benefit among scenarios. When based on effective population sizes, expected benefit was often high under scenarios that had low average persistence probabilities (Figure 5). This pattern was particularly prominent in Murray cod and Yarra pygmy perch and, to a lesser degree, in Macquarie perch (Figure 5). In these species, it is possible that management action scenarios with low average persistence probabilities still might support highly effective population sizes, due to the large genetic diversity in some populations, which would become highly relevant if these management scenarios also had a low cost (Figure 5). Importantly, the expected benefit remained high in scenarios with high average persistence probabilities (Figure 5). Two other measures of expected benefit are (i) the probability that all populations persist and (ii) the probability that at least one population persists. The probability that all populations would persist was similar to the average persistence probability, but declined more steeply across management action scenarios and had lower absolute values under the top-ranked management action scenarios for platypus and Macquarie perch (Figure G1). The probability that at least one population would persist was similar to effective population size, with relatively high values even in management action scenarios with low average persistence probabilities (Figure G2). The different results obtained using different measures of expected benefit are due entirely to the definitions of these measures, which highlights the fact that the choice of measure used can have substantial impacts on inferences and subsequent decisions.

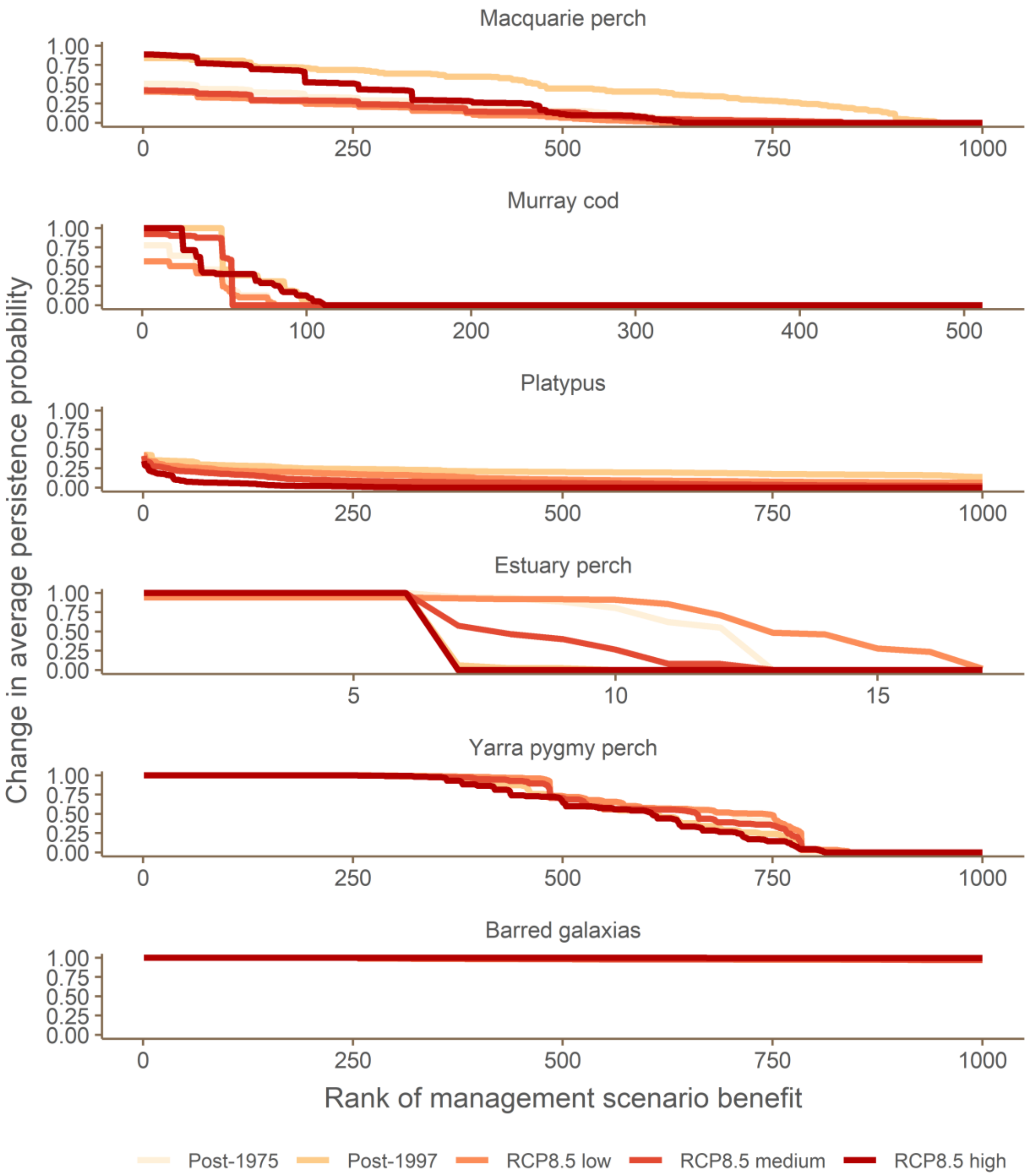

Figure 3. Changes in expected benefit across the top-ranked combinations of management actions for each species. The *x*-axis indicates the rank of each management scenario, and the *y*-axis indicates the change in the average persistence probabilities for all populations (expected benefit) for each management scenario relative to a do-nothing scenario. Scenario ranks are ordered from most to least beneficial, so that low rank numbers denote higher probabilities of population persistence. A maximum of 1000 scenarios is shown for each species. The top-ranked (far left) scenario is listed in Table 2, and the frequency of occurrence of management actions in highly ranked scenarios is listed in Table 3.

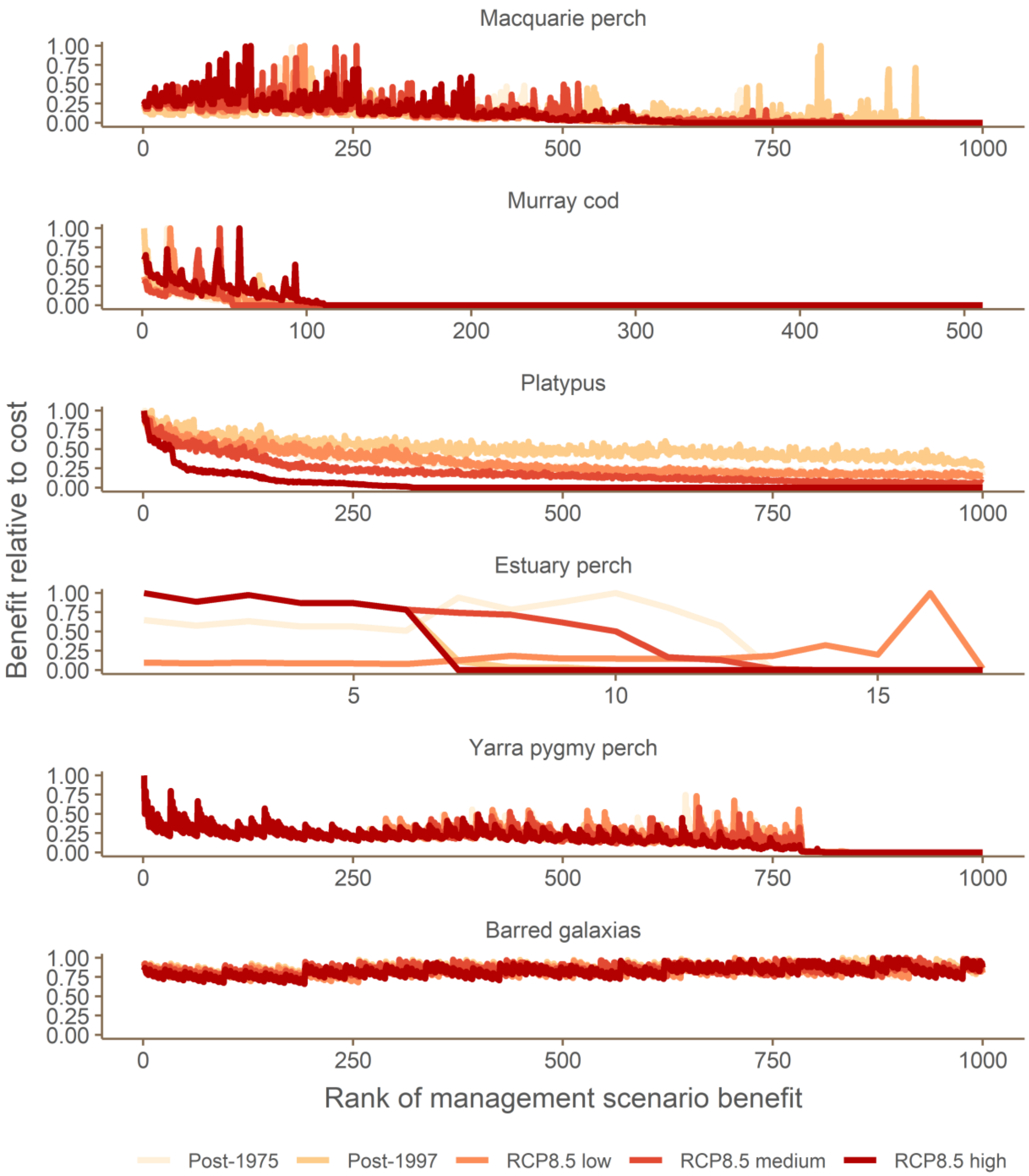

Figure 4. Changes in cost-weighted benefit across the top-ranked combinations of management actions for each species. The *x*-axis indicates the rank of each management scenario based on average persistence probabilities (from Figure 3), and the *y*-axis indicates the benefit (average population persistence) relative to cost for each management scenario. Scenario ranks are ordered from most to least beneficial, so that low rank numbers denote higher probabilities of population persistence. Values are standardised so that the maximum benefit–cost ratio under each climate scenario is equal to 1. A maximum of 1000 scenarios is shown for each species. RCP8.5, Representative Concentration Pathway 8.5.

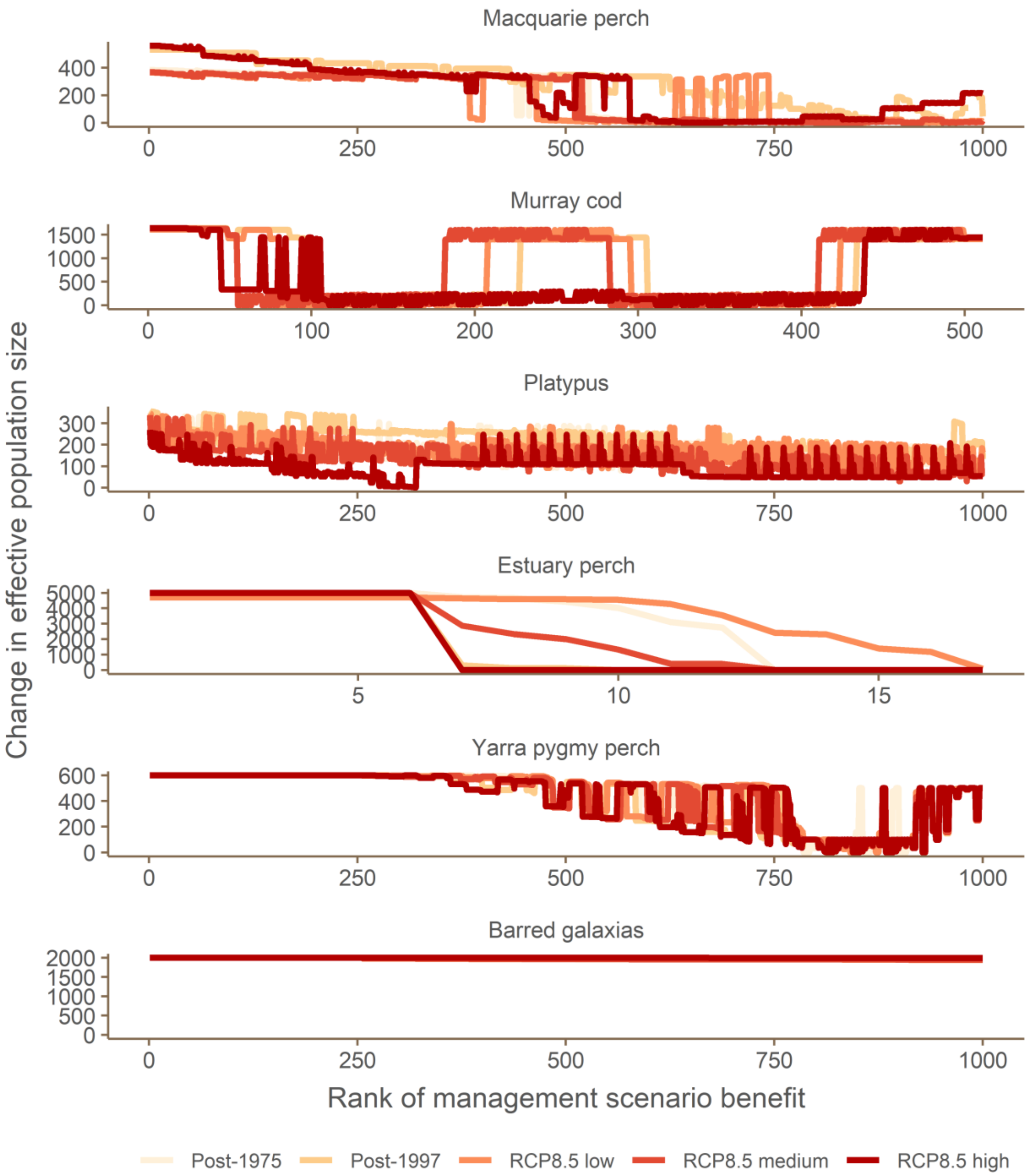

Figure 5. Changes in expected benefit based on effective population size across the top-ranked combinations of management actions for each species. The *x*-axis indicates the rank of each management scenario based on average persistence probabilities (from Figure 3), and the *y*-axis indicates the change in weighted effective population sizes for each management scenario relative to a do-nothing scenario. Scenario ranks are ordered from most to least beneficial, so that low rank numbers denote higher probabilities of population persistence. A maximum of 1000 scenarios is shown for each species.

3.2 Species case studies

This report describes emergent, high-level results relevant to the use of population models for estimating expected benefit for many species in many places. As a result, the following focuses on results at the species level (i.e. aggregated over all populations). Many applications of population models seek to determine the responses of each population in detail, which is highly relevant to management decisions targeting conservation outcomes for one or a few populations of a given species.

3.2.1 Macquarie perch

The most beneficial set of actions for Macquarie perch included all possible actions in Seven Creeks; genetic mixing and fishing regulations in the Yarra River; and stocking in Lake Dartmouth (Table 2). Genetic mixing replaced stocking in Lake Dartmouth under the RCP8.5 medium climate scenario (Table 2). Aggregating across all highly ranked management action scenarios, defined as those with expected benefit within 0.05 of the top-ranked scenario, the most frequently included actions were stocking in Seven Creeks, and fishing regulations and genetic mixing in the Yarra River (Table 3). No actions were included consistently in highly ranked scenarios in Lake Dartmouth (Table 3).

3.2.2 Murray cod

Murray cod had the best outcomes when fishing regulations were applied to all three populations, with instream habitat rehabilitation in the Goulburn River and environmental water under the RCP8.5 medium climate scenario in the Campaspe River (Table 2). Fishing regulations were included in all highly ranked management action scenarios, as was stocking in the Campaspe River (Table 3). Environmental water was included in 70% of highly ranked scenarios under the RCP8.5 medium climate scenario (Table 3).

3.2.3 Platypus

Drought refuges or relief and habitat rehabilitation (instream and riparian) led to the best outcomes for platypus (Table 2). These actions were included consistently in highly ranked management action scenarios; note, only one scenario was within the 0.05 threshold of the top-ranked scenario (Table 3). In addition, platypus scenarios were restricted to a maximum of three actions per population, because allowing all actions simultaneously would have required consideration of more than 4 billion scenarios. If this restriction were removed, it is likely that additional actions would improve outcomes for platypus populations.

3.2.4 Estuary perch

The delivery of environmental water was critical to the persistence of estuary perch in the Snowy River (Table 2). The top-ranked management action scenario included large environmental water allocations (45 000 ML per year), with a smaller allocation (20 000 ML per year) being insufficient to support estuary perch except in the wettest climates (RCP8.5 low, post-1975) (Figure 3). Stocking and instream habitat rehabilitation were included in some highly ranked scenarios, but these actions were largely ineffective in the absence of large environmental water allocations (Table 3).

3.2.5 Yarra pygmy perch

The top-ranked management action scenario included genetic mixing in both Yarra pygmy perch populations (Table 2). Genetic mixing was included in 70–80% of highly ranked scenarios, with other actions being included in 50–60% of these scenarios (Table 3). The consistently high expected benefit in many management action scenarios including genetic mixing indicated that this management action is critical to the persistence of Yarra pygmy perch populations, with other actions improving population outcomes to a lesser extent (Figure 3).

3.2.6 Barred galaxias

Barred galaxias were dependent on the control and removal of introduced fish species, or the establishment of new populations; note, population establishment implicitly included control of introduced fish species at the newly established site (Table 2). Genetic mixing and *ex situ* care improved population outcomes and were included in 50–70% of highly ranked scenarios, but they were ineffective without some form of predator control (Table 3). The consistently high expected benefit for barred galaxias with some form of introduced fish species control (Figure 3) indicated the centrality of introduced fish species control among the many possible combinations of management actions and populations; note, the approximately 2000 highly ranked scenarios all included either control and removal of introduced fish species, or population establishment (Table 3).

Table 2. Top-ranked management scenarios in each case study

The most-beneficial management actions are listed for each species under a post-1975 climate scenario that reflects historical conditions and an RCP8.5 medium climate scenario that reflects reduced runoff predicted under the RCP8.5 emissions scenario. Rankings are based on expected benefit, defined as average (geometric mean) persistence probability of all populations in a species minus average persistence probability under a do-nothing scenario. Average persistence is one possible measure of expected benefit, and three alternative measures are included. The top-ranked scenario based on average persistence is not necessarily the top-ranked scenario under other measures of expected benefit, and lower-ranked scenarios may have similar absolute values of expected benefit (see Table 3). Benefit–cost ratios are based on average persistence probabilities divided by relative costs, which are not comparable among species.

Species	Climate	Actions	Change in average persistence probability	Change in probability that at least one population persists	Change in probability that all populations persist	Change in effective population size	Benefit–cost: persistence probability
Macquarie perch	Post-1975	Seven Creeks: genetic mixing, stocking, instream habitat rehabilitation, flow protection Yarra River: genetic mixing, fishing regulations Lake Dartmouth (Mitta Mitta River): stocking	0.51	0.08	0.13	383	1.53
	RCP8.5 medium	Seven Creeks: genetic mixing, stocking, instream habitat rehabilitation, flow protection Yarra River: genetic mixing, fishing regulations Lake Dartmouth (Mitta Mitta River): genetic mixing	0.42	0.08	0.08	367	1.1
Murray cod	Post-1975	Goulburn River: instream habitat rehabilitation, fishing regulations Campaspe River: stocking, fishing regulations Murray River: fishing regulations	0.78	1.00	0.47	1619	3.45
	RCP8.5 medium	Goulburn River: instream habitat rehabilitation, fishing regulations Campaspe River: fishing regulations, environmental water Murray River: fishing regulations	0.92	1.00	0.78	1631	1.94
Platypus	Post-1975	Upper Campaspe River: permanent drought refuges, riparian revegetation, instream habitat	0.42	0.95	0.03	349	0.43

		rehabilitation Mackenzie River: permanent drought refuges, riparian revegetation, instream habitat rehabilitation Ovens River: temporary drought relief, riparian revegetation, instream habitat rehabilitation Little Yarra River: environmental water, temporary drought relief, riparian revegetation					
	RCP8.5 medium	Upper Campaspe River: permanent drought refuges, riparian revegetation, instream habitat rehabilitation Mackenzie River: permanent drought refuges, riparian revegetation, instream habitat rehabilitation Ovens River: temporary drought relief, riparian revegetation, instream habitat rehabilitation Little Yarra River: environmental water, temporary drought relief, riparian revegetation	0.41	0.94	0.03	338	0.43
Estuary perch[1]	Post-1975	Snowy River: environmental water (large allocation)	1.00	1.00	1.00	5000	0.50
	RCP8.5 medium	Snowy River: environmental water (large allocation)	1.00	1.00	1.00	5000	0.50
Yarra pygmy perch	Post-1975	Deep Creek: genetic mixing Armstrong Creek: genetic mixing	1.00	1.00	1.00	600	1.00
	RCP8.5 medium	Deep Creek: genetic mixing Armstrong Creek: genetic mixing	1.00	1.00	1.00	600	1.00
Barred galaxias	Post-1975	Rubicon River: predator detection and removal, population establishment Howqua River: predator detection and removal, population establishment Delatite River: predator detection and removal, population establishment Perkins Creek: predator detection and removal,	1.00	1.00	1.00	2000	1.25

		population establishment					
	RCP8.5 medium	Rubicon River: predator detection and removal, genetic mixing, population establishment Howqua River: predator detection and removal, population establishment Delatite River: predator detection and removal, population establishment Perkins Creek: predator detection and removal, population establishment	1.00	1.00	1.00	2000	1.18

[1]Estuary perch represented by a single population (Snowy River). RCP8.5, Representative Concentration Pathway 8.5.

Table 3. Frequency of actions in highly ranked scenarios

This table provides the proportion of highly ranked scenarios including each action. Highly ranked scenarios are those with an expected benefit within 0.05 of the most-beneficial scenario listed in Table 2. The estimates of expected benefit under each action are sensitive to the modelled effectiveness of each action, which is optimistic in some cases (e.g. habitat rehabilitation, drought mitigation) and highly constrained in others (e.g. environmental water, flow protection), so the results here represent hypotheses to be tested in field studies. Values of *n* (left-hand column) denote the number of highly ranked scenarios under the post-1975 and RCP8.5 medium climate scenarios, respectively. Actions in boldface are those included in more than 70% of highly ranked scenarios under either climate change scenario.

Species	Population	Action	Proportion under post-1975 climate change scenario	Proportion under RCP8.5 medium climate change scenario
Macquarie perch (*n* = 64, 112)	Seven Creeks	Genetic mixing	0.5	0.5
		Stocking	1.0	1.0
		Instream habitat rehabilitation	0.5	0.5
		Flow protection	0.5	0.5
	Yarra River	**Fishing regulations**	1.0	1.0
		Genetic mixing	1.0	1.0
		Environmental water	0.0	0.0
	Lake Dartmouth (Mitta Mitta River)	Fishing regulations	0.5	0.5
		Genetic mixing	0.5	0.5
		Stocking	0.5	0.5
Murray cod (*n* =16, 46)	Goulburn River	**Fishing regulations**	1.0	1.0
		Instream habitat rehabilitation	0.5	0.5
		Stocking	0.5	0.5
		Environmental water	0.5	0.5
	Campaspe River	**Fishing regulations**	1.0	1.0
		Stocking	1.0	0.7
		Environmental water	0.0	0.7
	Murray River	**Fishing regulations**	1.0	1.0
		Environmental water	0.5	0.5
Platypus (*n* = 2, 1)	Upper Campaspe River	Temporary drought relief	0.5	0.0

		Permanent drought refuges	0.5	1.0
		Riparian revegetation	1.0	1.0
		Instream habitat rehabilitation	1.0	1.0
		Barrier removal	0.0	0.0
		Environmental water	0.0	0.0
	Mackenzie River	**Permanent drought refuges**	1.0	1.0
		Riparian revegetation	1.0	1.0
		Instream habitat rehabilitation	1.0	1.0
		Temporary drought relief	0.0	0.0
		Barrier removal	0.0	0.0
		Environmental water	0.0	0.0
		Genetic mixing	0.0	0.0
		Translocations	0.0	0.0
	Ovens River	**Temporary drought relief**	1.0	1.0
		Riparian revegetation	1.0	1.0
		Instream habitat rehabilitation	1.0	1.0
		Barrier removal	0.0	0.0
		Environmental water	0.0	0.0
	Little Yarra River	**Temporary drought relief**	1.0	1.0
		Riparian revegetation	1.0	1.0
		Environmental water	1.0	1.0
		Instream habitat rehabilitation	0.0	0.0
		Barrier removal	0.0	0.0
Estuary perch (n = 6, 6)	Snowy River	**Environmental water (large allocation)**	1.0	1.0
		Environmental water (small allocation)	0.0	0.0
		Instream habitat rehabilitation	0.5	0.5
		Stocking (high)	0.3	0.3
		Stocking (low)	0.3	0.3
Yarra pygmy perch (n = 458, 398)	Deep Creek	**Genetic mixing**	0.8	0.8

		Population establishment	0.6	0.6
		Wetland habitat rehabilitation	0.5	0.5
		Drought refuges	0.5	0.6
		Predator detection and removal	0.6	0.6
	Armstrong Creek	**Genetic mixing**	0.7	0.8
		Population establishment	0.6	0.6
		Wetland habitat rehabilitation	0.5	0.5
		Drought refuges	0.5	0.6
		Predator detection and removal	0.5	0.5
Barred galaxias (*n* = 2484, 1867)	Rubicon River	*Ex situ* care	0.6	0.6
		Genetic mixing	0.6	0.6
		Introduced fish species ssdetection and removal	0.8	0.9
		Population establishment	0.9	0.9
	Howqua River	*Ex situ* care	0.6	0.5
		Genetic mixing	0.6	0.7
		Introduced fish species detection and removal	0.8	0.8
		Population establishment	0.8	0.8
	Delatite River	*Ex situ* care	0.5	0.5
		Genetic mixing	0.7	0.7
		Introduced fish species detection and removal	0.9	0.8
		Population establishment	0.8	0.8
	Perkins Creek	*Ex situ* care	0.6	0.6
		Genetic mixing	0.7	0.7
		Introduced fish species detection and removal	0.8	0.8
		Population establishment	0.8	0.8

RCP8.5, Representative Concentration Pathway 8.5.

4 Discussion

This report presents a population-modelling framework for predicting the effects of management actions on population persistence under multiple threat scenarios. Six case studies of threatened species illustrated how this approach quantifies the key processes that determine population dynamics and links these processes to long-term (50-year) population outcomes under various threats and management actions. This report presents a framework in which population models can be used to link management actions with expected benefit. The results of this study are subject to multiple uncertainties (particularly in the definitions of the various management actions) and are dependent on assumptions; they should not be used to compare specific management actions without additional testing and verification. Such uncertainties and assumptions are embedded in all biodiversity decision-support tools. One benefit of the approach illustrated here is that these assumptions and uncertainties are made explicit. The population-modelling framework employed was developed using new, open-source software for modelling population dynamics, which supports fast simulation of population outcomes and rapid updating of processes, threats and actions as new information becomes available.

4.1 Patterns in expected benefit

The rankings of expected benefit across many management action scenarios revealed several benefit ‘archetypes’, with species conforming to one of three patterns. First, three species (Macquarie perch, platypus and Yarra pygmy perch) had steady declines in expected benefit from the top-ranked to lower-ranked scenarios. For these species, management actions tended to cause incremental changes in population outcomes, so the addition or removal of a single action rarely caused a large change in expected benefit. Second, one species (Murray cod) had a rapid decline in benefit beyond the approximately 50 top-ranked scenarios, which reflected the effectiveness of a few critical actions, without which at least one of the target populations would not persist. Third, one species (barred galaxias) had consistently high estimates of expected benefit across many management action scenarios. Given that expected benefit is defined relative to a do-nothing scenario (i.e. high values indicate a need for at least some management intervention), this pattern suggests that many different combinations of management actions are equally effective. In this case, barred galaxias management is still constrained due to their dependence on the detection and removal of introduced fish species, which was included in all of the top 1000 management action scenarios due to a large number of possible action combinations (4 populations and 4 actions gives 2^{16} = 65 536 combinations). Estuary perch broadly conformed to the first archetype (gradual declines in benefit), but are difficult to categorise with certainty, given the case study included only a single population at a single location and relatively few sets of actions. Case studies of more species would clarify whether the observed archetypes result from characteristics of the species or from the definitions of management actions.

Identifying the consequences of a species’ benefit archetype with respect to management decisions may help identify the situations in which a population-modelling approach will (or will not be) informative. The first archetype (gradual declines in benefit) indicates that there can be substantial flexibility in management decisions, with many potentially effective actions that may be further distinguished based on metrics such as cost. The case study of Macquarie perch presents an example of this situation, with moderate to large differences between estimates of expected benefit and cost-effectiveness (compare Figure 3 and Figure 4). However, this pattern was less apparent in the case studies of platypus and Yarra pygmy perch because, for these species, incremental changes in benefit coincided with incremental changes in cost, resulting in similar rankings of management action scenarios based on benefit and cost-effectiveness. The second archetype (rapid declines in benefit) indicates there is a highly constrained set of beneficial decisions, with a small number of essential actions and little flexibility in their implementation. This pattern may reflect ecological and practical constraints on the implementation and effectiveness of management interventions, or a constrained analysis with too few management options. The final archetype (constant benefit across many scenarios) indicates high flexibility in management decisions. Although barred galaxias were highly dependent on a single action (predator detection and removal), flexibility arises around the inclusion of other possible management interventions alongside predator control.

Differences among climate change scenarios did not substantially alter the rank order of management action scenarios or the most-beneficial actions at the population or species level. Similar responses to different climates may be due to the inclusion of the Millennium drought in all climate change scenarios. Although discharge and water temperature differed among climate change scenarios, the extreme nature of the Millennium drought may have caused population declines even under moderate climate change scenarios (post-1975, RCP8.5 low). With few exceptions, the top-ranked scenarios were consistent among climate change scenarios, regardless of the measure of benefit used. Exceptions to this pattern arose primarily in the patterns of cost-effectiveness, where smaller sets of actions sometimes were more cost-effective under less-extreme climates. At the population level, the specific actions included in highly ranked scenarios were almost identical between the two climate change scenarios shown. Climate change scenarios did affect the absolute values of expected benefit, so the need for management intervention still may depend on climate.

4.2 Benefits of population models

The development of a population-modelling framework for the estimation of expected benefit was undertaken due to the advantages of this approach compared with commoner elicitation methods. First, population models require explicit, quantitative estimates of management impacts. All six case studies had data or published studies providing this information (described in Appendices A–F), including the magnitude of each impact as well as specification of the underlying mechanisms. Specifying the effects of management actions in this way encourages clarity around why a particular action could be expected to be effective and, in some cases, can provide common ground that improves communication between modellers and species experts. Attempts to quantify the effects of actions can also identify knowledge gaps, the consequences of which can be explored in the low-cost, low-risk, virtual environment of a population model. Second, population models are transferrable to new systems or scenarios, albeit with some risk of less-reliable outputs due to extrapolation. This transferability allowed the impact of combinations of actions to be modelled, despite the underlying models only requiring estimates of the independent effects of individual actions. Similarly, most case studies were based on population models that were developed for a single system, with these same models being transferred to new systems with minimal changes (e.g. updated carrying capacity, new environmental variables). The transferability of population models also allows rapid updating of models in response to new information or a new decision context. This is a core feature of the population-modelling software described in this report—a single population model (a 'template') can be used with multiple scenarios, parameters, and summary functions.

A broader benefit of population models, and process-explicit models more generally, is their focus on measurable processes, both as inputs and outputs. The input processes that specify a population model can be measured, tested and updated as required, although the required field studies may be logistically challenging in many cases. Measurable outputs are useful from several perspectives. Population models directly predict quantities of interest (e.g. persistence, and long-term abundance trends), along with intermediate outputs (e.g. annual abundances). These outputs can be used to validate model predictions on short time scales and to make predictions about specific demographic processes, such as recruitment and population age structure. In addition, short-term predictions of quantities such as abundance can be used to communicate the likely outcomes of management actions to stakeholders with less uncertainty than occurs with long-term predictions (e.g. 50-year persistence).

4.3 Further work

This report presented six case studies based on population models with different levels of development (Table 1). The less-developed models require expert assessments to determine the appropriateness of the target populations and the feasibility and potential effectiveness of the included management actions. The more-developed models, particularly those of Macquarie perch and Murray cod, include detailed site descriptions and actions based on current management of these populations. In these cases, the critical next step is to begin validating these models with new data to identify where and when model predictions are reliable. Assessing model predictions in this way will highlight gaps in the current knowledge of species biology and will test the hypothesised responses to management interventions. Importantly, the need to validate models does not preclude decisions based on the model outputs. Management of both Macquarie perch and Murray cod populations is currently informed by outputs of the population models used here (Koehn and Todd 2012; Todd and Lintermans 2015); the implementation and monitoring of highly ranked actions is an important aspect of model validation. Implementing and testing various management actions in the field is critical, because practical constraints may result in the implemented actions being substantially different from those used in the modelling. Input from the managers involved will allow models to be refined based on current management practices and constraints.

Stochastic population models incorporate uncertainty at multiple steps and propagate this uncertainty through to the model outputs. This uncertainty can reflect true variation in ecological processes, as well as lack of knowledge about specific processes. Sensitivity analysis is a common technique for identifying the sources and consequences of uncertain model inputs, and is a logical and straightforward extension of the case studies presented here (Caswell 2001; Dietze et al. 2018). A broader challenge is to generate informative summaries of model outputs without dismissing model uncertainty. The population-modelling framework generates detailed outputs at the level of individual populations. The results presented in Table 2 simplify this detailed output by focusing on the most beneficial set of actions (for each population of each species), whereas the results in Table 3 indicate the level of consistency within a potentially large number of highly ranked scenarios. The usefulness of the different summaries will depend on the decision context, but it is likely that new approaches will be required to capture fully the uncertainty within populations and the differences between populations.

Most process-explicit models create close mappings from inputs to outputs. A consequence of this mapping is that modelling decisions, such as mathematical descriptions of management actions, can have a strong influence on model outputs. An example of this in the present study is the definition of population establishment. New populations were modelled as fully connected extensions of existing populations but with key threats removed. This definition was used for computational convenience, but it resulted in potentially optimistic assessments of population establishment that implicitly incorporated multiple interventions (e.g. predator removal, habitat rehabilitation, translocations). A more realistic assessment of population establishment would require details of suitable sites without extant populations, with this information being used to introduce site-specific threats and necessary management interventions. The strong influence of modelling decisions extends to model summary measures. This report presented four measures of expected benefit that had a common basis (quasi-extinction risk) but differed in their interpretations. It would be informative to explore a wider range of summary measures and determine the decision contexts under which different measures are suitable.

It is unknown whether population-model outputs can be used for large-scale prioritisation of management actions for many species or taxa (e.g. Thomson et al. 2020). Two challenges arise in relation to this. First, planning for multiple species requires consideration of an expanded set of management actions, so large-scale, often spatially explicit management actions must be included as inputs to population models. Similarly, the impacts of population-specific management actions (e.g. habitat restoration, stocking) on non-target species need to be estimated, with these actions included alongside other actions in the prioritisation process. Second, population-level responses need to be estimated for all extant populations, including newly established populations, with these responses then aggregated into a species-level metric of persistence that can be used alongside existing decision-support tools (e.g. Strategic Management Prospects and Specific Needs). This report demonstrates several ways in which estimates of (population-level) benefit can be aggregated for multiple populations. However, extending population models to include all known populations may require substantial extrapolation to new locations, in which case the reliability of extrapolated model outputs would require careful assessment. Additionally, extending population models in this way will likely require distinct models for different locations or contexts (e.g. with location-dependent actions, climates, ecosystems, and river morphology). An alternative approach might consider closer integration of spatial threat maps and population models, with spatial information incorporated directly into a spatially explicit population-modelling framework (e.g. Visintin et al. 2020).

4.4 Conclusion

Estimating the expected benefit of management interventions requires detailed knowledge of species biology, management constraints, and a range of threatening processes. Although expert elicitation is an efficient and useful method for collating this knowledge, the mechanisms and assumptions underpinning expert assessments are often implicit or unspecified. Population models are a complementary approach for the estimation of expected benefit. Population models still require expert knowledge of species and systems, but capture this knowledge in explicit, quantitative statements based on measurable ecological processes. The benefits and limitations of population models stem from this focus on ecological processes, which increases the complexity of species assessments while greatly improving their interpretability and testability. Comparisons of population-model outputs with equivalent, expert-elicited outputs would shed considerable light on the similarities and differences between these two approaches.

This report discusses the development of new, open-source software for the simulation of population dynamics, as illustrated through six case studies of threatened species in Victoria. These case studies demonstrated that population models can be developed from a range of levels of existing knowledge, with model outputs likely to be most informative when based on detailed, species-specific knowledge. Such knowledge often exists for species with high economic, cultural or conservation value. However, even when knowledge of population processes was limited, the approach introduced here still generated quantitative outputs that might inform decisions or clarify the need for additional field studies or expert elicitations. Although widespread development of population models for many species is likely to be infeasible, this report illustrates an approach to population modelling that is neither conceptually nor computationally demanding. The development of a preliminary population model for a previously unmodelled species can be as simple as a literature search and some basic model checks (see Appendix F and https://aae-stats.github.io/aae.pop/ for examples). Regardless of whether a model is preliminary or highly developed, a critical next step is to assess and increase the reliability of these models through an iterative cycle of validation and model updating.

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Appendix A: Macquarie perch case study

The Macquarie perch population model was based on the model described in Todd and Lintermans (2015) (Table A1). Todd and Lintermans's model includes age-specific and density-dependent rates of survival and reproduction for all life stages and ages, from eggs to a maximum age of 30 years, and can accommodate the stocking and harvesting of individuals of any age. We extended the model to include the effects of hydrological conditions on survival and reproduction (Tonkin et al. 2017, 2019), the effects of instream habitat availability on maximum population size (Kitchingman et al. 2020; Tonkin et al. 2020), and the effects of stocking with genetically diverse, hybrid individuals, recently shown to improve early life survival (Lutz et al. 2021). Specific actions and their effects on vital rates are described in Table A2.

This case study focuses on three extant Victorian populations of Macquarie perch (in Seven Creeks, the Yarra River and Lake Dartmouth). These populations span a range of population sizes, genetic diversity, ecosystem types, and stocking histories, and are broadly representative of all known Victorian populations of Macquarie perch.

Table A1. Structure of Macquarie perch population model

Table A1 provides details of the population structure and the key parameter values for the models of Macquarie perch population dynamics in each population. Specific parameterisations and R code for each model are available at https://github.com/aae-stats/aae.pop.templates.

Model component	Description	References
Matrix type	Leslie matrix	Todd and Lintermans (2015)
Number of classes	Juveniles: eggs, larvae, young-of-year, 1–2 years old Adults: 3–30 years old	
Sex ratio	Model included females only, with males assumed not to be limiting due to an approximately even (50:50) sex ratio.	
Density dependence	Positive: Allee effect on egg production due to fewer available mates when population sizes are small Negative: Top-down density dependence based on total habitat availability; defined by population-specific carrying capacities	Todd and Lintermans (2015)
Carrying capacity	Carrying capacity, defined as the maximum number of breeding adult females supported by the available habitat, determines the strength of the negative density dependence. Parameter estimates were: Lake Dartmouth (Mitta Mitta River): 1000 Seven Creeks: 100 Yarra River: 1000	Pavlova et al. (2017) Tonkin et al. (2018, 2020)
Environmental stochasticity	Unit-scaled Gaussian variation in survival estimates and zero-truncated normal variation in reproduction estimates	Todd and Ng (2001) Todd and Lintermans (2015)
Demographic stochasticity	Poisson variation in number of recruits and binomial variation in number of surviving adults	Todd and Lintermans (2015)
Covariate effects	Flow conditions affected Macquarie perch population dynamics. All flow variables were defined based on water years (July–June), with the following effects: 1. Low flows in rivers have a negative effect on survival of individuals aged 2 years and older (not applicable in Lake Dartmouth). 2. Very low or very high flows during the spawning period (Nov–Dec) have a negative effect on reproduction and recruitment. 3. Highly variable flows during the spawning period (Nov–Dec) have a negative effect on recruitment. 4. Increases in mean daily flow or lake level relative to the previous year have a positive effect on recruitment.	Tonkin et al. (2017, 2019)

Table A2. List of actions included in Macquarie perch scenarios

Table A2 provides descriptions of the management actions applied to each population. Population trajectories were simulated for every feasible combination of management actions in each system. Relative costs (per year and location) were included to demonstrate the calculation of benefit–cost ratios. All costs are illustrative only and may not reflect actual cost estimates for each action.

Action	Relevant populations	Description	Relative cost	Uncertainties
Gene mixing	Seven Creeks Yarra River Lake Dartmouth	Stocking of hybrid offspring (from Lake Dartmouth × Yarra River broodfish), demonstrated to increase survival of eggs, larvae and young-of-year by approximately 80% in Seven Creeks and Lake Dartmouth, and by approximately 10% in the Yarra River (Lutz et al. 2021).	1000	Limited data on survival and reproductive output of adult hybrids due to the short period since the hybridisation program began in 2013
Stocking	Seven Creeks Lake Dartmouth	Stocking of approximately 20 000 fingerlings into a system, assuming a 50:50 sex ratio and fingerling survival equal to that of the natural population.	500	Limited data on survival and reproductive output of stocked individuals, particularly on whether outcomes for stocked individuals differ from those of natural recruits
Fishing regulations	Yarra River Lake Dartmouth	Produce educational material on regulation of illegal harvesting of adult Macquarie perch. Assumed baseline losses followed a Poisson distribution with a mean of 10 adult females removed per year, with regulations reducing this number to zero (Hunt et al. 2011).	25	Numbers of individuals caught and removed not known with certainty Effectiveness of regulation uncertain
Instream habitat rehabilitation	Seven Creeks	Increase available instream habitat to estimated natural (historical) levels (Kitchingman et al. 2020), estimated to increase Macquarie perch carrying capacity by approximately 50% (Tonkin et al. 2020).	100	Estimates of current carrying capacity are uncertain in Seven Creeks. Uncertainties in the feasibility of increasing instream habitat to natural levels, and in the impacts of these increases on population carrying capacity
Protection of natural flows	Seven Creeks	Decommissioning 50 dams above Gooram Weir, with individual dams selected through an optimisation process to maximise natural runoff into the Seven Creeks system (HARC 2020).	200	Uncertainties in the effectiveness of decommissioning dams
Environmental water	Yarra River	Adjust discharge to reflect current seasonal watering plan in the Yarra River.	300	Uncertainties in the availability of environmental water in each year, especially under climate change

Appendix B: Murray cod case study

The population model used here was based on an existing model of Murray cod population dynamics (Todd et al. 2004) (Table B1). Todd et al.'s model includes age-specific and density-dependent rates of survival and reproduction of all life stages and ages, from eggs to a maximum age of 50 years, and can accommodate the stocking and harvesting of individuals of any age. Harvesting is incorporated with a submodel that relates fish ages to lengths, because current fishing regulations use a length-based slot limit (550–750 mm). We extended this model to include the effects of hydrological conditions on recruitment and survival (Tonkin et al. 2021), and the effects of instream habitat availability on maximum population size (Tonkin et al. 2020). Specific actions and their effects on vital rates are described in Table B2.

This case study focuses on populations of Murray cod in three rivers in the southern Murray–Darling Basin: the Goulburn River (downstream of Lake Nagambie), the Campaspe River (downstream of Elmore) and the Murray River (Yarrawonga to Barmah). These populations span a range of stocking histories, recreational and commercial fishing histories, and local environmental conditions (e.g. discharge conditions and instream habitat availability). The three populations are genetically distinct and are representative of most self-sustaining populations of Murray cod in the southern Murray–Darling Basin.

Table B1. Structure of Murray cod population model

Table B1 provides details of the population structure and the key parameter values for models of the Murray cod population dynamics in each population. Specific parameterisations and R code for each model are available at https://github.com/aae-stats/aae.pop.templates.

Model component	Description	References
Matrix type	Leslie matrix	Todd et al. (2004, 2005) Sherman et al. 2007)
Number of classes	Juveniles: eggs, larvae, young-of-year, 1–4 years old Adults: 5–50 years old	
Sex ratio	Model included females only, with males assumed not to be limiting, due to an approximately even (50:50) sex ratio	
Density dependence	Negative: Top-down density dependence based on total habitat availability, defined by population-specific carrying capacities	Todd et al. (2004, 2005) Sherman et al. 2007)
Carrying capacity	Carrying capacity, defined as the maximum number of breeding adult females supported by the available habitat; determines the strength of negative density dependence. Parameter estimates were: Goulburn River: 10 000 Campaspe River: 1000 Murray River: 10 000	Pavlova et al. (2017) Tonkin et al. (2020)
Environmental stochasticity	Unit-scaled Gaussian variation in survival estimates and zero-truncated normal variation in reproduction estimates	Todd and Ng (2001) Todd et al. (2004, 2005) Sherman et al. 2007)
Demographic stochasticity	Poisson variation in number of recruits and binomial variation in number of surviving adults	Todd et al. (2004, 2005) Sherman et al. 2007)
Covariate effects	Flow conditions affected Murray cod population dynamics. All flow variables were defined based on water years (July–June), with the following effects: 1. Extreme low flows (below 5 ML/day) result in a 20% reduction in the survival of individuals aged 2 years and older. 2. Antecedent maximum daily flow has a positive effect on recruitment in the Murray River and Goulburn River and a negligible effect in the Campaspe River. 3. Increased spring flow has a positive effect on recruitment. 4. Increased summer flow has a negative effect on recruitment. 5. Increased winter flow has a positive effect on recruitment in the Murray River and Goulburn River and a negligible effect in the Campaspe River.	Tonkin et al. (2021)

	6. Increased variability in spawning flow has a negative effect on recruitment in the Murray River and Goulburn River and a negligible effect in the Campaspe River. 7. Increased spawning temperature has a negative effect in the Murray River and Goulburn River and a positive effect in the Campaspe River. 8. Increased discharge over the previous five water years has a positive effect on habitat availability and, therefore, on carrying capacity.

Table B2. List of actions included in Murray cod scenarios

Table B2 provides descriptions of the management actions applied to each population. Population trajectories were simulated under every feasible combination of management actions in each system. Relative costs (per year and location) were included to demonstrate the calculation of benefit–cost ratios. All costs are illustrative only and may not reflect actual cost estimates for each action.

Action	Relevant populations	Description	Relative cost	Uncertainties
Stocking	Goulburn River Campaspe River	Stocking of approximately 100 000 fingerlings into the Goulburn River and 50 000 fingerlings into the Campaspe River	500	Limited data on system-specific survival of stocked individuals
Fishing regulations	Goulburn River Campaspe River Murray River	Complete closure of the recreational fishing season for Murray cod. Assumed baseline losses of approximately 10% of legal-size individuals each year (550–750 mm), with regulations reducing this number to zero	250	Numbers of individuals caught and removed in each river system not known with certainty Effectiveness of regulation uncertain Alternative scenarios not considered (e.g. alternating years of closed and open seasons)
Instream habitat rehabilitation	Goulburn River	Increase available instream habitat to estimated natural (historical) levels (Kitchingman et al. 2020), estimated to increase Murray cod carrying capacity by approximately 50%	1000	Uncertainties in the feasibility of increasing instream habitat to natural levels, and in the impacts of these increases on population carrying capacity
Environmental water	Goulburn River Campaspe River Murray River	Adjust discharge sequences to reflect years in which environmental water was delivered (approximately 2000–2020) and calculate counterfactual scenarios that reflect these same discharge sequences without environmental water	3000	Uncertainties in the availability of environmental water in each year, especially under reduced-runoff scenarios of climate change

Appendix C: Platypus case study

The population model used here was based on an existing model of platypus population dynamics (Bino et al. 2020) (Table C1). Bino et al.'s model includes stage-specific and density-dependent rates of survival and reproduction of juveniles and adults. We extended this model to include the effects of hydrological conditions, instream and riparian habitat condition, and predators on recruitment and survival. Specific actions and their effects on vital rates are described in Table C2.

This case study focuses on populations of platypus in four rivers in Victoria: the upper Campaspe River in central Victoria, the Mackenzie River in north-west Victoria, the Ovens River in north-east Victoria, and the Little Yarra River east of Melbourne. The four populations are representative of a range of conditions in which platypus populations occur, with rivers differing in size, average rainfall, and levels of regulation (see Section 2.1.3 above).

Table C1. Structure of platypus population model

Table C1 provides details of the population structure and the key parameter values for models of platypus population dynamics in each population. Specific parameterisations and R code for each model are available at https://github.com/aae-stats/aae.pop.templates.

Model component	Description	References
Matrix type	Lefkovitch matrix	Bino et al. (2020)
Number of classes	Juveniles Adults	
Sex ratio	Model included females only, with males assumed not to be limiting due to an approximately even (50:50) sex ratio	
Density dependence	Negative: Bottom-up density dependence based on a two-parameter Ricker model that reduced reproductive output when adult populations were large	
Carrying capacity	Carrying capacity, defined as the maximum number of breeding adult females supported by the available habitat; determines the strength of the negative density dependence. Parameter estimates were: Upper Campaspe River: 400 Mackenzie River: 200 Ovens River: 400 Little Yarra River: 400	Bino et al. (2015)
Environmental stochasticity	Unit-scaled Gaussian variation in survival estimates and Poisson variation in reproduction estimates	Todd and Ng (2001)
Demographic stochasticity	Poisson variation in number of surviving juveniles and adults	
Covariate effects	Flow conditions affected platypus population dynamics. All flow variables were defined based on water years (July–June), with the following effects: 1. Flows during spring, summer and winter were positively associated with survival of juveniles and adults. 2. Extreme high flows or unusually variable flows were negatively associated with survival of juveniles and adults. 3. Cease-to-flow events were negatively associated with survival of juveniles and adults.	

Table C2. List of actions included in platypus scenarios

Table C2 provides descriptions of the management actions applied to each population. Population trajectories were simulated under every feasible combination of management actions in each system. Relative costs (per year and location) were included to demonstrate the calculation of benefit–cost ratios. All costs are illustrative only and may not reflect actual cost estimates for each action.

Action	Relevant populations	Description	Relative cost	Uncertainties
Environmental water	All	Reduce the duration of cease-to-flow events by 80% through targeted water deliveries.	1000	Uncertainties in the availability of environmental water in each year, especially under reduced-runoff scenarios of climate change Included in models of all populations, despite some systems likely being unregulated
Permanent drought refuges	Upper Campaspe River Mackenzie River	Identify and maintain permanent drought refuges in the two most flow-stressed systems, eliminating cease-to-flow events in these systems.	800	Limited information on the feasibility of creating permanent drought refuges in flow-stressed systems
Temporary drought relief	All	Create short-term refuges through water deliveries or assisted dispersal in response to cease-to-flow events, reducing the duration of cease-to-flow events by 90%.	300	Limited information on the feasibility or effectiveness of short-term drought relief in all systems
Riparian revegetation	All	Improve the condition of riparian vegetation; estimated to increase platypus carrying capacity by 10% and to reduce natal and adult mortality due to predation by 5% and 40%, respectively.	1000	Uncertainties in the capacity for riparian revegetation in each system, and in the effects of this action on predation or habitat availability
Instream habitat rehabilitation	All	Increase available instream habitat to estimated natural (historical) levels; estimated to increase platypus carrying capacity by approximately 10% and reduce natal mortality due to predation by 40%.	800	Uncertainties in the capacity for instream habitat rehabilitation in each system, and in the effects of this action on predation or habitat availability
Barrier removal	All	Remove barriers to dispersal, increasing carrying capacity by 5% through increases in available habitat.	500	Effects of barrier removal do not consider increased connectivity of populations, due to

				limited knowledge of dispersal distances of adult females and the effects of barriers on platypus dispersal
Genetic mixing	Mackenzie River	Increase local genetic diversity in small, isolated populations; estimated to increase juvenile survival by 20%.	700	No data on the feasibility of genetic mixing or its consequences for vital rates of juveniles or adults
Translocations	Mackenzie River	Supplement small, isolated populations through annual translocations of two adult females from other sites.	300	Potentially unrealistic scenario, due to repeated translocations No data on the survival or subsequent reproduction of translocated individuals

Appendix D: Estuary perch case study

The population model used here was based on an existing model of estuary perch population dynamics (Stoessel et al. 2020) (Table D1). Stoessel et al.'s model includes age-specific and density-dependent rates of survival and reproduction of all life stages from eggs to a maximum age of 40 years and can accommodate the stocking and harvesting of individuals of any age. We extended this model to include the effects of hydrological conditions and instream habitat condition on recruitment and survival. Specific actions and their effects on vital rates are described in Table D2.

This case study focuses on a single estuary perch population in the Snowy River in eastern Victoria.

Table D1. Structure of estuary perch population model

Table D1 provides details of the population structure and the key parameter values for models of estuary perch population dynamics in each population. Specific parameterisations and R code for each model are available at https://github.com/aae-stats/aae.pop.templates.

Model component	Description	Reference
Matrix type	Leslie matrix	Stoessel et al. (2020)
Number of classes	Juveniles: eggs, larvae, young-of-year, 1–2 years old Adults: 3–40 years old	
Sex ratio	Model included females only, with males assumed not to be limiting due to an approximately even (50:50) sex ratio	
Density dependence	Negative: Bottom-up density dependence based on a two-parameter Ricker model that reduced reproductive output when adult populations were large	
Carrying capacity	Carrying capacity, defined as the maximum number of breeding adult females supported by the available habitat; determines the strength of the negative density dependence. Parameter estimates were: Snowy River: 30 000	Stoessel et al. (2020)
Environmental stochasticity	Unit-scaled Gaussian variation in the survival estimates and Poisson variation in the reproduction estimates	Todd and Ng (2001)
Demographic stochasticity	Poisson variation in the number of recruits and binomial variation in the number of surviving adults	Stoessel et al. (2020)
Covariate effects	Flow conditions from June to December affected estuary perch spawning and recruitment through a sequence of four steps: 1. One fifth of the spawning population is assumed to move to spawning habitat when daily flow exceeds 15 000 ML between June and November. 2. Individuals in spawning habitat are more likely to spawn early in the spawning period (August to December), so late movement cues are associated with reduced probabilities of spawning. 3. Extreme high flows following spawning reduce egg survival. 4. Larval survival is dependent on sufficient productivity, which was modelled as a positive function of daily flow.	Stoessel et al. (2020)

Table D2. List of actions included in estuary perch scenarios

Table D2 provides descriptions of management actions applied to the Snowy River population. Population trajectories were simulated under every feasible combination of management actions in each system. Relative costs (per year and location) were included to demonstrate the calculation of benefit–cost ratios. All costs are illustrative only and may not reflect actual cost estimates for each action.

Action	Relevant populations	Description	Relative cost	Uncertainties
Environmental water	Snowy River	Adjust discharge sequences to include specific water deliveries that provide migration cues and enhance post-spawning survival. Two levels considered: Low: two deliveries of 10000 ML/day on August 1 and September 1 in each year. High: three deliveries of 15000 ML/day on August 1, September 1, and October 1 in each year.	Low: 10 000 High: 20 000	Uncertainties in the availability of environmental water in each year, especially under reduced-runoff scenarios of climate change
Stocking	Snowy River	Stocking of approximately 10000 (low scenario) or 15000 (high scenario) fingerlings into the Snowy River.	Low: 2000 High: 3000	Limited data on system-specific survival of stocked individuals
Instream habitat rehabilitation	All	Increase available instream habitat to estimated natural (historical) levels, estimated to increase estuary perch carrying capacity by approximately 10%.	2500	Uncertainties in the capacity for instream habitat rehabilitation in the Snowy River and in the effects of this action on habitat availability

Appendix E: Yarra pygmy perch case study

The population model used here was based on an existing model of southern pygmy perch population dynamics (Todd et al. 2017) (Table E1). Todd et al.'s model was assumed to represent an upper bound on Yarra pygmy perch population dynamics under scenarios of increased genetic diversity. This model includes age-specific and density-dependent rates of survival and reproduction of all life stages from eggs to a maximum age of 4 years and can accommodate the stocking and removal of individuals of any age. We extended this model to include the effects of hydrological conditions, wetland habitat condition, introduced fish species, and population genetic diversity on recruitment and survival. Specific actions and their effects on vital rates are described in Table E2.

This case study focuses on populations of Yarra pygmy perch in two Victorian Rivers: Deep Creek in central Victoria and Armstrong Creek near Geelong.

Table E1. Structure of Yarra pygmy perch population model

Table E1 provides details of the population structure and the key parameter values for models of Yarra pygmy perch population dynamics in each population. Specific parameterisations and R code for each model are available at https://github.com/aae-stats/aae.pop.templates.

Model component	Description	Reference
Matrix type	Leslie matrix	Todd et al. (2017)
Number of classes	Juveniles: eggs, larvae, young-of-year Adults: 1–4 years old	
Sex ratio	Model included females only, with males assumed not to be limiting due to an approximately even (50:50) sex ratio	
Density dependence	Negative: Bottom-up density dependence based on a two-parameter Ricker model that reduced reproductive output when adult populations were large	
Carrying capacity	Carrying capacity, defined as the maximum number of breeding adult females supported by available habitat, determines the strength of the negative density dependence. Parameter estimates were: Deep Creek: 500 Armstrong Creek: 500	Todd et al. (2017)
Environmental stochasticity	Unit-scaled Gaussian variation in the survival estimates and Poisson variation in the reproduction estimates	Todd and Ng (2001)
Demographic stochasticity	Poisson variation in the number of recruits and binomial variation in the number of surviving adults	Todd et al. (2017)
Covariate effects	Covariates were simulated to capture the effects of drying events and introduced fish species: 1. Dry conditions occurred with a 3% probability in any year under wetter climates (post-1975, RCP8.5 low), increasing to a 10% probability per year in drier climates (post-1997, RCP8.5 high). When these events occurred, adult survival was reduced by 80%. 2. Introduced fish species were initially absent at both sites, with a 10% probability of introduction in any year. Once present, introduced fish species remained present unless removed through management intervention. Presences were common to all scenarios (including all climate change scenarios). Introduced fish species reduced the survival of all life stages by a minimum of 10%, with larger negative effects when habitat condition was poor (up to a maximum 55% reduction in survival).	

RCP8.5, Representative Concentration Pathway 8.5.

Table E2. List of actions included in Yarra pygmy perch scenarios

Table E2 provides descriptions of the management actions applied to each population. Population trajectories were simulated under every feasible combination of management actions in each system. Relative costs (per year and location) were included to demonstrate the calculation of benefit–cost ratios. All costs are illustrative only and may not reflect actual cost estimates for each action.

Action	Relevant populations	Description	Relative cost	Uncertainties
Wetland habitat rehabilitation	All	Increase condition of and connections to wetland habitat, estimated to increase Yarra pygmy perch carrying capacity by approximately 20% and reduce the negative effects of predators (described in Table E1, above).	2500	Uncertainties in the availability of suitable wetland habitat and in the capacity to restore and provide connections to this habitat
Drought refuges	All	Identify and protect persistent water sources that provide permanent water during drought conditions, particularly through restrictions on groundwater and surface-water extractions. Provision of drought refuges was assumed to eliminate drying events in all years (see Table E1, above).	10 000	Uncertainties in the effectiveness of restrictions on water extraction, especially under reduced-runoff scenarios of climate change
Introduced fish species detection and removal	All	Annual monitoring and removal of introduced fish species (e.g. redfin, gambusia), assumed to reverse their negative impacts (described in Table E1, above).	5000	Uncertainties in the absolute effects of introduced fish species on survival of Yarra pygmy perch
Genetic mixing	All	Increase local genetic diversity in small, isolated populations. Estimated to restore survival of all life stages to values observed in southern pygmy perch. In the absence of genetic mixing, survival of all life stages was decreased by 25% relative to that of southern pygmy perch.	5000	No data on the feasibility of genetic mixing or its consequences for vital rates of juveniles or adults Genetic mixing assumed to have equal effects in both populations, despite relatively lower genetic diversity in the Deep Creek population
Population establishment	All	Establish a new population connected to an existing population (directly or through translocations), assuming a complete absence of predators and availability of high-quality wetland habitat.	10 000	Limited data on locations with suitable habitat for establishment of Yarra pygmy perch populations Limited knowledge of the outcomes of Yarra pygmy perch stocking and translocations

Appendix F: Barred galaxias case study

The population model used here was developed from broad life history characteristics of barred galaxias and mountain galaxias *Galaxias olidus* (Table F1). The following characteristics of these two species were used to develop the population model:

- open-water and substratum egg scatterers with no parental care
- spawning events typically producing egg numbers in the 100s
- maximum recorded length of 150 mm, commonly 80–105 mm
- juveniles <50 mm in length
- maximum recorded age of 15 years in barred galaxias (4 years in mountain galaxias)
- spawns from August to October
- densities recorded at 0.11–0.50 fish per m^2, reduced to 0.003–0.027 fish per m^2 during early trout incursion.

These characteristics were used to estimate the rates of reproduction and early life survival (based on approximate egg numbers and the lack of parental care) and the survival and transition probabilities of each life stage (based on longevity and a relatively long juvenile phase). The population model included the stage-specific and density-dependent survival and reproduction rates of two juvenile and two adult life stages and can accommodate removals and translocations of individuals from all life stages. This model additionally included the effects of hydrological conditions, bushfires, introduced fish species, and population genetic diversity on recruitment and survival. Specific actions and their effects on vital rates are described in Table F2.

This case study focuses on populations of barred galaxias in four Victorian rivers: the Rubicon River on the slopes of Lake Mountain, the Howqua River on the slopes of Mt Stirling, the Delatite River north of Mt Buller, and Perkins Creek near Woods Point, north-east of Melbourne.

Table F1. Structure of the barred galaxias population model

Table F1 provides details of the population structure and the key parameter values for models of barred galaxias population dynamics in each population. Specific parameterisations and R code for each model are available at https://github.com/aae-stats/aae.pop.templates.

Model component	Description	Reference
Matrix type	Lefkovitch matrix	
Number of classes	Juveniles: two stages Adults: two stages	
Sex ratio	Model included females only, with males assumed not to be limiting due to an approximately even (50:50) sex ratio	
Density dependence	Negative: Bottom-up density dependence based on a two-parameter Ricker model that reduced reproductive output when adult populations were large	
Carrying capacity	Carrying capacity, defined as the maximum number of breeding adult females supported by the available habitat, determines the strength of the negative density dependence. Parameter estimates were: Rubicon River: 500 Howqua River: 500 Delatite River: 500 Perkins Creek: 500	
Environmental stochasticity	Unit-scaled Gaussian variation in survival estimates and Poisson variation in reproduction estimates	Todd and Ng (2001)
Demographic stochasticity	Poisson variation in the number of recruits and binomial variation in the number of surviving adults	
Covariate effects	Covariates were simulated to capture the effects of cease-to-flow events, bushfires, and introduced fish species: 1. Cease-to-flow events occurred with a 50% probability (75% in Perkins Creek) when discharge fell below the long-term (1970–2020) 5th percentile in a single year, or below the long-term 10th percentile for two consecutive years. Cease-to-flow events reduced early life and juvenile survival by 50–80% and reduced adult survival by 20–40% depending on the condition of the riparian vegetation. 2. Bushfires occurred with an initial probability of 1% (5% in Perkins Creek), with this probability doubling when discharge fell below the long-term 5th percentile in a single year, or below the long-term 10th percentile for two consecutive years, assuming no bushfires in the past 10 years. Bushfire events reduced early life and juvenile survival by 0–50% and reduced adult	

	survival by 0–25% depending on the condition of the riparian vegetation. 3. Introduced fish species were initially absent at both sites and had a 10% probability of introduction in any year. Once present, introduced fish species remained present unless removed through management intervention. Presences were common to all scenarios (including clsimate change scenarios). Introduced fish species reduced the survival of juveniles by 30% and reduced the survival of adults by 100%.

Table F2. List of actions included in barred galaxias scenarios

Table F2 provides descriptions of the management actions applied to each population. Population trajectories were simulated under every feasible combination of management actions in each system. Relative costs (per year and location) were included to demonstrate the calculation of benefit–cost ratios. All costs are illustrative only and may not reflect actual cost estimates for each action.

Action	Relevant populations	Description	Relative cost	Uncertainties
Introduced fish species detection and removal	All	Annually monitor and remove introduced fish species (trout); assumed to reverse their negative impacts (described in Table F1, above).	5000	Uncertainties in the effectiveness of introduced fish species removal
Genetic mixing	All	Increase local genetic diversity in small, isolated populations; estimated to increase survival of all life stages by 20%.	5000	No data on the feasibility of genetic mixing or its consequences for vital rates of juveniles or adults
Ex situ care	All	Protect 150 adult females in the event of a bushfire or cease-to-flow event. These individuals are assumed to be returned to the population without loss in the year following bushfire or cease-to-flow event.		Uncertainties in the number of fish protected in any year and their survival in aquarium conditions
Population establishment	All	Establish a new population connected to an existing population (directly or through translocations), assuming a complete absence of predators in the newly established site.	10 000	Limited data on locations with suitable habitat for establishment of barred galaxias populations Limited knowledge of the outcomes of barred galaxias stocking and translocations

Appendix G: Additional figures

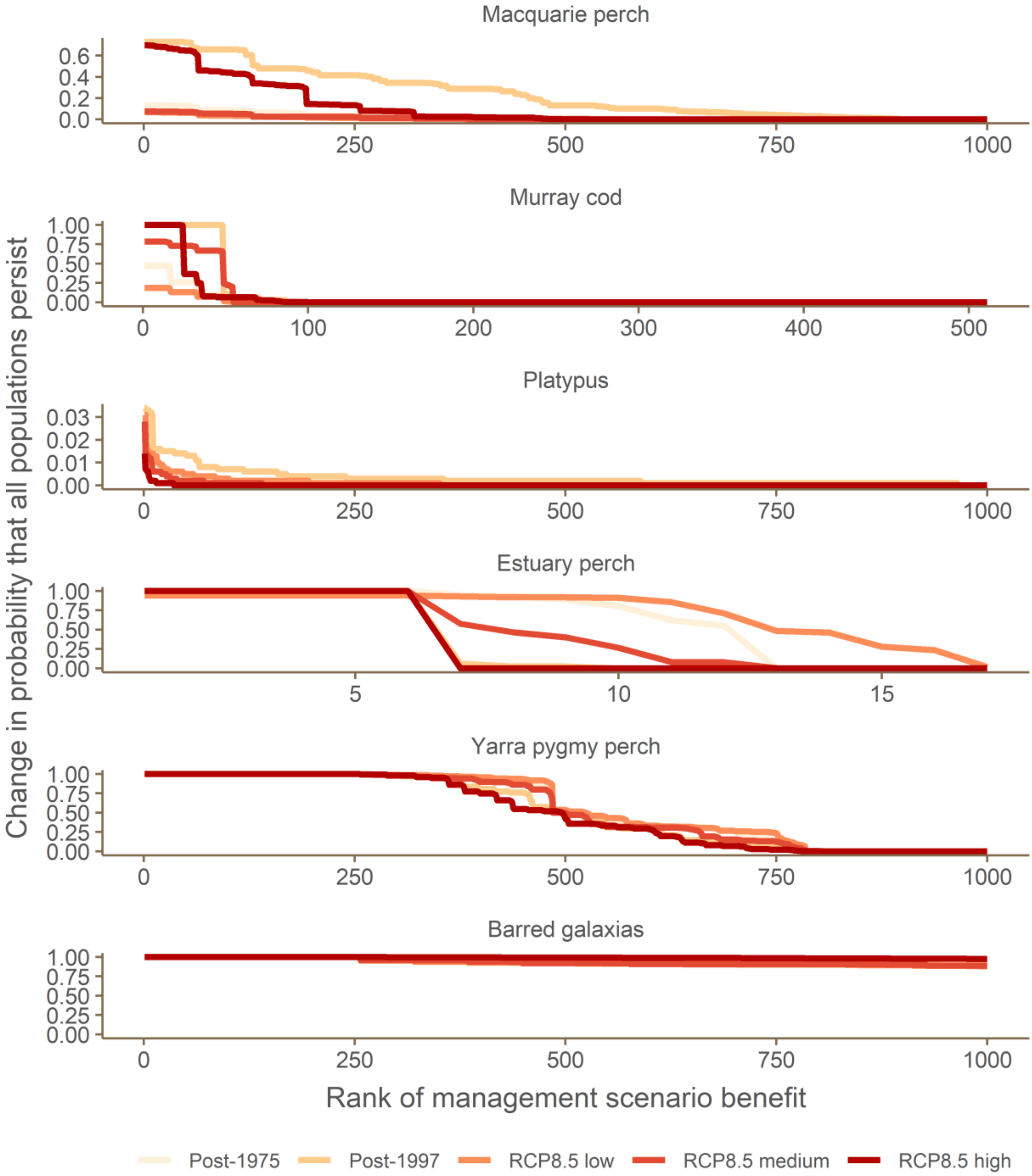

Figure G1. Changes in expected benefit based on the probability that all populations persist across the top-ranked combinations of management actions for each species. The *x*-axis indicates the rank of each management action scenario based on average persistence probabilities (from Figure 3), and the *y*-axis indicates the change in weighted effective population sizes for each management scenario relative to a do-nothing scenario. Management scenario ranks are ordered from most to least beneficial, so that lower rank numbers denote higher probabilities of population persistence. A maximum of 1000 scenarios is shown for each species.

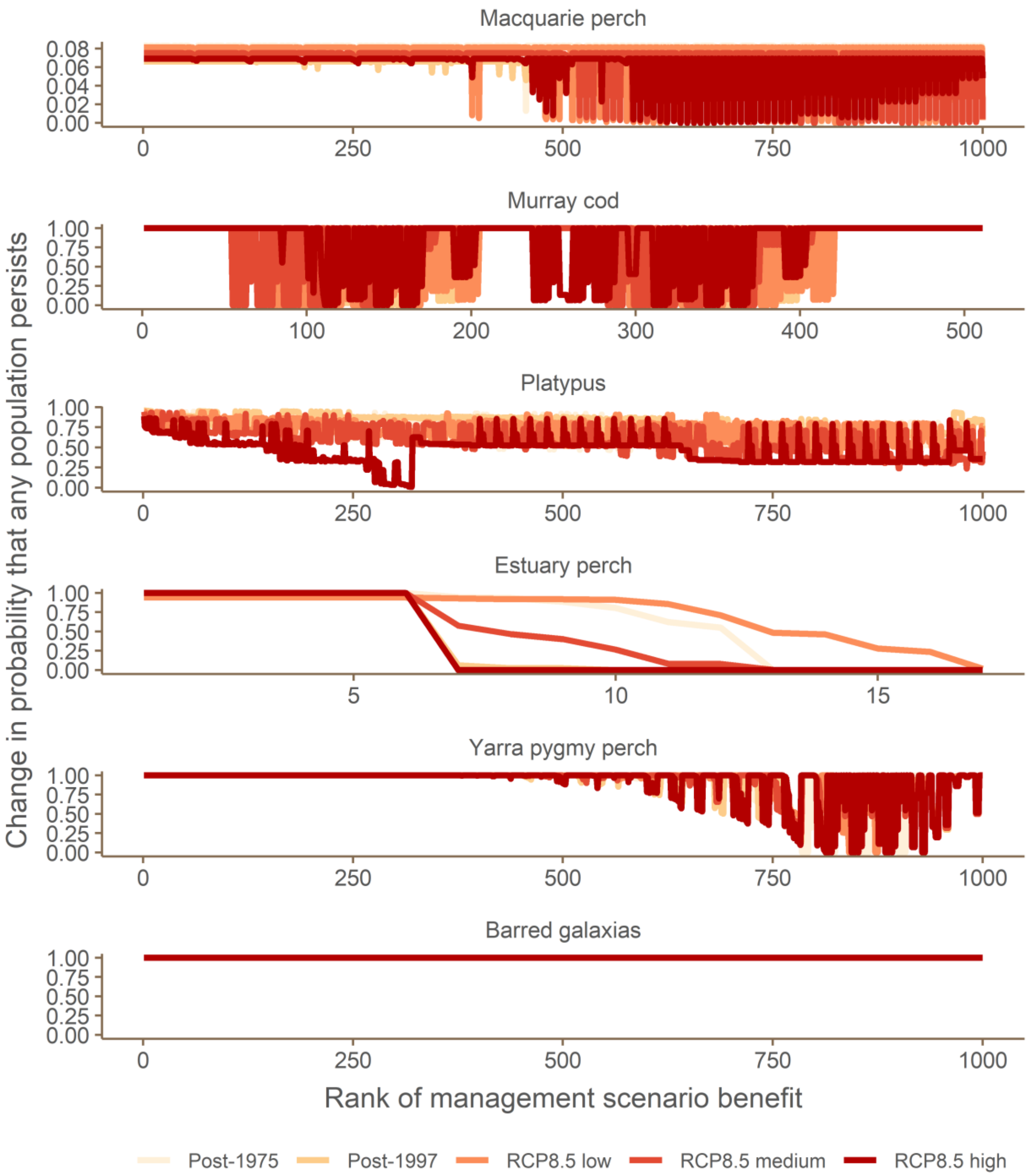

Figure G2. Changes in expected benefit based on the probability that any population persists across the top-ranked combinations of management actions for each species. The *x*-axis is the rank of each management scenario based on average persistence probabilities (from Figure 3), and the *y*-axis is the change in weighted effective population sizes for each management scenario relative to a do-nothing scenario. Scenario ranks are ordered from most to least beneficial, so that low-rank numbers denote higher probabilities of population persistence. A maximum of 1000 scenarios is shown for each species.

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