Priorities and Uncertainties of Predicted Impacts of Climate Change on Freshwater Biodiversity in New South Wales

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Cover image: Platypus (*Ornithorhynchus anatinus*) swimming. Credit: Sharon Wormleaton / OEH

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Executive Summary

Alongside a growing awareness that climate change represents a substantial threat to biodiversity in New South Wales, it has become increasingly evident that we cannot afford to wait until climatic shifts result in confirmed impacts to ecosystems. The scale of projected changes, and the significant implications these will have for the functioning of ecological communities mean we must act early to reduce the risk posed by climate change, in addition to multiple other processes driving biodiversity loss. Freshwater systems are challenging environments to manage for multiple stakeholders, and climate change will further exacerbate many existing conflicts or threats to biodiversity. This report provides guidance for land and water managers on conservation management may improve the long-term capacity of freshwater ecosystems to adapt and reduce biodiversity loss. A comprehensive assessment was made of the risk posed by climate change to the persistence of over 500 freshwater plants and animals in the basins of New South Wales. The report deals first with the projected vulnerability of those species to the impacts of climate change, and to what extent sources of uncertainty influences our assessment, and ultimately our choice of management priorities. The second part focuses in detailed approach to understanding how environmental management of non-climatic threats at local to regional scales could be best used to alleviate the impacts of climate change to fish species and communities.

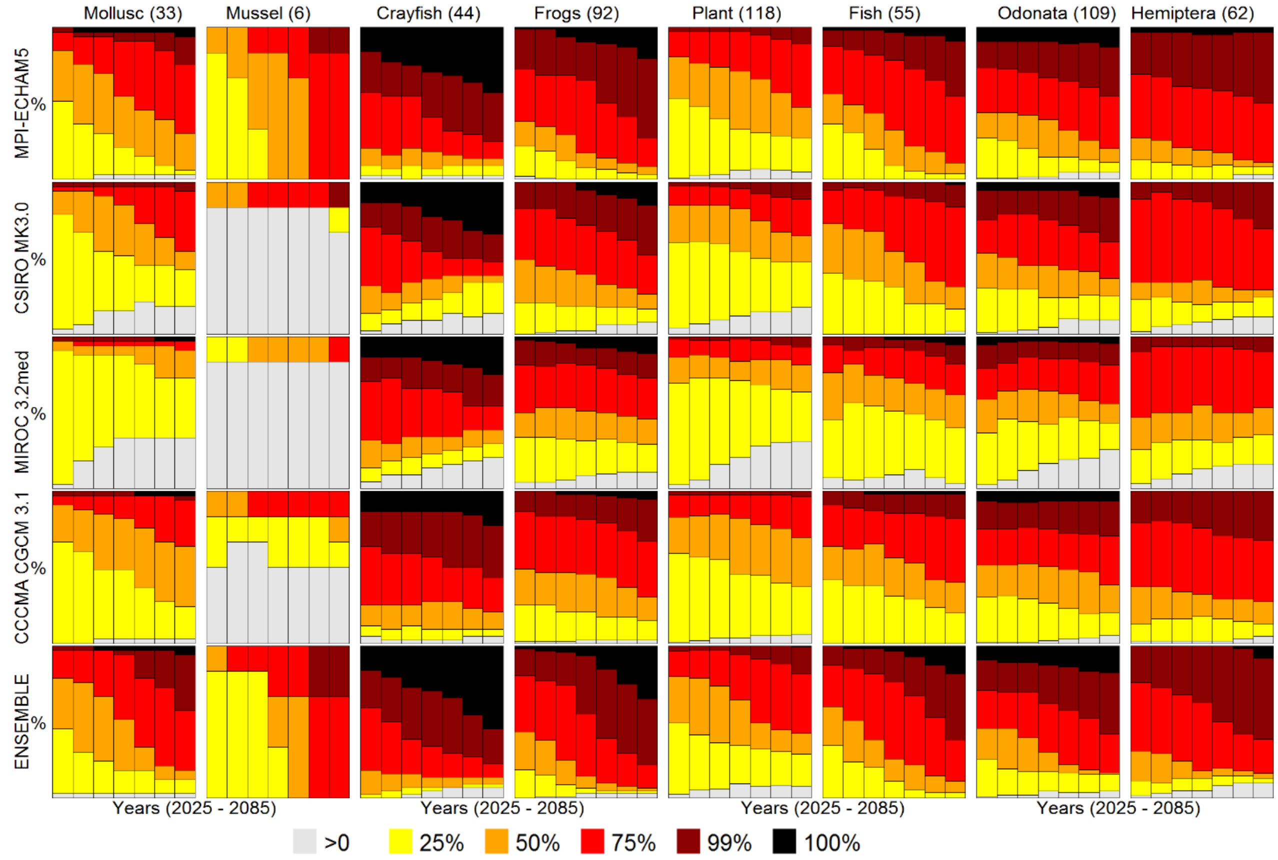

Figure 1. Proportion of species in eight taxa (number of taxa in brackets) within six categories of habitat loss (loss >0 indicate potential range expansion). Projections used the most probable dispersal rates for each group, under five climate change scenarios for 2085 (RCP 8.5).

Recommendations

- **Emissions reductions.**
  It is worth stating the highest priority remains a rapid reduction in our greenhouse gas emissions. The risk of significant biodiversity loss is greatly reduced under lower emissions scenarios, as well as the rate of change which allows time for adaptation measures to operate. Without emissions reductions the potential for unforeseen feedback mechanisms increases, and could as a result surpass our ability to mitigate impacts on biodiversity.
- **Climate change represents a serious threat to freshwater biodiversity in NSW**.
  As shown in Fig.1, by 2085 the majority of species modelled by this study were expected to decline to some extent. The severity of declines were clearly contingent on the type of climate models and emissions scenarios used in projections, but were also contingent on dispersal rate and mode. In particular, loss of environmentally suitable habitat would most heavily affect groups like the crayfish, frogs, Odonata and Hemiptera. For example, many species of spiny crayfish (*Euastacus* spp.) are endemic to NSW, restricted in their dispersal capacity, and in many projected scenarios, were severely threatened by extinction due to climate change.

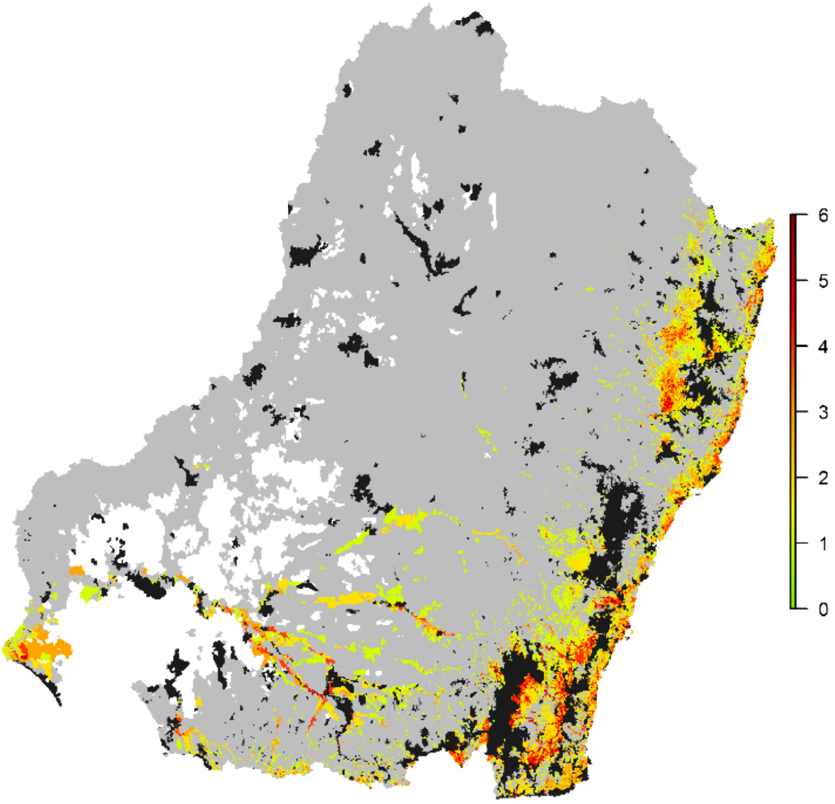

Figure 2. Overlap between conservation priorities based on projections for five GCM scenarios (RCP 8.5, 2085) with existing protected areas marked in black (same as Fig.4 in main text).

- **Protected areas will not ensure persistence.**
  As climates shift, representation of environmental conditions within reserves will

also change. Many environments at high altitudes are already protected, and could serve as refugia for other species, but a high proportion of the taxa most threatened by climate change are already restricted to headwaters, and isolated from other upland environments. Translocation may be an option, but has been a contentious subject, and will not be practical for all freshwater taxa. As such, the only option may be intervention to modify habitats and create refugia that will allow species to remain *in-situ* and resist change. In particular, novel measures may be required to mitigate climatic extremes (e.g. drought, heat waves) throughout the reserve network and in the right places may serve an entire community rather than just a single threatened species.

- **Expansion of the protected area system.**
  Under climate change, reserves will not remain safe havens for all species, but systematic additions to the reserve system could mitigate the degree of species loss from reserves. The project identified significant differences among priorities for different climate scenarios but also substantial areas of agreement (Fig. 2), and these represent low-risk options to begin planning conservation for freshwater under climate change. One of the most significant regional patterns to emerge was the importance of long stretches of the coastal fringe that face intense pressure from human development (Port Jackson, Manning, Hastings and Richmond basins). Many other conservation priorities were clustered around existing high-altitude protected areas, and would align well with OEH's involvement in the Great Eastern Ranges Initiative.
- **Planning beyond the protected area system**.
  The persistence of many, if not most species in the landscape during on-going climate change or extreme events, is likely to rest on their capacity to seek refuge in relatively small reserves or managed systems. An issue that distinguishes freshwater from terrestrial conservation is that the valuable characteristics of important sites (e.g. permanent flows, cold water and flushing floods) typically require extensive management or intervention at other locations, often beyond the boundaries of reserves. Therefore, a priority will be to secure good ecological condition of focal conservation areas, and alongside this provide the appropriate degree of connectivity within the landscape that allows species to move among core habitats, or recolonise from them.
- **Management effort may be wasted, or inefficient.**
  This project convincingly demonstrated that over the long-term declines in habitat suitability could greatly undermine the potential benefits management of non-climatic threats is expected to have. This outcome does not suggest we should not address the multitude of other threats faced by freshwater, rather we must use the best available knowledge to guide where there is the greatest potential to ensure persistence. The same considerations should be applied above when selecting focal habitats for conservation.
- **Low-risk options do exist.**
  Whilst emphasising caution above, there is also a need to begin taking actions where possible, and in many catchments, there appear to be low-risk options that are effective for fish under all climate scenarios. Even within the Murray-Darling basin, in which the worst-case scenarios are well below current projections, there

are different alternatives that have higher gains in the long-term. Furthermore, note that in both parts of the project conservation priorities included areas such as the Lower Murray, Coorong, and mid-Murray River that may require collaboration of OEH with other states authorities.

- **Make conservation representative of biodiversity, not surrogates.**
  For freshwater species as a whole, dispersal rates, climate models and emissions scenarios had relatively little impact on the broad distribution of conservation priorities. However, incomplete representation of taxonomic groups (or different dispersal modes) will influence conservation priorities and emphasises the need for planning to be more representative of biodiversity and have less reliance on surrogates that often are not effective proxies for deciding where and how to manage biodiversity. Likewise evaluating the relative benefits of any management action will be dependent on the taxa targeted.
- **Find common ground between the needs of freshwater and terrestrial systems.**
  Many of these recommendations could also be applied to terrestrial taxa e.g. vulnerability of high-altitude refugia, greater connectivity and new reserves to compensate losses under climatic shifts. Nonetheless, terrestrial management plans should more clearly identify how features critical to freshwater ecosystems will be preserved as these are often dependent on action beyond the freshwater habitats themselves. For example, expansion of riparian restoration schemes at strategic locations are known to serve multiple goals that improve resilience of reserves within the landscape.
- **Further data collection will support better decision making.**
  Possibly the greatest obstacle to improving the effectiveness and efficiency of management for freshwater biodiversity is the lack of survey data. The species modelled by this study indicate regions that are likely to provide refuge to a higher proportion of threatened species, but actions locally will require more information to determine how best this is achieved. Continued support for simple projects like cataloguing the locations of in-stream barriers are critical to a well-informed management strategy.

- For species threatened with extinction, these is no value in preparing management plans if funding is not provided to act on recommendations.
- Although still considered contentious, the risk of extinction for some species would suggest establishing multiple populations through translocation as a precaution is at the very least worth considering

Introduction

Efforts to control global greenhouse gas emissions continue to be delayed, further increasing the likelihood of substantial changes to global climate and threatening biodiversity at all levels of organisation (Woodward *et al.*, 2010). Freshwater ecosystems already face numerous threats from human activity, and are at particular risk because species typically have less capacity to adapt than those of terrestrial or marine systems (Ormerod, 2009). Although under lab testing the physiological tolerances of some macroinvertebrate orders indicate they have capacity to tolerate change (Barbara *et al.*, 2013), the combination of rising temperatures and altered flow regimes appear to already be affecting many freshwater systems globally (Mouthon & Daufresne, 2006; Daufresne & Boët, 2007; Chessman, 2009). Freshwater taxa have repeatedly been shown to be highly sensitive to minor changes in climatic conditions, suggesting that current ecological communities are unlikely to be resilient to significant changes *in-situ* (Bradley & Ormerod, 2001; Durance & Ormerod, 2007). Furthermore, many species may be inherently limited in their ability to track suitable conditions due to their mode of dispersal in relation to the structure of riverine habitats (Hughes *et al.*, 2009; Kappes & Haase, 2012). The majority of research into managing the impacts of climate change on biodiversity has focused on identifying priorities for restoration and protection in terrestrial habitats (Heller & Zavaleta, 2009; Hughes, 2011). However, there has been an increasing awareness that the structure and ecology of freshwater systems must be considered explicitly, alongside those of terrestrial and even marine systems, to avoid more significant losses of biodiversity (Adams *et al.*, 2013; Bond *et al.*, 2014).

Sources of Uncertainty Forecasting Vulnerability of Freshwater Species

Species distribution models (SDMs) are one of the most important methods being used to assess the potential impacts of climate change on species and to guide conservation planning (Wiens *et al.*, 2009), with an increasing number of applications to freshwater streams (Domisch *et al.*, 2015; Kuemmerlen *et al.*, 2015). In this approach, information on species occurrence, together with climate data, to build models that can project the environmental suitability of a location, and how that suitability might change as the climate shifts. However, a range of factors influence the output of SDMs which then have knock-on consequences for decision making with regard to conservation planning. Variation in SDMs can stem from choices regarding species occurrence data (Stankowski & Parker, 2011), modelling algorithm (Elith *et al.*, 2006), study spatial extent (VanDerWal *et al.*, 2009a; Barbet-Massin *et al.*, 2012), environmental variables used (Porfirio *et al.*, 2014), presence/absence threshold (Liu *et al.*, 2005), model complexity (Warren & Seifert, 2010), and for future climate data, the emission scenario and global climate model used (Buisson *et al.*, 2010). As the outcomes of any particular study can be highly context-specific (region, taxa, data quality), no universal agreement exists on an optimal modelling strategy, and users are strongly advised to be mindful of uncertainty. Without being quantified, multiple sources of potential error can contribute to a sense of mounting uncertainty in projections that poses challenges for decision-making. Thus it is important to understand which factors are most influential in affecting results, so we can assess what potential management actions are least at risk of failure (Kujala *et al.*, 2013; Wright *et al.*, 2015).

There is currently a relatively poor understanding of dispersal capacity for many species: information is either not available or is of poor quality, limiting the effectiveness of biodiversity management (Driscoll *et al.*, 2014). Dispersal capacity is key to species' potential for tracking suitable environmental conditions and this study therefore investigated how the rate and type of dispersal of different taxa affect their projected responses under climate change, and to what extent this uncertainty could alter conservation priorities. The study was aimed at providing an assessment of the future vulnerability of freshwater species within New South Wales (NSW), but also included species from catchments in which NSW has a management role such as the Snowy and Genoa basins, and Murray-Darling basin. Recent studies have shown freshwater communities in south-eastern Australia are responding rapidly to climatic changes (Chessman, 2009; Chessman, 2013; Mac Nally *et al.*, 2014) and that many species are likely to be at risk in the future (James *et al.*, 2013). There has also been rapid growth in the number of studies aimed at modelling the future distribution of freshwater taxa globally (e.g. Buisson & Grenouillet, 2010; Domisch *et al.*, 2012; Domisch *et al.*, 2015). Several studies have also been performed for Australian taxa, including fish (Bond *et al.*, 2011; James *et al.*, 2013), crayfish (James *et al.*, 2013) and Odonata (Bush *et al.*, 2014a). Here we assess a diverse set of taxonomic groups including fish, plants, molluscs, frogs, crayfish, Hemiptera (aquatic bugs), Odonata (dragonflies and damselflies) and the platypus, that vary in ecology and dispersal capacity.

The Importance of Dispersal under Climate Change

This study estimates the vulnerability of freshwater species in south-eastern Australia under climate change, with a particular emphasis on the sensitivity of those predictions to the dispersal capacity of individual species. Movement of species' and populations are the result of complex and, in many cases, difficult to observe interactions between individuals and their environment. Straight-line measurements of distance (i.e. Euclidean) are often used for convenience but this may not reflect species' habitat requirements and their preferred dispersal pathways (Grant *et al.*, 2010) and this is especially true of structured landscapes such as river networks (Sutherland *et al.*, 2014). For species that cannot tolerate changes in their environment *in-situ*, their capacity to move and track suitable climatic conditions or reach refugia will be key to their survival (Hannah, 2010). Therefore vulnerability to extinction rapidly increases when ecological or topographic constraints reduce the potential for a species to track suitable conditions (e.g. Robson *et al.*, 2011; Reside *et al.*, 2012). As a result, freshwater taxa typically face greater adaptation challenges than terrestrial taxa, not only because dispersal along river systems may not follow climatic shifts, but also because suitable habitats are often fragmented within river basins.

In NSW there is particular cause for concern regarding the adaptive response of freshwater species to climate change because the coastal rivers flowing eastwards typically do not encompass a large latitudinal range and therefore restrict movement that may otherwise track suitable climatic conditions (Turak *et al.*, 2011a). Ranges may also be constrained within catchments by obstacles such as waterfalls or dams (Pittock & Finlayson, 2011) and altitudinal gradients may simply lead species into climatic *cul-de-sacs* (Domisch *et al.*, 2011). Whilst dispersal capacity is likely to be a critical factor in determining vulnerability to climate change, it is not clear whether vulnerability among freshwater taxa is primarily dictated by their opportunities for dispersal, or due to differences in their exposure and sensitivity to climatic change (Foden *et al.*, 2013; Bush *et al.*, 2014a). In Europe, some of the most rapid observed shifts in response to climate change have been among freshwater taxa such as fish and insects, enabled by the relatively high connectivity of habitats across the landscape provided by canals (Hickling *et al.*, 2006; Comte *et al.*, 2012). Assuming other freshwater taxa have similar sensitivities to climate change, then the context of a species range within the landscape, such as its connection to headwaters or the potential to move south within the same basin, will be likely to play an important role in its ability to offset losses within their current range (Swab *et al.*, 2012). Differences between the life-histories of groups of freshwater taxa affect the chance an individual has to disperse and colonise new suitable habitats, and thus define the landscape in which taxa must move or adapt. Water-dependent taxa like fish may be able to travel large distances, but are restricted to the stream channel, whereas crawling taxa such as crayfish and frogs may have the ability to colonise new river basins, but are less likely to move large distances. If freshwater taxa with higher dispersal capacity are at lower risk on average, management actions can focus on refugia suited to the dispersal-limited taxa like fish and molluscs (Davis *et al.*, 2013; Robson *et al.*, 2013) whilst promoting less intensive improvements to landscape connectivity for others (Mackey *et al.*, 2010; Bush *et al.*, 2014g).

In addition to implications of dispersal capacity for individual species, a number of studies have shown that despite modelling uncertainty, in order for reserves to remain effective and representative in the future, it is important to include the effects of climate change in reserve selection models and select new reserves that consider the potential for distributions to change (e.g. (Araújo *et al.*, 2004; Hannah *et al.*, 2007; Carvalho *et al.*, 2011). Given our limited understanding of range shifts, and the multiple sources of uncertainty in predicting species distributions in the future, to what extent does dispersal capacity influence a species’ vulnerability under climate change, and what are the consequences of uncertainty for planning management actions spatially?

Mitigating Climate Change Impacts by Managing Alternative Stressors

Freshwater ecosystems are already under serious pressure from a multitude of threats, in addition to climate change (Vörösmarty *et al.*, 2010). The state of freshwater biodiversity has been in decline both nationally and globally (Stendera *et al.*, 2012). These trends stem from large scale alteration of hydrological systems that has affected almost every river in Australia (Stein *et al.*, 2002). In addition to habitat loss and fragmentation, many fish are prone to the loss of seasonal flows and subsequent decline in wetlands, artificial barriers that block movement, riparian degradation, pollution, and introduced species (e.g. Pusey & Arthington, 2003; Rowe *et al.*, 2008). These stressors threaten freshwater biodiversity globally, and within Australia are common factors threatening many species (Pusey *et al.*, 2004; Dudgeon *et al.*, 2006).

Climate change is predicted to place further pressure on freshwater ecosystems and exacerbate other threatening processes (e.g. water pollution and eutrophication: Hering *et al.*, 2010), although the synergistic impacts of multiple threats is poorly understood (Ormerod *et al.*, 2010). Given we are already well aware of numerous threats facing freshwater fish in NSW, the projections of climate change scenarios cannot be considered in isolation. Furthermore, many gaps remain in our understanding of the ecology of freshwater species, particularly among non-commercial species. Threats are most often inferred because declines may only be recognised over long periods or across disparate monitoring programs that make attributing cause difficult (Jackson & Füreder, 2006; Davies *et al.*, 2010). There is growing evidence that the decades of investment to improve environmental health and water quality in Europe have partially offset expected changes due to climate (Durance & Ormerod, 2009), and therefore maintaining those standards could reduce the stress on freshwater ecosystems from climate change (Palmer *et al.*, 2008; Palmer *et al.*, 2009).

Whilst the threat of climate change is increasingly well recognised, it can be difficult to directly link local management actions to mitigation of, and adaptation to, a threat felt across the landscape. A common balance between the growing awareness of climate change and the often pressing need to act is to continue promoting management of other threatening processes in the short-term, so that once the effects of climate change begin to be felt, natural systems may be more resilient and have greater capacity for autonomous adaptation. Clearly the trajectory of many populations suggests that without intervention, they face local or global extinction *before* climate change becomes a critical issue (e.g. Hunter *et al.*, 2009; Hardie, 2013). For example, although not originally intended to counteract climate change effects, in Europe the widespread improvements to water quality though the Habitats Directive have led to a significant improvements in aquatic biodiversity, and run counter to the observed trends in communities elsewhere (Floury *et al.*, 2012; Vaughan & Ormerod, 2012).

A number of modelling studies have shown that climate change is likely to affect the distribution of freshwater fish and alter the composition of natural communities (Buisson & Grenouillet, 2010; Comte *et al.*, 2012; Barmuta *et al.*, 2013). Many studies analysing management adaptation options have responded to concerns that rising stream temperatures will have negative impacts on commercially important salmonids (Beechie *et al.*, 2013; Katz *et al.*, 2013), and in some cases this is coupled with threats from introduced species (Wenger *et al.*, 2011; Lawrence *et al.*, 2013).

These studies typically show that management can promote species’ persistence within watersheds, although this may require a high degree of investment (Wade *et al.*, 2013). Despite general agreement that management of non-climatic stressors will improve the persistence of threatened species under climate change, there has been relatively little done to consider where management actions would be best placed given the long-term outlook under climate change. The combined impact of additional threatening processes has received more focus in terrestrial systems, such as habitat loss due to land use change (e.g. Ponce-Reyes *et al.*, 2013), or changing fire regimes (Swab *et al.*, 2012).

This study demonstrates the relative benefits of catchment management actions to alleviate the impact of three major threats affecting freshwater fish in NSW: human-made barriers to movement (dams and weirs), landscape modification and introduced species. Based on our understanding of species sensitivities to non-climatic stressors the aims of this project were to answer: 1) which species are likely to be most affected by non-climatic stressors, and are these threats expected to remain as influential under climate change?, 2), where are management actions to reduce or remove these threats from sub-catchments likely to be most effective?, 3) are those benefits likely to be sustained under climate change?, and 4) do management priorities differ when we consider only climate-sensitive or diadromous species, rather than all native fish?

PART ONE – Impact of Dispersal Uncertainty on the Vulnerability of Freshwater Biodiversity under Climate Change

There are many potential sources of uncertainty that should be accounted for when estimating vulnerability under climate change using species distribution models. Previous studies have shown that factors such as the choice of global climate model (GCM), and emissions scenario (hereafter referred to as RCPs (relative concentration pathways)(van Vuuren *et al.*, 2011)), all significantly affect the predicted impacts of climate change on species (Buisson *et al.*, 2010; Porfirio *et al.*, 2014). Consequently, to assess the importance of dispersal capacity to individual species vulnerability or conservation prioritisation, each species was projected across all combinations of factors that described the major sources of uncertainty.

Methods

Species Data

Species' occurrence records were collated from a wide range of sources including state and museum collections, government survey records, local catchment authorities, scientific literature and private collectors (Appendix 1). The final assessment included models of the platypus (*Ornithorhynchus anatinus*), 109 Odonata, 62 Hemiptera, 92 frogs, 44 crayfish, 57 fish, 118 aquatic or semi-aquatic plants, six freshwater mussels and 39 freshwater molluscs. Mussels were treated separately to the other molluscs because of the very important role fish play in their life history (Schwalb *et al.*, 2011), which influences both their distribution and their future dispersal pathways. The choice of taxonomic groups was primarily driven by the goal to represent a diverse array of life histories, but was inevitably also influenced by data availability. Whilst in most groups it was possible to model a large proportion of the species known from the region, the focus for groups such as plants and molluscs was to include species with a wide variety of distributions and range sizes such that the influence of climate scenario and dispersal rates could be estimated independent of these effects. Attempts were made to include several other macro-invertebrate orders, but unfortunately many institutions have not entered their collections into electronic format, which often led to a severe spatial bias in the collection and entry of records. Although the entire breadth of freshwater biodiversity in NSW is not included, the species included do represent a broad range of taxa that vary in their range extent, environmental sensitivity and dispersal ability. These differences have both implications for how species distributions are best modelled, and how they are likely to respond under climate change.

Environmental Data

Climate change projections were based on Representative Concentration Pathways (RCPs), which are the standardised warming trajectories developed for the

Intergovernmental Panel on Climate Change's Fifth Assessment Report in 2013 (Moss *et al.*, 2010; van Vuuren *et al.*, 2011). The RCPs used in this study describe a range of stabilisation, mitigation and non-mitigation pathways that under medium or high emissions scenarios result in radiative forcing reaching 6 and 8.5 W/m2 respectively by 2100, equivalent to global average temperatures increasing 3.0 and 4.9°C respectively (Rogelj *et al.*, 2012). Inevitably, the choice of global climate model (GCM) plays a significant role in the outcome of climate change assessments. Research has shown that climate ensembles can perform better than single GCMs in simulating observed conditions (Fordham *et al.*, 2011; Fordham *et al.*, 2012), and multiple scenarios are useful to span the range of uncertainty in predicting future climates (Buisson *et al.*, 2010). Within NSW, climate projections from the NARCliM project are likely to become instrumental to planning management in the study region (Evans *et al.*, 2014). Based on a review of all GCMs, including their predictive performance in south-east Australia, their independence from one another (Evans & Ji, 2012), and availability of the data, the four GCMs chosen for the NARCliM projections were MIROC 3.2 (med), MPI-ECHAM 5, CCCMA CGCM 3.1 and CSIRO MK3.0. To provide output relevant to these anticipated projections being released, the species models were forecast for each of these four GCMs, and a fifth climate projection was added based on a seven-GCM ensemble of models that performed well for the region (Fordham *et al.*, 2011). The ensemble GCM scenario, hereafter referred to as "Ensemble", was averaged equally across projections from MRI, HadCM3, MIROC 3.2 (med), CSIRO MK3.0, HadGEM, GFDL2.0, and ECHAM5, and although it included two of the NARCliM GCMs tested separately, patterns of warming and precipitation change were not correlated. Lower emissions scenarios were omitted from this study as all indications suggest achieving the necessary reductions are unlikely (Peters *et al.*, 2013). Therefore, for each GCM only medium and high emissions scenarios were considered (RCP6 and RCP8.5 respectively).

Climate data for the selected GCMs were originally provided by the Tyndall Centre at the University of East Anglia, UK (available at http://climascope.wwfus.org) and downscaled as part of the National Climate Change Adaptation Research Flagship (James *et al.*, 2013). Rather than using gridded data, models were based on the stream network from the National Catchment and Stream Environment Database V.1.1.3, part of the Australian Hydrological Geospatial Fabric (GEOFABRIC, 2013). Organising the modelling environment and predictor variables to reflect the structure of a freshwater system is important when predicting habitat suitability because it can influence the accuracy of freshwater SDMs without necessarily affecting performance metrics (Domisch *et al.*, 2013). James *et al.* (2013) aggregated climate data to the same watersheds to generate broad hydrological parameters using a bucket model outlined by (Donohue *et al.*, 2012). Local differences in precipitation can be poor proxies for changes to runoff (Chiew & McMahon, 2002), and hydrological forecasts can therefore greatly improve projections of habitat suitability for freshwater species.

Connectivity

Given that shifts in the distributions of freshwater species are highly dependent on species' ability to disperse and track suitable climatic conditions, assessing response and vulnerability to climate change relies on realistic incorporation of movement patterns. Dispersal constraints were used to select appropriate background inputs to the species' distribution models (SDM), limit the extrapolation of baseline predictions, and then varied to assess how different dispersal rates and modes affected the outcome of the vulnerability assessment.

An understanding of dispersal can be very useful when selecting appropriate background data for SDMs because we wish to represent the range of environments historically accessible to that species (Barbet-Massin *et al.*, 2012). Distant sites beyond the dispersal reach of the target species are typically more environmentally dissimilar and may lead to model over-fitting and a reduction in performance at the scales relevant to the species (VanDerWal *et al.*, 2009a). Connectivity was measured on the basis of the shortest distance between each stream segment and those where the target species had been observed. In those groups where recording effort has been relatively high (fish, frogs and plants), connections were drawn from only the observations made in the last 30 years. Three modes of dispersal were considered to describe broad differences in movement: swimming for water-dependent taxa (fish, mussels and molluscs), mixed-movement for taxa with the capacity to disperse overland (platypus, crayfish, frogs), and direct dispersal for taxa with passive or aerial dispersal (plants and insects).

Dispersal Mode

For freshwater-dependent taxa, distances between nodes of the stream hierarchy were calculated in R using the network analysis package *igraph* (Csardi & Nepusz, 2006). Sub-catchments in other neighbouring basins were therefore disconnected unless the species was considered to have potential to tolerate marine environments such as those species with diadromous life histories (Miles *et al.*, 2013). Although natal-homing can still result in populations becoming structured by distance from one another (Schmidt *et al.*, 2014), marine-tolerant species were allowed the opportunity to move to new river basins via connections to neighbouring catchments, based on distances from the river mouth, up to 200km away. Without specific information on the costs of dispersal for freshwater fish in the sea, which can be either passive or directed, the connections to new basins were made without any penalty for distance from occupied basins. This distance was used because it is the minimum distance populations of some species would need to make to cross the Bass Strait.

For the stream-dependent taxa, up and downstream movements were considered equally likely. However, natural barriers such as waterfalls can play a key role in the understanding of distribution, and movement upstream was prevented at these points (1312 listed in eastern Australia; GEOFABRIC, 2013). For the purposes of this study, artificial barriers were not included because the data available did not cover the majority of eastern Australia, but their impact was addressed in depth in Part Two of this report. Consequently, all sub-catchments upstream of waterfalls from occurrence records were considered disconnected. An exception to this constraint was made for five species of fish (*Anguilla australis* and *A.reinhardtii, Galaxias brevipinnis, Gobiomorphus coxii* and *Mordacia mordax*) that are able to navigate around or over river barriers.

Movement for groups of taxa with semi-terrestrial dispersal options was primarily directed along river system corridors among sub-catchments, but connections were added to stream hierarchies between headwater sub-catchments so that movement could occur across catchment divides. Connections from a sub-catchment were made to the nearest cross-headwater sub-catchment not immediately connected downstream. The “cost” of crossing these additional connections was set 25 times higher than the real distance to emphasise the low likelihood associated with inter-basin movement, without excluding the possibility. Multiple headwater crossings were possible below a given dispersal threshold limit was reached. This approach allowed movement between separate river basins, but also shorter pathways between headwaters of catchments that are only otherwise connected far downstream. For species with active flight (insects) or passive dispersal (plants), we simply considered the straight line distance between two sub-catchments.

Current Understanding of Dispersal Rates

Whilst dispersal is a key process responsible for structuring freshwater communities (Heino, 2013), it has been difficult to quantify the movement of any group. Molluscs are best known for being slow dispersers, and the majority may have extremely restricted dispersal capacity (Ponder & Colgan, 2002). However, molluscs do disperse, and some snails and mussels are well known invasive species (Kappes & Haase, 2012). For example, a number of families display upstream migrations (~ 1km) thought to compensate for downstream drift, and in the lab *Potamopyrgus antipodarum* can move 15 cm upstream in 30 min, equivalent to a maximum of 2.6 km yr^{-1} (Haynes *et al.*, 1985). Some molluscs may also be ingested by fish and transported short distances unharmed, and there are many examples of passive lateral dispersal, both by biotic vectors such as insects, birds, mammals (including humans), and by abiotic vectors such as tornadoes (Kappes & Haase, 2012). Passive dispersal may play an even greater role in the dispersal of aquatic plants, with waterbirds in particular regularly transporting seeds (Brochet *et al.*, 2009). In fact, birds may be responsible for transporting a large variety of taxa (Green *et al.*, 2002), including juvenile crayfish (Águas *et al.*, 2014) and insects that lack aerial adults (Laux & Kölsch, 2014). Much of our knowledge regarding passive movements is anecdotal and so its regularity is unknown, but it may be that to track suitable climate conditions, the dispersal of less-mobile taxa will depend on the prevalence of rare, long-distance “jumps”.

By contrast, most aquatic insects have the ability to actively fly along the stream corridor and overland (Didham *et al.*, 2012). A study by Macneale *et al.* (2005) is a rare example in which dispersal was recorded directly by flooding a stream with a chemical isotope and then trapping adult stoneflies in the vicinity. The maximum dispersal distance was under 1 km, but females were also caught at least 0.5 km into the forest, showing that cross-catchment dispersal does occur. Mosquitoes may move between 1 and 5 km (Service, 1997), and the population genetics of the montane caddisfly *Drusus discolour* suggests that dispersal effects only become evident at distances >20 km (Geismar *et al.*, 2015). Jaeschke *et al.* (2012) estimated that allowing for larval development, six European Odonata could disperse between 0.5 and 14 km per year in response to climate change. In fact, some of the most rapid northward range shifts observed in response to climate change in the UK have been among the aquatic Hemiptera (~4 km y^{-1}) and Odonata (~6.8 km y^{-1}) (Hickling *et al.*, 2005; Hickling *et al.*, 2006). Even higher rates are possible. Over 11 years, *Anax imperator* expanded its northern range margin at an average rate of 88 km y-1 (Flenner & Sahlén, 2008). Nonetheless, whilst these observations lend support to our assumption that insects have greater capacity for dispersal than other aquatic taxa, these species may simply be the most visible examples, and do not necessarily represent dispersal of all Odonata, let alone other aquatic insects.

Dispersal rates are highly variable between frog species (Driscoll, 1997; Smith & Green, 2006; Sinsch, 2014), and the role of landscape resistance in explaining this variation remains unclear (Stevens *et al.*, 2006). Movement detected in tracking or mark–recapture studies of frogs are usually far below the corresponding estimates based on molecular gene-flow data which should indicate long-term averages including rare dispersal events (Bohonak & Jenkins, 2003). For example, the European tree frog covers distances ranging between 0.3 and 4.0 km yr^{-1} (Angelone & Holderegger, 2009), whereas natterjack toads may move 12 km between breeding ponds (Sinsch *et al.*, 2012). *Litoria aurea*, one of the species modelled in this study, is known to have moved 11 km in the course of a year during a recent “boom” in its population cycle (Daly, 2014). Rates of invasion by several introduced toad species also show there is potential for frogs to disperse faster still (Lobos & Jaksic, 2005; Phillips *et al.*, 2010).

Crayfish are to known to move within the basin along river channels during their lifetime, and on rare occasions are found moving overland between streams, but otherwise their cryptic nature makes observing movement very difficult (McCormack, 2012). Thus the majority of our knowledge comes from genetic studies that show populations of *Euastacus* in particular, are highly structured (e.g. Hughes, 2007; Whiterod *et al.*, 2014). Movement of platypus is also rarely recorded, but genetic information indicates their populations display an isolation-by-distance structure (Kolomyjec *et al.*, 2009). Movement of females can be limited to quite short distances but inter-basin migration has also been recorded (~7-8 km; Furlan *et al.*, 2013).

Finally, the movement of fish has perhaps received the greatest attention, particularly in south-east Australia where a large proportion of species have diadromous life-histories (Miles *et al.*, 2013). Some species like *Macquaria ambigua* have been recorded migrating huge distances (>2000 km), but importantly, migration events do not necessarily occur annually; and the majority of tagged individuals moved much shorter distances or not at all (Reynolds, 1983). Yet other fish species do not make any long distance movements, e.g. *Gadopsis* spp. and *Tandanus tandanus* appear to be highly sedentary (Koster & Crook, 2008; Broadhurst *et al.*, 2011; Hammer *et al.*, 2014; Koster *et al.*, 2014). A number of fish have been shown to have significant genetic population structure, both among and within drainage basins e.g. *Craterocephalus stercusmuscarum* and *Pseudomugil signifier* (McGlashan & Hughes, 2000; McGlashan *et al.*, 2001), and yet others show less structure than we might expect e.g. *Hypseleotris compressa* (McGlashan & Hughes, 2001).

Consideration of Dispersal Uncertainty

Unfortunately, whilst the sources above show research on dispersal of freshwater taxa is an area of active interest and have been able to employ an expanding range of methods (e.g. Broadhurst *et al.*, 2011; Hobson *et al.*, 2012; Crook *et al.*, 2013), quantitative estimates of species' ability to track shifting climates are lacking in the majority of cases, particularly for Australian species. To manage this uncertainty for particular species, six functions were created to describe the probability a species within that group would be able to move between 0 and 12 km yr^{-1} (Fig.1). Suitability scores were relatively stable to changes in these functions when used to weight dispersal constraints, except among molluscs for whom small increases in dispersal rate were proportionally larger. Nonetheless, the exponential decline in dispersal probability used was considered a reasonable a reflection of dispersal limitation in that group. Note the functions assume dispersal rate is not equivalent among taxa and describe our uncertainty in the dispersal capacity of a species within each group.

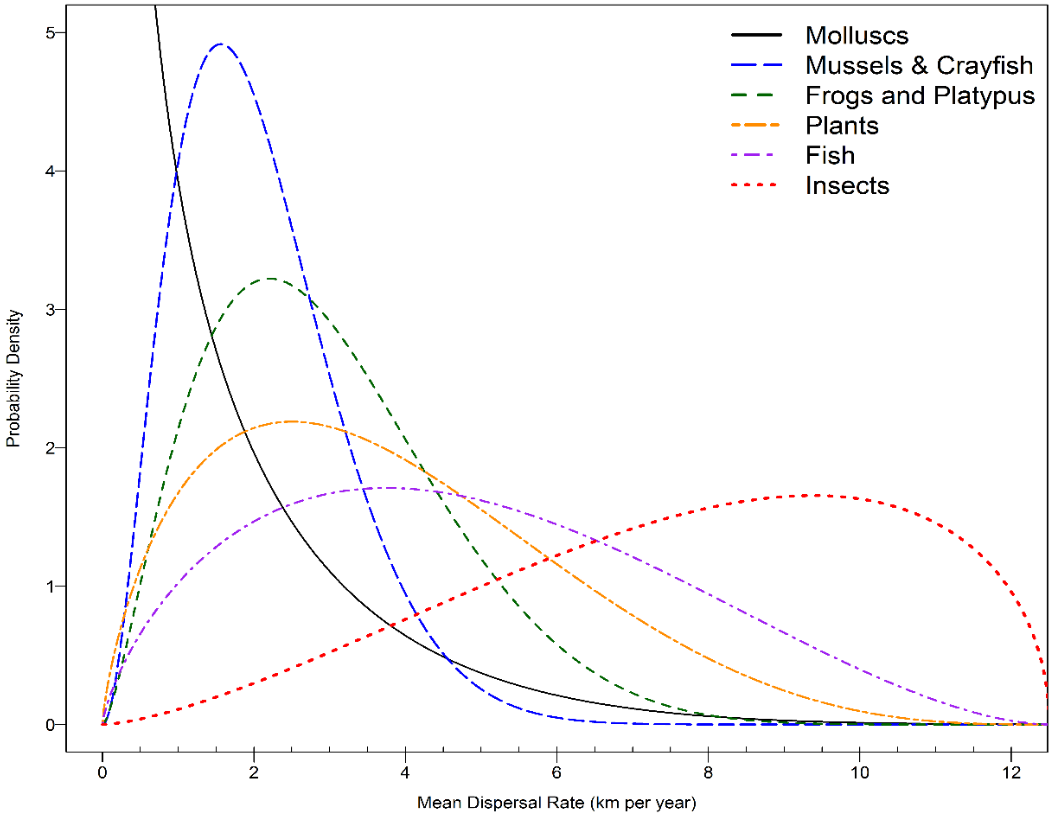

Figure 1. Estimated likelihood of potential dispersal rates for the taxonomic groups used in this study.

Habitat suitability was constrained according to dispersal mode and rate, on the basis of a dispersal kernel following a four-parameter logistic curve (Bush *et al.*, 2014a). The value of the kernel drops from an upper to lower bound (1 to 0), passing through 0.5 at the inflection point, at a rate determined by the decay constant. Species' suitability scores for any sub-catchment were weighted as a function of the dispersal kernel by their distance from occurrence records. Suitability scores thus remained unchanged close to species' known ranges, before being reduced eventually to zero at distant locations. To allow species' current ranges to extend beyond their immediate vicinity a starting distance value was included equivalent to 50 years at the given rate of dispersal, or 300 km (Bush *et al.*, 2014a), whichever was smallest. Therefore the distance to the inflection point for projections was the sum of the starting value and the dispersal rate multiplied by years between 2012 (very few species had records from 2013) and the future time step.

Species Distribution Modelling

Habitat suitability was modelled using an ensemble of five common algorithms; generalised linear models (GLM), generalised boosted models (GBM), generalised additive models (GAM), multivariate adaptive regression splines (MARS), and Maxent (Elith *et al.*, 2006). All models were run with 10 replicates, using a standard 70/30 split for training and testing data. Algorithms were run in R using the packages *dismo* (Hijmans *et al.*, 2013) and *biomod2* (Thuiller *et al.*, 2009), using the default settings and adjusted as follows: GLM, polynomial terms were ranked by AIC; GBM, fourfold cross-validation and a maximum of 2,000 trees; GAM, degree of smoothing of four. Model evaluation was primarily conducted using the True Skills Statistic (TSS) and used in weighting model importance for ensemble projection (Liu *et al.*, 2013). Recently, it has been shown that consensus methods are able to cope with prediction variability by combining an ensemble of predictions from different modelling methods (Araújo & New, 2007). By calculating the general trend among various statistical methods, ensemble modelling generally provides more accurate predictions, and is therefore recommended as an approach for dealing with inter-model variability in predictive performance (Marmion *et al.*, 2009). TSS was also used to determine a lower threshold boundary for species suitability scores and when calculating binary species range maps (Liu *et al.*, 2013). TSS scores range from −1 to 1, with 0 indicating no skill and 1 a perfect ability to distinguish positive and false scores. Poor performance of models appeared to be more common among the most widespread species, which could still have low-prevalence within their range and also be recorded from isolated, often artificial or ephemeral waterbodies in arid zones (e.g. *Bush et al. 2014*).

When modelling species using presence-only data, the spatial distribution of sites from which absences are inferred is important to model calibration and fitting (Barbet-Massin *et al.*, 2012). Previous research has suggested a radius of 200 km (VanDerWal *et al.*, 2009a) but after trialling a subset of species from each group this was found to be too restrictive and so was increased to 300 km. Background points were sampled to reflect a species' dispersal capacity, meaning a simple radius for aerial-dispersers, network-distance (including headwater crossings) for groups with mixed movements, and in the case of freshwater dependent taxa, background sites were simply limited to the same catchment, and not by distance. Locations where other species from the same taxonomic group had been observed were used first, with the remainder drawn at random (Phillips & Dudík, 2008). Reducing the overall extent of pseudo-absences meant projections were more likely to extrapolate beyond the known species-environment relationships, potentially overestimating suitability in different environments (VanDerWal *et al.*, 2009a). To counter extrapolation, projections were limited using environmental clamping whereby suitability of a sub-catchment is reduced when more than one environmental factor was outside the limits used in model construction (Elith *et al.*, 2011). Allowing extrapolation of one factor meant the model was flexible to small changes, but constrained projection under novel future climates to predict suitable habitat only in similar environmental conditions to the present.

Selection of environmental predictor variables was based on prior testing in Maxent and using AIC (Akaike Information Criterion: Akaike, 1973) as outlined by Warren and Seifert (2010). For each taxonomic group, forward selection added parameters from a set of ecologically relevant variables, excluding highly correlated alternatives when selections were made. In addition, to calibrate the Maxent model, we also used this approach to assign an appropriate beta-multiplier which controls the generality of Maxent predictions. Variable selection and evaluation statistics are provided for all species in Appendix 2 and 3. In the case of freshwater mussels, an additional predictor variable of fish community dissimilarity was introduced based on the expected fish community drawn from their SDMs (see similar approach by Lois *et al.*, 2014).

Analysis

Species distributions were projected under future climate scenarios for each of the five GCMs, two emissions scenarios (RCP 6 and 8.5) and time periods between 2025 and 2085 at decadal intervals. For each of the 70 future projections, connectivity files were used to constrain habitat suitability at 25 potential rates of dispersal. In each case, to standardise the relative impact of climate change based on species range size, the vulnerability of the species' was assessed using a species sensitivity index (SSI), following the methods described in Crossman *et al.* (2011) as the ratio between the change in habitat suitability, over the total suitability of the future. The suitability of a projection was based on the sum of stream reach lengths weighted by their modelled suitability (where the suitability exceeded the species TSS-threshold). Species with negative sensitivity values are likely to expand their range or have higher overall suitability in the future, whereas higher values occur when the species' habitat either contracts in area, or becomes less suitable.

Given the nested nature of the data, linear mixed-effects models were used to test whether the mode of dispersal significantly influenced species' projected vulnerability under climate change, and the extent to which rate of dispersal had an impact (Bolker *et al.*, 2009). Variation caused by the GCM, RCP and year used in a projection, as well as inter-species differences reflected in their current range size, could be accounted for. Dispersal rate and mode were used as nested fixed effects (for each species within a given climate change scenario). As a ratio, sensitivity scores rise exponentially as range losses approach 100% and so to control for the skewed distribution of residual variance the scores were capped at 100 (indicating range reductions of 99% or greater) and then log transformed. The GLMM were fit and analysed using the R packages *lme4*, *nlme* and *MuMln* (Barton, 2014; Bates *et al.*, 2014; Pinheiro *et al.*, 2014). In addition to the main causes of variation listed above, other species-specific factors linked to vulnerability were tested as part of the model, including mean altitude, mean latitude and climatic exposure. But ultimately only mean altitude was included. Exposure was calculated as the maximum percentage of a species' range to fall two SDs or more outside the current range for any climatic or hydrological variable used to model their distribution (Beaumont *et al.*, 2011; Bush *et al.*, 2014a). Additional tests were run using projections that included barriers to the movement of swimming taxa (fish and mussels).

Although the effect of dispersal mode was independent of dispersal rate, the dispersal rates of molluscs, frogs and crayfish are typically quite low and therefore the impact of dispersal rate uncertainty on the selection of management actions spatially may be negligible. To estimate the effect that dispersal uncertainty has on choosing conservation priorities, individual reaches were ranked according to the summed sensitivity score of species weighted by their habitat suitability under a given future climate scenario. This weighting increases the emphasis on smaller areas that contain suitable habitat for species that are most vulnerable for the chosen climate scenario. To reduce the importance of reaches whose importance has associated uncertainty, scores were discounted based on the standard deviation driven by differences among modelling methods (Carvalho *et al.*, 2011; Kujala *et al.*, 2013). Variation due to dispersal rates was then also removed from species' sensitivity scores so that comparisons for each climate scenario before and after the dispersal uncertainty was reduced.

The conservation value of each sub-catchment was based on the summed sensitivity scores of each modelled species, meaning reaches that retained suitable habitat for species that had experienced significant declines were weighted higher (Crossman *et al.*, 2011). Within each group, the conservation value was estimated 1000 times, in each cases selecting a projection for the species using a given dispersal rate, based on the likelihood function for that group. Thus in each permutation some species were assumed to have higher or lower rates than others. Conservation scores were then averaged across all permutations. Reach scores were then discounted by subtracting the standard deviation in scores resulting from differences in GCM, RCP, taxonomic group, dispersal rate and mode (Kujala *et al.*, 2013), and balanced per species to reflect the varying size of taxonomic groups. The effect of spatial uncertainty was analysed using projections for 2085 only because this encompassed the broadest range of projected differences. The resulting scores reflect the relative importance of each reach to conserving species most threatened by climate change, after minimising the error due to that particular factor. Approximately 10% of the region's streams (by length) are within sub-catchments that are at least 50% within protected areas, and assuming protected areas in the region are eventually expanded to the 17% aerial target agreed by the Convention on Biological Diversity (CBD, 2010a, c), there is the potential for new reserves to improve the capacity of existing protected areas to shelter species affected by climate change. Current protected areas are biased towards elevated sections of the Great Divide and consequently already protect many areas that are inherently important to taxa moving towards cooler habitats (e.g. Kosciuszko, Barrington Tops and Gibraltar Range N.P.'s). Thus to consider the effect of uncertainty in the SDM on conservation policy comparisons were based on the degree of overlap between the highest ranking 7% of reaches that would be selected outside of existing reserves. Reserve systems were rated according to the mean proportion of species suitable habitat protected within them, and subsequently the reduction should alternative climate scenarios take place. Overlap of conservation priorities was also compared using different taxonomic groups to rank sub-catchments.

Results

Effect Size of Dispersal Capacity Uncertainty

A wide range of plausible climate and dispersal scenarios were deliberately selected to account for the full spectrum of potential outcomes and whilst it is not possible to detail outcomes for each species, maps and summaries of trends for each are available as supplementary material (Appendix 4). Overall, the proportion of species threatened by climate change was lower under GCM scenarios based on MIROC 3.2med, CSIRO MK3.0 and CCCMA CGCM 3.1, and higher based on the MPI-ECHAM 5 and Ensemble projections, and there was a clear increase in vulnerability shared among species between RCP 6 and RCP 8.5. Figure 2 emphasises the breadth of responses among species and though some species may potential expand their suitable range (category 1), there are many more likely to experience declines. At a group level, freshwater plants, molluscs and mussels appear relatively resistant to the effects of climate change, whereas each of the other groups contain taxa that face serious threats due to loss of suitable habitat under climate change.

The results of the mixed models showed that after accounting for random-effects driven by differences in the projected climate scenarios (GCM ,RCP and Year), taxonomic group, and differences at the species-level in current range-size and mean-altitude, species vulnerability to climate change was strongly influenced by dispersal rate, and this was also dependent on their mode of dispersal (Table 1). As would be expected, the relationship between sensitivity score and dispersal rate was negative, demonstrating that as the rate of dispersal is increased species' sensitivity was on average reduced. The exception to this is demonstrated by the significance of dispersal mode and the order of sensitivity scores among dispersal modes. Model estimates of sensitivity scores were lowest among the "swimming" taxa (fish, mussels and molluscs), marginally higher among "flying" taxa (insects and plants), and highest among the "mixed" dispersal mode groups including frogs and crayfish. Thus overall, our uncertainty regarding species' vulnerability to climate change is likely to be most critical when predicting the shifts among these "mixed" dispersal groups rather than fish or plants. Including all random effects, the GLMM had an estimated r^2 of 0.832, of which the dispersal rate and mode explained approximately 7.0% and 3.4% respectively. The estimated variation explained by the remaining random effects included in the model were approximately 31.5% for the GCM, RCP and year of the projected climate scenario, 36% due to current range size, and 4.1% due to altitude. Thus although frogs and crayfish were allowed to move further and cross catchment boundaries, they were inherently more sensitive to projected scenarios of climate change than taxa such as molluscs and mussels confined to basins.

Table 1. Results of GLMM testing changes to species' (n=454) sensitivity score after controlling for effects of GCM, RCP, projection year, altitude and species current range size.

Response Parameter	Fixed Effects	Est.	SE	df$_1$	df$_2$	*F*	*p*
Species Sensitivity	Intercept	1.56	0.294	1	846889	699.81	<0.0001
	Dispersal Mode	-0.09	0.402	2	35403	741.82	<0.0001
	Dispersal Rate	0.23	0.001	1	846889	50148.7	<0.0001
	Rate x Mode	-0.11	0.455	2	846889	116.86	<0.0001
	Random Effects		SD				p
	GCM		0.181				<0.0001
	RCP		0.073				<0.0001
	Year		0.081				<0.0001
	Group		0.474				<0.0001
	Current Range		0.722				<0.0001

Est: Estimated parametric coefficient; SE/SD: estimated parametric coefficient standard error and deviation.

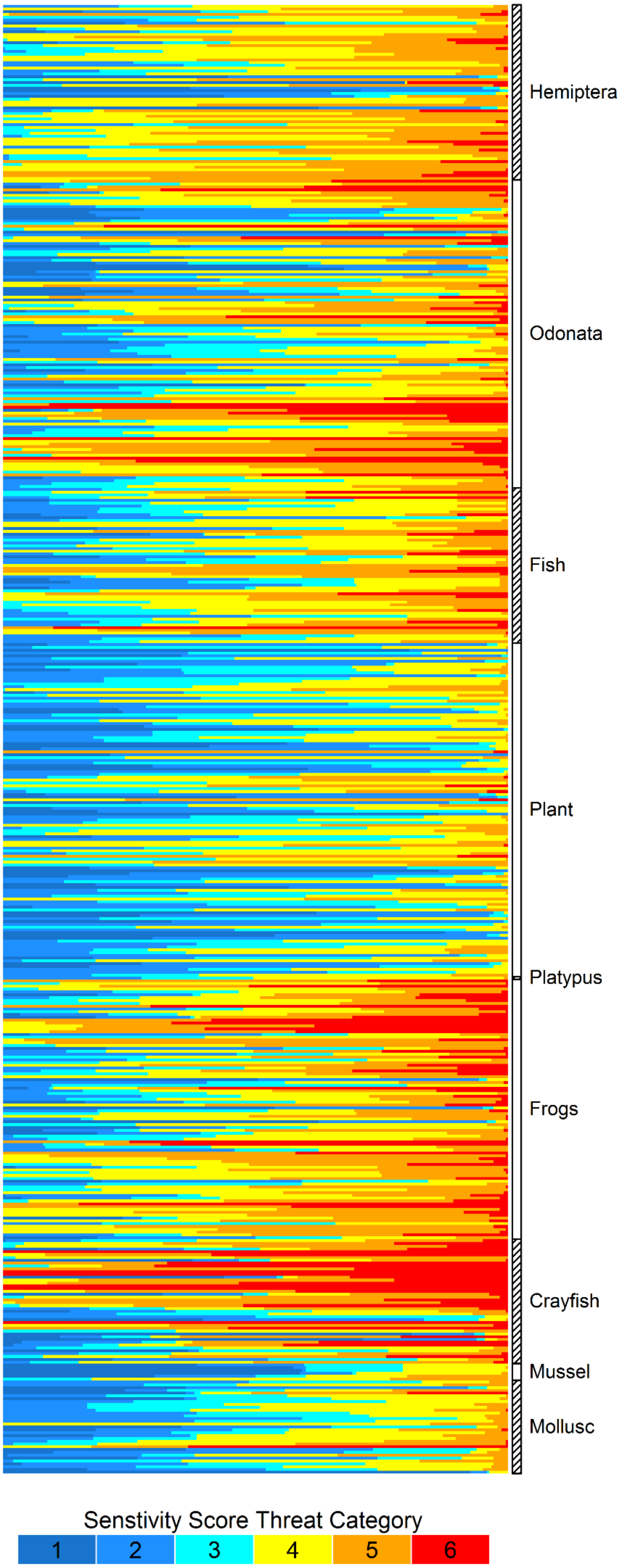

Figure 2. Vulnerability of modelled species to climate change (*n*=520), divided by taxonomic groups, under 250 different scenarios projected to 2085 (2-RCP x 5-GCM x 25-Dispersal Rate). Sensitivity scores were categorised as follows; 1) Range expansion, 2) 0-25% decline, 3) 25-50% decline, 4) 50-75% decline, 5) 75-95% decline, and 6) >95% decline.

Effect of Uncertainty on Spatial Conservation Priorities

The most importance factor affecting the vulnerability score of sub-catchments was the variation driven by selection using different taxonomic groups (Fig.3 a). Minimising the uncertainty due to GCM, dispersal rate and mode (Fig3. b, c, d) displayed some differences, but overall the importance of key regions, particularly along the coast, was still evident. Differences between RCPs appeared almost uniform and uncertainties were therefore not influential for prioritisation. Interestingly the scores for headwaters, particularly around the northern coastal catchments (Macleay and Clarence basins) were greatly reduced when variation due to dispersal mode was discounted. By contrast, whilst the importance of coastal lowlands for conserving species threatened by climate change was not common to all taxonomic groups (Fig. 3a), they scored highly across different dispersal modes. The value of headwaters was typically high, but reduced if we discount on the basis of GCM uncertainty because under some GCM projections, suitable habitat for some climate threatened taxa is lost entirely. Different sources of uncertainty also had varying spatial patterns within the MDB, but the highest conservation priorities were consistently in main stem sections of the Murray and Murrumbidgee.

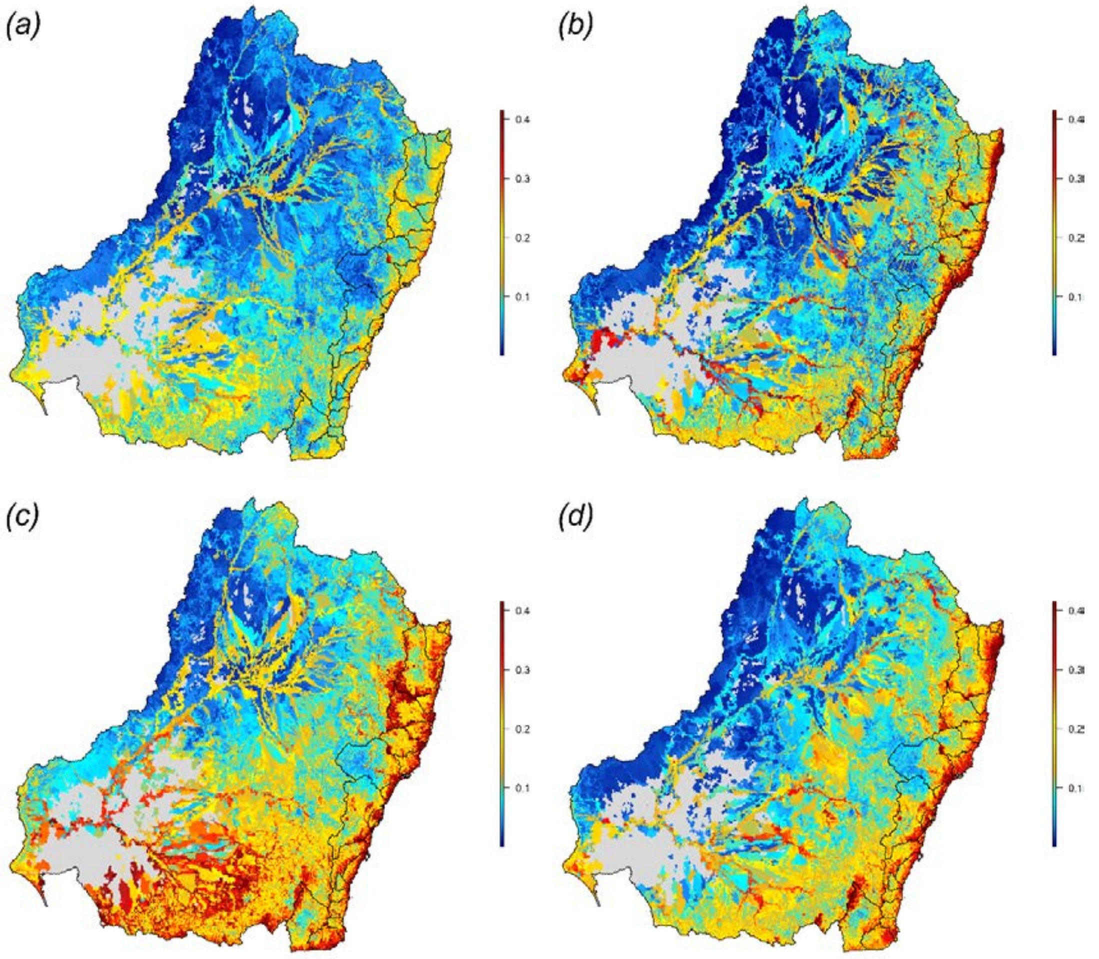

Figure 3. Mean species sensitivity to climate change in 2085 after discounting uncertainty due to a) taxonomic group, b) GCM, c) dispersal rate, and d) dispersal mode. Black lines mark catchment boundaries.

Although not the main focus of this study, the reduction of sub-catchment scores due to higher variability of scores for different taxonomic groups (Fig.3 a) demonstrates the importance of ensuring environmental decisions are based on a representative selection of taxa. Conservation priorities will vary among different taxa and further demonstration is provided in Appendix 5.

As predicted by the mixed-effects models, even after accounting for and discounting dispersal uncertainty from conservation priorities, a significant challenge will be balancing the differences between preferred solutions chosen using projections from different GCMs. One approach is to consider the conservation opportunities lost if a conservation strategy was based on planning for a particular climate scenario, but future climates followed an alternative pathway (Table 3). Conservation priorities for different GCMs showed only modest similarity, with the greatest overlap between the sub-catchments chosen using the Ensemble and MPI projections. By using one climate model to plan conservation priorities, under alternative scenarios the mean representation of species was significantly reduced. However, despite these differences, there are many areas of overlap common to most climate scenarios in which action could be taken (Fig. 4). Furthermore, discounting sub-catchment scores to minimise uncertainty due to dispersal did not significantly affect the agreement between priorities from different GCMs (Fig. 4b). This suggests that sub-catchments with potentially suitable habitat for threatened species is typically not beyond the dispersal reach of most taxa.

Table 3. Overlap (bold), average representation (diagonal) and expected conservation losses (italics) when planning conservation priorities for a climate scenario, and another takes place (based on RCP 8.5 in 2085). Representation is based on all taxa accounting for only range extent within the NSW study region. Sub-catchments scores were discounted Dispersal uncertainty was reduced before reach priorities were calculated.

Scenario planned for:	CCCMA CGCM 3.1	CCSR MIROC 32med	CSIRO MK3.0	Ensemble	MPI ECHAM5
Scenario that takes place					
CCCMA CGCM 3.1	21.5	**43** (*2.2*)	**44** (*3.4*)	**26** (*6.1*)	**42** (*4.8*)
CCSR MIROC 32med	**52** (*2.9*)	25.1	**35** (*5.2*)	**26** (*5.2*)	**34** (*5.2*)
CSIRO MK3.0	**57** (*1.8*)	**33** (*5.8*)	29.7	**58** (*2.7*)	**62** (*2.7*)
Ensemble	**35** (*6.7*)	**19** (6.*9*)	**52** (*3.8*)	28.3	**66** (*2.6*)
MPI ECHAM5	**51** (*3.7*)	**29** (*5.7*)	**62** (*1.9*)	**67** (1.9)	26.3

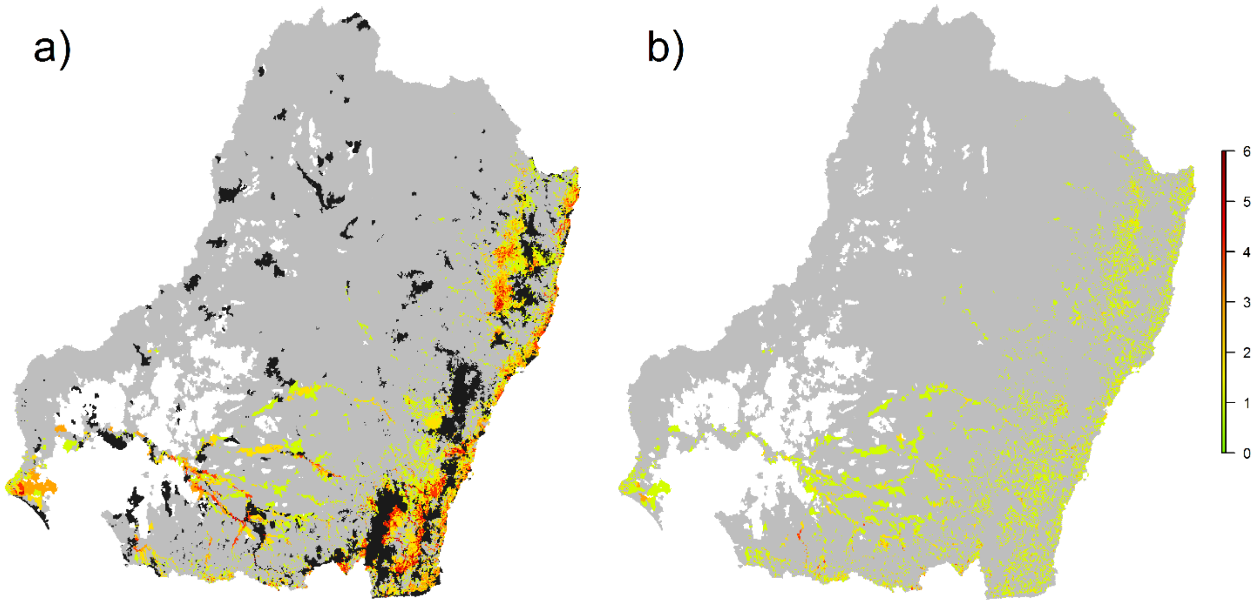

Figure 4. Overlap between conservation priorities based on projections for five GCM scenarios (RCP 8.5, 2085) with existing protected areas marked in black (a), and difference in the number of overlapping priorities before and after discounting for uncertainty in dispersal (b).

PART TWO - Alleviating Non-Climatic Stressors: Uncertainty in Management Action and Location

Methods

To test whether climate change has the potential to influence the decision making about managing non-climatic threats to species, we focused on the fish fauna of New South Wales (NSW) in south-east Australia. As before, though the focus was on NSW, the study boundaries included all 22 coastal catchments within the state, and the entire Murray-Darling Basin (MDB: Fig. 5). Species distribution-models were prepared for 55 fish present in the region, including seven non-native exotic species, and their distributions projected under current and future climate scenarios to all reaches. Habitat suitability was then constrained by several functions using the intensity or strength of the local threat (e.g. land use intensity, exotic species habitat suitability) and the sensitivity of each native species to the stressor. Management actions were then tested to see what improvement in species overall suitability could be achieved by removing particular stressors.

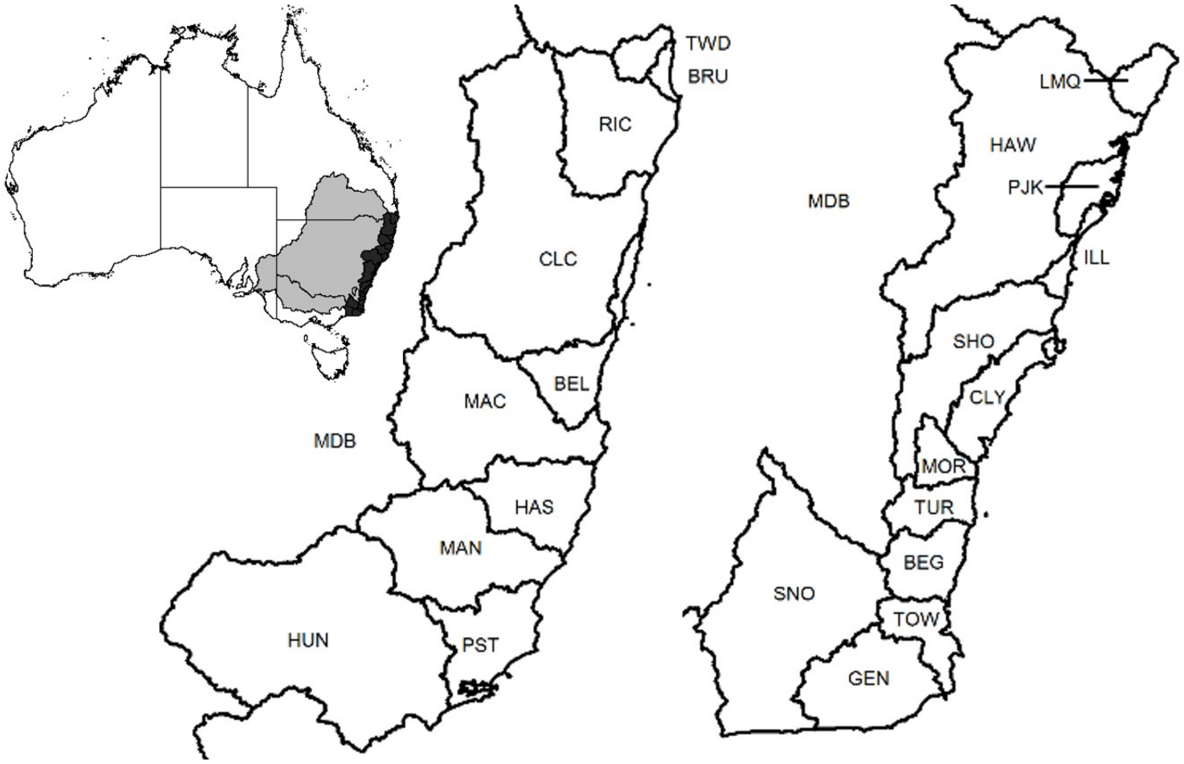

Figure 5. Study region within Australia, and the northern and southern halves of the coastal basins. The Murray-Darling Basin (MDB; light grey), and the coastal basins (dark grey) from north to south; Tweed (TWD), Brunswick (BRU), Richmond (RIC), Clarence (CLC), Bellinger (BEL), Macleay (MAC), Hastings (HAS), Manning (MAN), Port Stephens (PST), Hunter (HUN), Lake Macquarie (LMQ), Hawkesbury, Port Jackson and Georges River (PJK), Lake Illawarra (ILL), Shoalhaven (SHO), Clyde (CLY), Moruya (MOR), Tuross (TUR), Bega (BEG), Towamba (TOW), Genoa (GEN), Snowy (SNO).

The potentially suitable habitat of native species was constrained according to three threat categories; barriers to movement, catchment disturbance and introduced species. The habitat suitability projected by the SDMs describes the potential for suitable habitat to exist based on the environmental conditions, but additional processes such as connectivity and anthropogenic threats prevent the species from occupying all climatically suitable locations (Fig. 6). Thus, species suitability was firstly limited to only those portions of the stream network that remained accessible after considering the position of occurrences in relation to natural and artificial barriers. Species suitability was then further constrained based on exposure to a local threat. The reduction in suitability at each site was a function of the species sensitivity to that threat and the local threat intensity. The degree of threat posed by a particular stressor was determined through species recovery plans, catchment management reports and the scientific literature. Although fish management receives a relatively high profile in freshwater ecology, and a large body of literature describes the threats faced by many native species, the severity of threat could only be scaled according to a qualitative ranking (Drew *et al.*, 2008)(Appendix 6). Species sensitivity to a threat was assessed in relation to degradation through changes in land-use, modification of the flow-regime and the presence of each of the seven exotic species. Class 1 indicated no impact and implies that the native species are tolerant of the presence of certain stressors. Classes 2 and 3 were applied when the threat would cause a small or moderate reduction in population size or abundance over time. Class 4 stressors to species occurred in cases where a threat is likely to cause the loss of a species from an area over time. The impact that management actions could have were therefore determined by the increase in potential habitat fish can occupy after artificial barriers are removed, or the intensity of local threats is reduced. There was insufficient information to estimate how the impact of multiple stressors could combine and so suitability was reduced according to the maximum constraint of any single local threat.

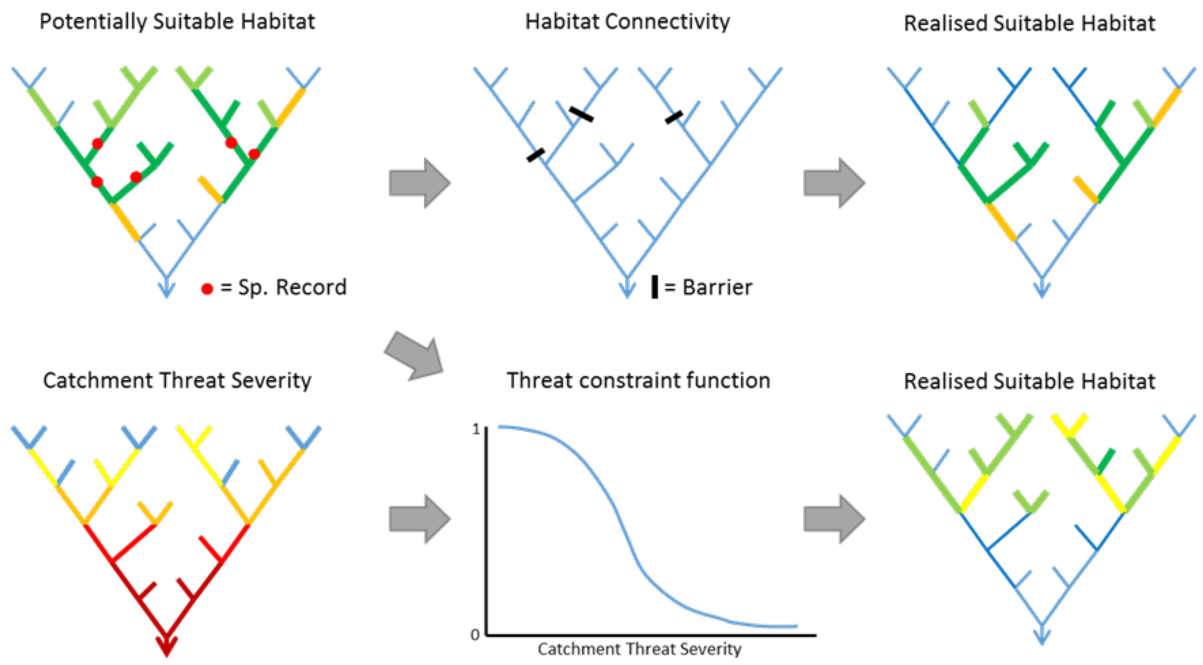

Figure 6. Process by which potentially suitable habitat for each species was constrained to reflect accessible habitat and the distribution of threats. In this example, the species favour mid-order streams in the network, but several headwater dams would prevent it from accessing upstream habitats. On the lower level, the constraint function declines in response to a threat that is greatest among high-order reaches. By weighting habitat suitability by the constraint function, the species habitat suitability is most affected in the higher-order streams.

Species suitability was then constrained across its range according to the level of exposure present and the species' sensitivity (Fig. 7). Species suitability was weighted by a constraint function that ranged between 0 and 1 such that if a species is not influenced by a particular threat then suitability is simply weighted by 1 and remains unchanged, or if a threat is serious, a weighting of 0 removes suitable habitat for the species in that area. The constraint function was based on a modified four-parameter logistic curve;

$$a + (b-a) / (1+ \exp(c \times (d - x)))$$

where a and b are the maximum and minimum asymptotes respectively, c is the slope factor that determines the rate of decline around the inflection point d. The strength of the stressor is indicated by x. By modifying b, the maximum reduction in response to a threat can be controlled. Likewise, the inflection point d can be altered to reflect at what strength the threat begins to reduce habitat suitability. The values of b were evenly split for each threat classes 1-4 (i.e. 1, 0.66, 0.33 and 0 respectively). The inflection point of a species' constraint function was determined by the 95th percentile of exposure values at occurrence sites. This ensured that constraints did not remove suitability from the majority of sites where species are known to have occurred, and constraining suitability among the most exposed sites where the species may occurrence may have been only temporary. Despite the well documented impact changes to hydrology have on many species, this was not well reflected by the scores of sites with occurrence records. This is likely to be due to many observations of fish species congregating beneath dams and weirs, in habitat that is otherwise locally unsuitable due to the modified hydrology. Instead, general constraint parameters were chosen for each species in each class sensitive to flow regime disturbance. The slope factor was also allowed to vary to both set a realistic graduated decline in suitability as the severity of a threat increased, whilst also ensuring that species' suitability at sites in good condition would not be constrained.

Connectivity and Barriers

Species distributions were modelled under current climate conditions and projected to 2085 under five alternative future climate scenarios. These included four GCMs that represent the breadth of possible outcomes, and an "ensemble" scenario that is a balance of what were considered the most likely GCM projections. Species' habitat suitability was projected for each stream reach and suitable habitat initially constrained based on species occurrence to the same catchments. A reach is the stream section that separates two junctions and are defined according to the Australian Geofabric V.2 database (GEOFABRIC, 2013). Catchment connections and movement pathways were modelled in R using the package *igraph* (Csardi & Nepusz, 2006).

Species' distributions were initially limited based on catchment boundaries. Distributions were then further limited by only allowing suitable habitat to occur in stream reaches that were connected to sites with species records. A species was considered connected to all reaches downstream of occurrence records, but disconnected from reaches that were either above a major waterfall, or above artificial barriers such as dams and weirs. Dam and weir locations in NSW were provided by the Department for Primary Industries: Fisheries (Matthew Gordos pers. comm.) and from the Murray Darling Basin Authority weirs information system (MDBA, 2013). Where recorded, barriers with fish-ways installed or those surveyed and determined to not be limiting movement were remove from the list of barriers. Although some road-crossings (over 5000 in the study area) may be barriers to movement, they were subsequently discarded from the analysis because the majority do not pose barriers to movement (NSW Department of Primary Industries, 2006a). Waterfall locations were drawn from the Geofabric database (GEOFABRIC, 2013). Overall, the analysis included 3,102 artificial barriers (weirs and dams), and 1,575 waterfalls (Table 4**)**. Additional multi-barrier removal options were then considered because multiple barriers often separate headwaters from the river mouth. Therefore, options for increasing connectivity included removal of single barriers and chains of up to nine artificial barriers downstream from each to the river mouth.

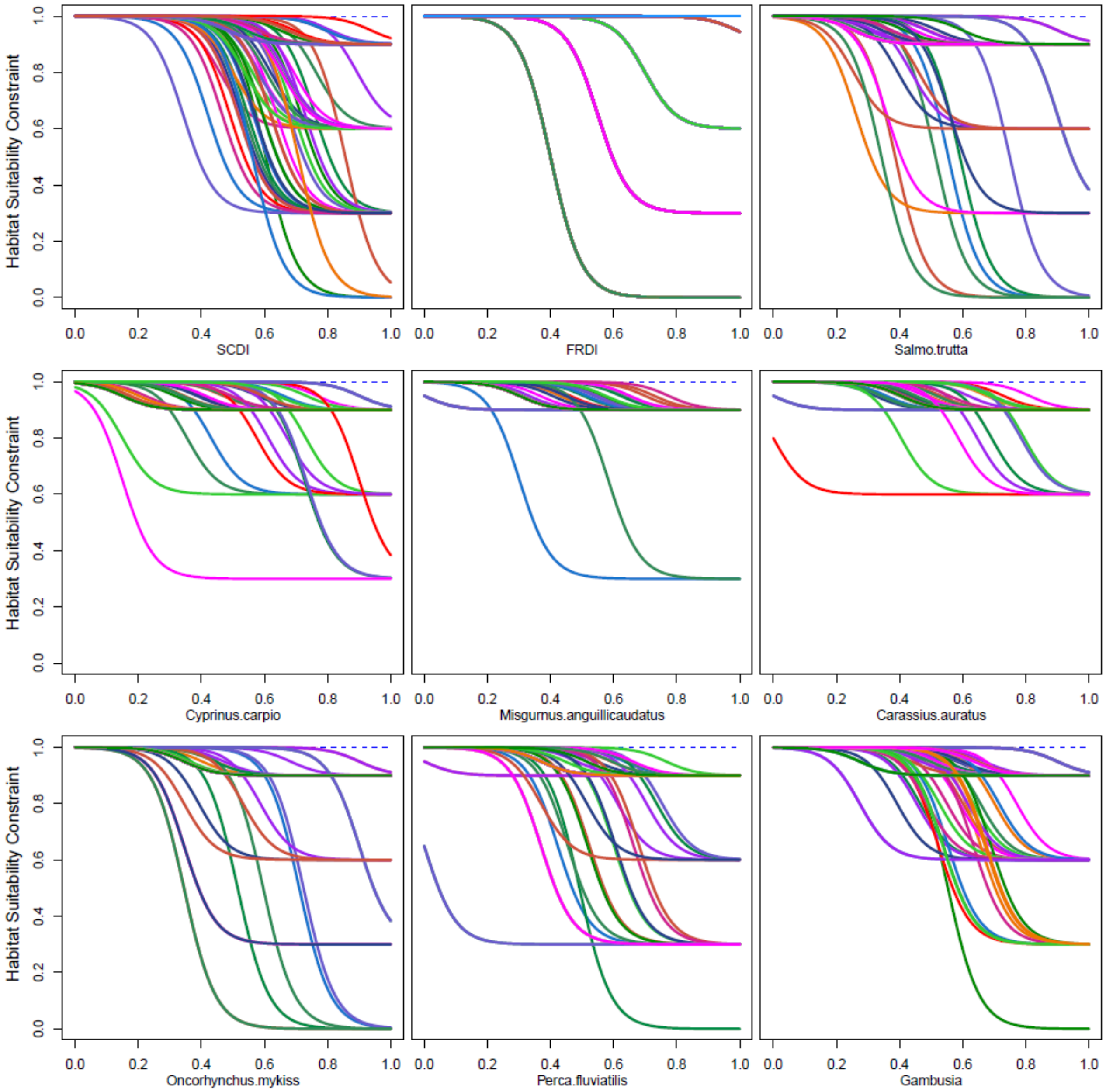

Figure 7. The habitat suitability constraint functions of 49 native fish species in response to changes in land-use (Sub-Catchment Disturbance Index: SCDI) and flow modification (Flow Regime Disturbance Index: FRDI), where 1 indicates the most disturbed reaches, and the modelled suitability of seven exotic species. The suitability of native species is reduced by weighting against the constraint function response to local threats.

Table 4 – Summary information for each catchment: number of reaches, number of barriers, and their combinations singly and as chains (Barriers X), the number of management combinations considered (Mgmt. X), the numbers of native (including potential marine immigrants), threatened and climate sensitive fish species considered.

Catchment	Reaches	Barriers	Barriers X	Mgmt. X	Native	Sensitive
Tweed	578	12	26	416	21	9
Brunswick	225	2	3	48	24	7
Richmond	2470	47	169	10816	26	12
Clarence	8669	40	117	29952	29	13
Bellinger	1410	1	2	32	21	8
Macleay	4514	40	55	56320	20	6
Hastings	1941	14	17	1088	20	6
Manning	3432	42	149	38144	18	3
Karuah	1064	6	7	112	18	5
Hunter	8536	63	79	20224	25	8
L. Macquarie	469	11	13	832	17	2
Hawkesbury	7364	121	342	350208	31	9
Georges R.	415	67	268	17152	21	4
Illawarra	257	13	24	1536	16	1
Shoalhaven	2064	31	57	14592	23	5
Clyde	1140	5	6	96	15	2
Moruya	424	3	4	64	17	2
Tuross	827	2	3	192	18	3
Bega	1019	12	14	3584	18	1
Towamba	782	3	4	256	17	1
Gippsland	1834	0	0	64	19	2
Snowy	4753	19	31	31744	23	3
Murray-Darling	166464	2523	23538	24102912	36	16

Many of the native fish in NSW coastal catchments are diadromous (i.e. they have marine phase in their life cycle) and this had two consequences for the modelling methods. First, rather than estimate connected habitat from occurrence records, connectivity was drawn from the river mouth because in the long-term, the species must be able to migrate upstream to access suitable stream reaches. Secondly, diadromous fish could potentially disperse to neighbouring catchments in the future and access suitable habitats there. As a result, alongside the resident species, potential for marine colonisers was added by adding pseudo-occurrence points to the river mouths of neighbouring catchments separated by up to 200 km. This distance was chosen to offer a relatively wide marine dispersal option, because juveniles may be passively carried great distances, but by the same token not so far that species could easily cross the Bass Strait, which has not occurred among some diadromous species. A final modification was to ignore the presence of barriers for five species (*Anguilla australis*, *Anguilla reinhardtii*, *Galaxias brevipinnis, Gobiomorphus coxii* and *Mordacia mordax*) because they have the ability to cross or circle around many barriers and their habitat connectivity would otherwise be poorly reflected by strict barrier limitation.

Land-use and Flow Modification

The local intensity of river disturbance was based on the River Disturbance Index (RDI) (Stein *et al.*, 2002) which estimates the potential human disturbance of rivers based on accumulated upstream impacts of human landscape modification and flow regulation. Although this analysis did not consider management actions that specifically reduced the disturbance resulting from flow modification, species suitability scores were also reduced where this was considered a threat because it was considered a key impediment to species recovery in some locations (Bunn & Arthington, 2002).

A major contributor to the decline of freshwater fish in Australia has been the alteration of flow regimes as these both shape the quality of habitats that species prefer, and provide cues for spawning and migration (Arthington & Pusey, 2003; Pusey *et al.*, 2004). Although the removal of barriers could involve the demolition of dams or weirs, improvements to fish passage are more often achieved by constructing or retrofitting fishways of various designs (NSW Department of Primary Industries, 2006c). The removal of movement barriers does not therefore necessarily result in restoration of natural flows. In Australia, restoring natural flows often relies on the release of environmental water, but due to the lack of coordinated research it is difficult to estimate the downstream ecological consequences (Gilligan & Williams, 2008). As a result, habitat suitability was constrained in reaches affected by flow modification but management actions to remediate its effect were not tested. Within the RDI, disturbance due to land-use was based on the Sub-Catchment Disturbance Index (SCDI), and flow modification on the Flow-Regime Disturbance Index (FRDI).

Introduced Species

The final stressor to be addressed considered the potential for habitat suitability to be reduced where there is predicted to be overlap with introduced species. As with river disturbance factors, the native fish species sensitive to the effects of introduced species, be it by competition, predation, habitat modification or spread of parasites and disease; are affected where their ranges are projected to overlap (Macdonald *et al.*, 2012; Vilizzi *et al.*, 2014). Habitat suitability is related to the maximum potential abundance (VanDerWal *et al.*, 2009e) and was therefore used to estimate the intensity of introduced species' interactions on native species. As with native species, habitat was also limited to reaches connected to occurrence records. Rather than consider a management action to remove or control a particular introduced species, these species were considered in four groups to better reflect both the requirements for their control, and the native species most likely to benefit (Rowe *et al.*, 2008). Group 1 included *Carassius auratus* (Goldfish), *Cyprinus carpio* (Common Carp) and *Misgurnus anguillicaudatus* (Oriental Weatherloach) because these species have some similarities and would all be affected if controlled by a method such as rotenone (Rowe *et al.*, 2008). Although rotenone has also been used to try and control *Gambusia holbrookii*, it was placed in Group 2 on its own because its control requires careful management and because of the collective threat it poses to a number of the native species. Group 3 included the game fish: Brown trout (*Salmo trutta*) and Rainbow Trout (*Oncorhynchus mykiss*), and Group 4 was for and *Perca fluviatilis* (Redfin Perch) only.

Threat Management

In this analysis we considered the removal of barriers to fish movement both on their own, and as chains to better understand their importance in restoring catchment connectivity (Hoenke *et al.*, 2014). Every barrier in the catchment was considered for removal singly, and then as part of a chain of up to nine to the river mouth. Barriers were assumed to affect all species equally, and that barrier management would then successfully allow the free passage of all species (Poulos *et al.*, 2014).

The impact of human land-use or the suitability of habitat for introduced species was reduced by managing either 50, 100 or 200 km of reaches within each catchment. To select where the management of land-use or exotics species occurred in a way that reflects more targeted management, a priority weighting was developed to reflect the potential benefit to native species in each reach. The priority weighting at each reach was the sum of species' sensitivity classes (1-4) weighted by their habitat suitability. Consequently the weighted intensity of threats within each reach identified where the greatest gain to native species could potentially be achieved. In the case of land-use, streams were restored to a relatively “good” condition of 0.10 whereas the suitability for introduced species at managed sites was reduced to zero. Although there are obvious difficulties in removing an invasive species entirely, the management action considered could either aim to eradicate the target species, or continue to reduce them to such low numbers they have no ecological effect. Unfortunately information on the true costs of these actions was not available so a more systematic cost-benefit analysis to prioritise options could not be done (O'Hanley *et al.*, 2013). Instead the "cost" of threat management was measured by either the number of barriers removed, the total reduction in SCDI scores, weighted by the length of the reaches, or the total suitability of invasive species removed, again weighted by reach length. Likewise, the improvements for native fish species were estimated on the basis of summed habitat suitability weighted by reach length. Under current climatic conditions and using projected habitat suitability under five future climate scenarios, habitat gains were summarised across all native species individually (*n*=49); for species currently present within the basin; for both species present and diadromous species that may colonise via marine dispersal; and for the subset considered sensitive to climate change (*n*=26, predicted to decline by 50% under at least one climate scenario). Species currently listed at state or federal level as threatened, (*n*=11) were only compared to other approaches within the MDB because few threatened species occur in any other basin.

Results

Impacts of Non-Climatic Threats

Within the study region, the presence of non-climatic stressors and constraints were estimated to have reduced the potential distribution of native fish species by an average of 48% (+/-26). On average artificial barriers alone were judged to be responsible for a 19% (+/-18) reduction in range extent, with further constraints due to overlaps with reaches affected by modified flow regimes (7%), human land-use (3%), and the groups of exotic species (4%, 17%, 12% and 3% respectively) (Fig.7)(see also Appendix 7 and 10). Note that these reductions were based on habitat within the NSW catchments only and many species have larger potential ranges e.g. *Leiopotherapon unicolor*. Under future climate scenarios the impact of stressors as a proportion of range extent declined; up to 10% based on the MPI climate model. Note that these effect sizes are reported proportionally, but should be interpreted in conjunction with projected range shifts as reductions in stressor impacts can result from expansion of exotics or contraction of species' ranges. This assumes without management action that threats remained constant, apart from changes in distribution of exotic species. Assuming no management took place, the proportion of projected ranges affected by the combination of constraints under future conditions was consistent with current conditions, although again there was significant variation among species e.g. lower proportional effect on *Tandanus tandanus*, and higher impact on *Potamolosa richmondia*. Further information on the sensitivity of species to each stressor and differences in the proportion of species' ranges affected can be found in Appendix 7.

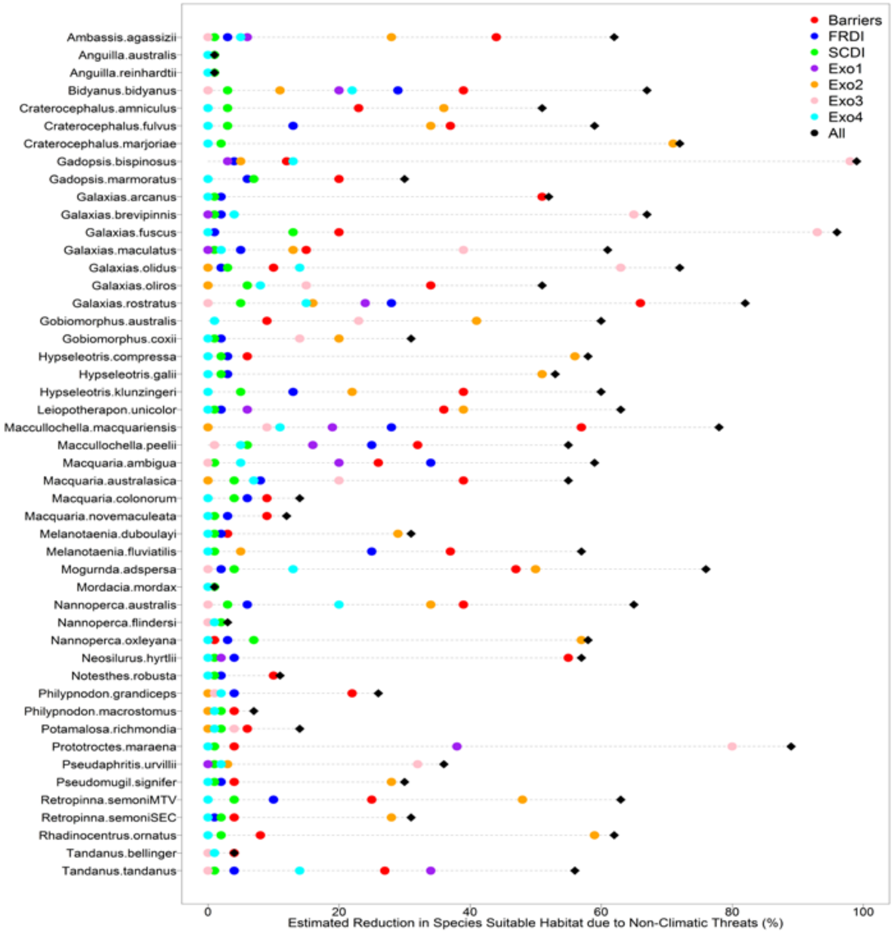

Figure 8 – Percent reduction in the current range (km of suitable habitat) of native fish species within New South Wales basins due to the presence of non-climatic threats including barriers to

movement, modified flow regimes (FRDI), land-use (SCDI), exotic species (see text for species), and the combination of all combined.

GUIDE TO MANAGEMENT ACTION PLOTS

The management actions for six categories of non-climatic stressors amount to 1,024 alternatives for each level barrier-removal, for which there may be multiple examples at different locations and as many as nine barriers. With so many options many plots only display a subset of the levels of barrier removal. To summarise this information the plots first display a colour-bar that indicates the levels of management for each category along the x-axis. Thus within each block for dam-removal at the highest level, there are four categories of land-use remediation (0, 50, 100 and 200 km). Within those are categories for management of exotics species Group 1 and so on. In some the colour bars are greyed out for a group of exotic species because they did not occur in a catchment and hence were not considered.

Below this bar are the potential gains of each management action under current and future climate projections. Each plot marks the mean value in black, and if this combination was used multiple times then the range is shown as a grey line. Thus it is possible to view the minimum, mean and maximum of

Potential Impacts of Management Actions for Species

Species' habitat-gains (increase in km's of suitable habitat potentially available) varied markedly, both in relation to climate change and management actions. Given the wide range of potential climate scenarios used, and hence the range of possible responses species could show, it was expected that identical management actions could offer divergent outcomes. Therefore, the benefits of a given management action in the future were conditional on the limitations placed by environmental factors for suitability. If a given species was projected to decline across a basin, the predicted benefits of any action, relative to outcomes under current conditions, would also decline.

Figure 9 shows an example of the predicted response to alternative management actions for freshwater herring (*Hypseleotris compressa*) in the coastal catchments of NSW under current and future climate projections (see box above to help with interpretation). For *H. compressa*, increases in suitable habitat rely on control of exotic group 2 (*Gambusia holbrooki*), and greater increases are subsequently possible if a single barrier is removed (in one of five catchments). Under future projections, barrier removal could support higher increases, although in each case the environmental suitability under a particular scenario controls the magnitude of gains. Under the MPI-ECHAM 5 projection the maximum possible gain is achievable with any combination that include at least one barrier being removed, whereas under the CSIRO model further gains still require control of *G. holbrooki*. Finally, there are also locations in which the same combinations result in negative outcomes for *H.*

compressa, even after managing *G. holbrooki*, because barrier removal enables *G. holbrooki* to access *H. compressa* habitat where it was previously assumed to be absent. These are noteworthy because barriers whose removal risks the introduction of exotic competitors/predators may not aid native species expansion, but could be key to ensuring their persistence. Barriers with negative outcomes are identified in the basin reports.

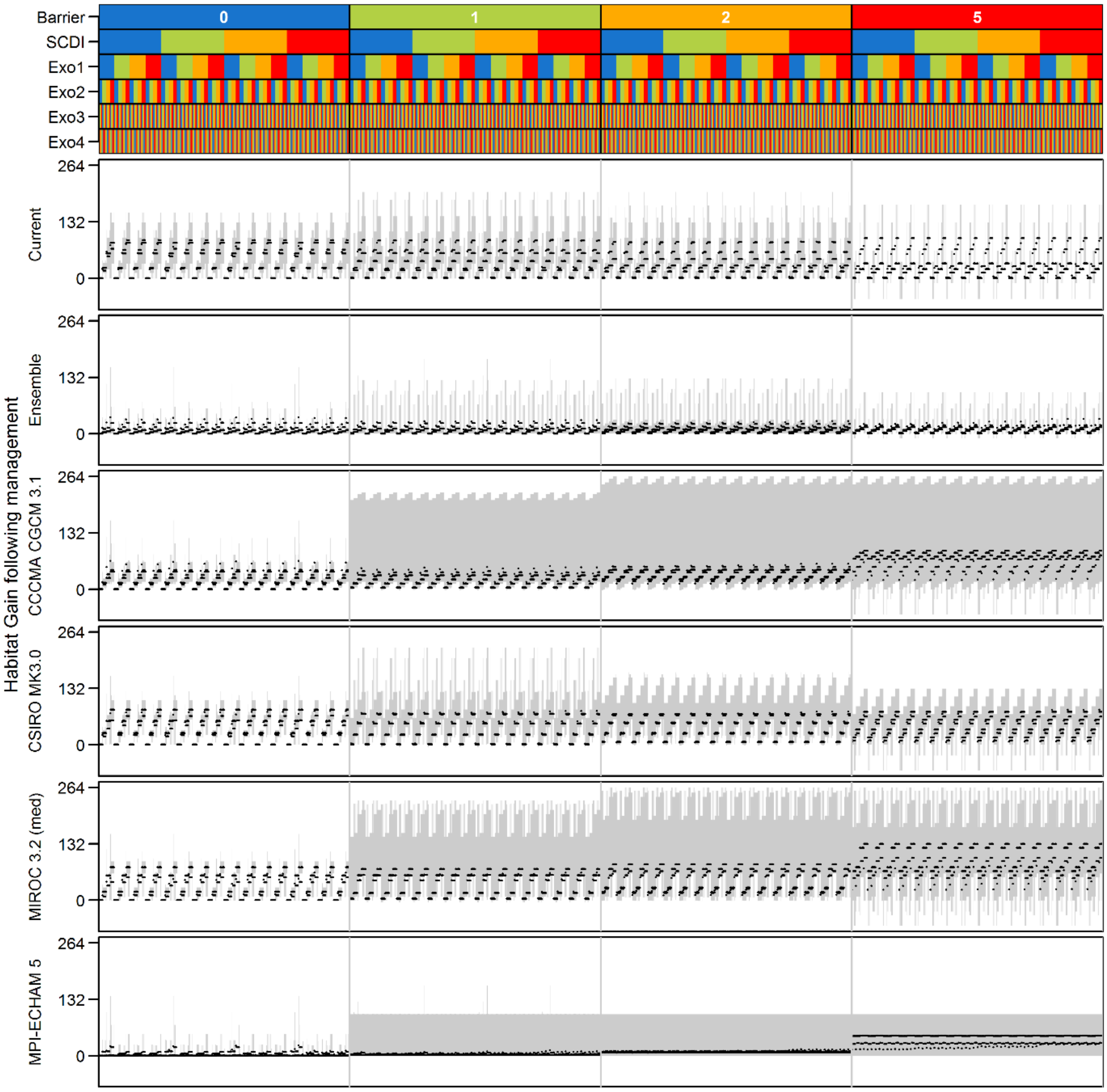

Figure 9 – Plot of management actions and potential habitat gains for *Hypseleotris compressa* under current and future climate scenarios in coastal catchments of NSW. Coloured bands indicate different management costs for each category (see the Guide to Management Action Plots for details); and removal of 0, 1, 2 and 5 barriers. Grey bars display the range of habitat gains (km of suitable habitat), with mean values marked in black.

In addition to considering what actions might best improve the long-term future of a species, it is also possible to consider which actions in each catchment offer the greatest improvements. For example, Fig. 10 shows the biggest potential gains for native species within the Bega basin would arise from better control of *G. holbrooki* (exotic group 2); and *Salmo trutta* and *Onchorhynchus mykiss* (exotic group 3). As before, the potential gains under future climate projections vary greatly from greater gains (CSIRO MK 3.0) to less than half that expected under the present conditions (MPI ECHAM 5). Under the MPI projection, habitat suitability for exotic group 3 as well as native species declined, meaning not only were potential gains lower but the impact of the exotics was reduced. Thus the control of exotic group 3 would remain a higher a priority if we expected future climates to follow the CSIRO projections, rather than the MPI model.

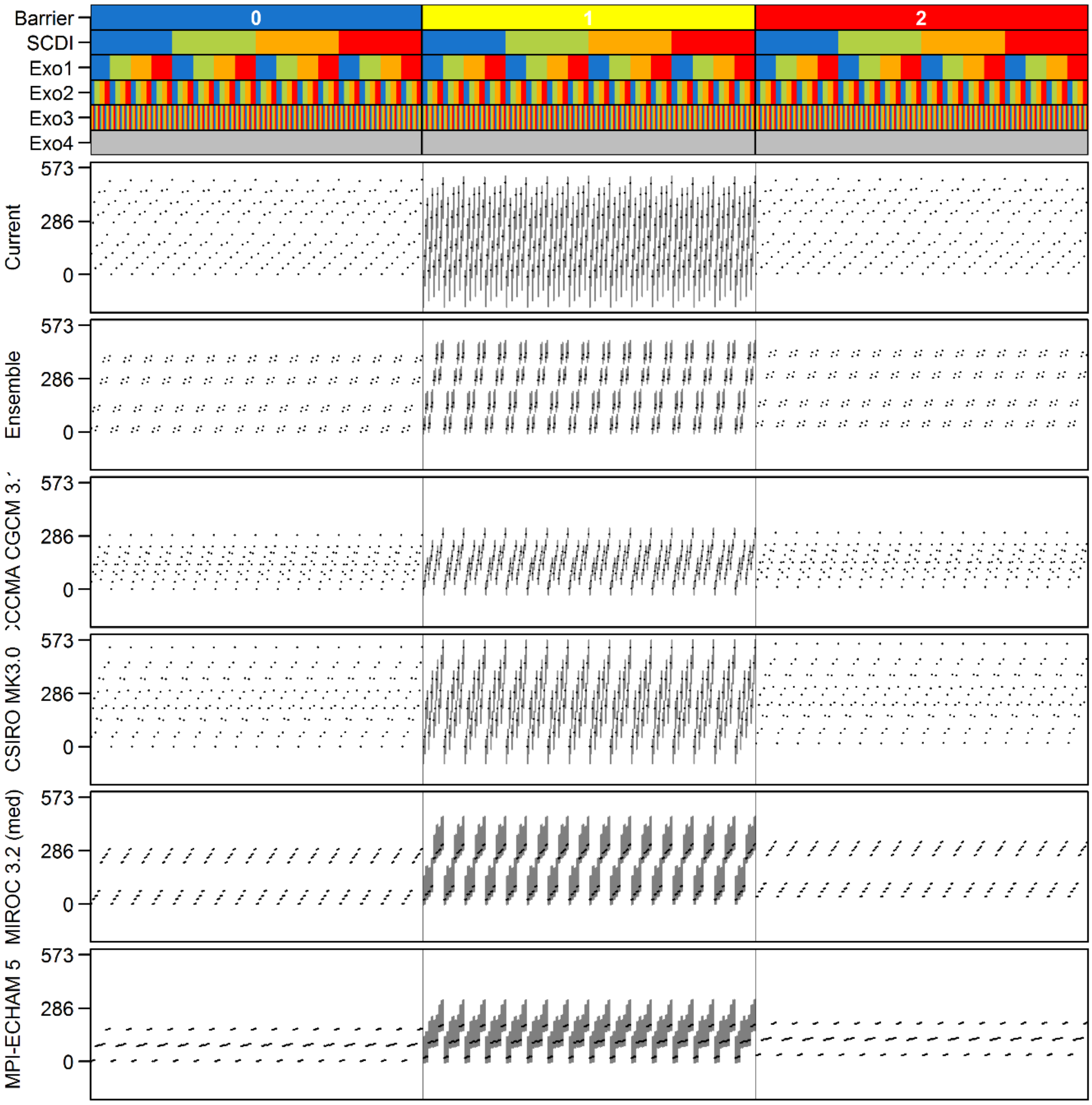

Figure 10 – Plot of management actions and potential habitat gains for the native fish community in the Bega basin under current and future climate scenarios. Coloured bands indicate different management costs for each category (see the Guide to Management Action Plots for details); and removal of 0, 1 and 2 barriers. Grey bars display the range of habitat gains (km of suitable habitat), with mean values marked in black. Fish in exotic group 4 did not occur in the Bega basin.

Management Priorities

The results above demonstrate that the benefits of management actions for native fish may be perceived very differently if we consider the impacts of projected climate change. Whilst this makes it more difficult to identify the optimal combination of actions required to meet specific targets, often the challenge is simply decide where and how to act first. Given the resources to achieve a particular combination of management actions, would the highest priority remain effective under alternative climate change scenarios, or would the same management action be better placed at other locations? Fig. 11 shows the efficiency of barrier removal within the Bega basin. If resources were available to remove a single barrier, the choice would depend on the group of fish used to scale priorities (current community, current plus possible diadromous immigrants, or the most climate-sensitive species). Furthermore, many of the barriers risk potentially allowing introduced exotic species downstream from invading upstream (Fig.11d). Among the other stressors considered within the Bega, the distribution of each particular threat was different (Fig.12), and within the scope of those reaches considered for management, their relative priorities were consequently also spatially variable. An interesting connection between combinations of actions is demonstrated by the priorities for managing exotic group 3 (Fig.12h). The current extent of these exotics does not extend to the north-west of the basin, but the reaches are still potential priorities because in many combinations considered the removal of a barrier allows access which impacts upon the community (Fig.11d). Finally, the third column of Fig.11 conveys the variation in reach management priorities between different GCM scenarios and demonstrates how the relative rank of current priorities may or may not continue to be reliable.

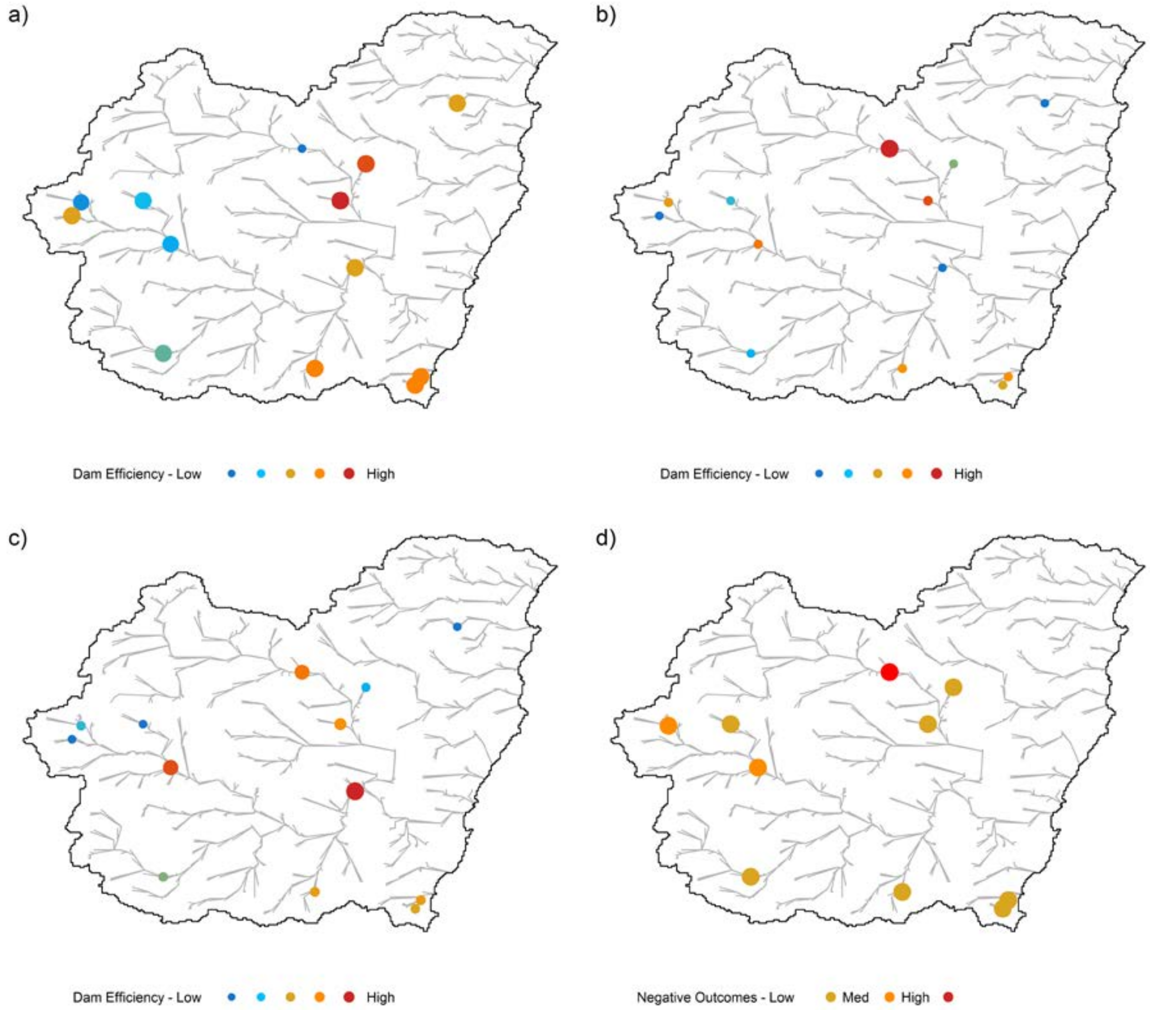

Figure 11 – Differences in the rank of artificial barriers in the Bega basin for removal. Barriers are scored either based on the increase in available habitat to; (a) all native fish present in the basin, (b) all native fish including potential colonisers with marine dispersal, and (c) among species considered most climate-sensitive. Panel (d) also indicates where barrier removals also result in negative outcomes for a single species. Low, med and high reductions were equivalent to losses <10, 10-50 and >50 km of suitable habitat.

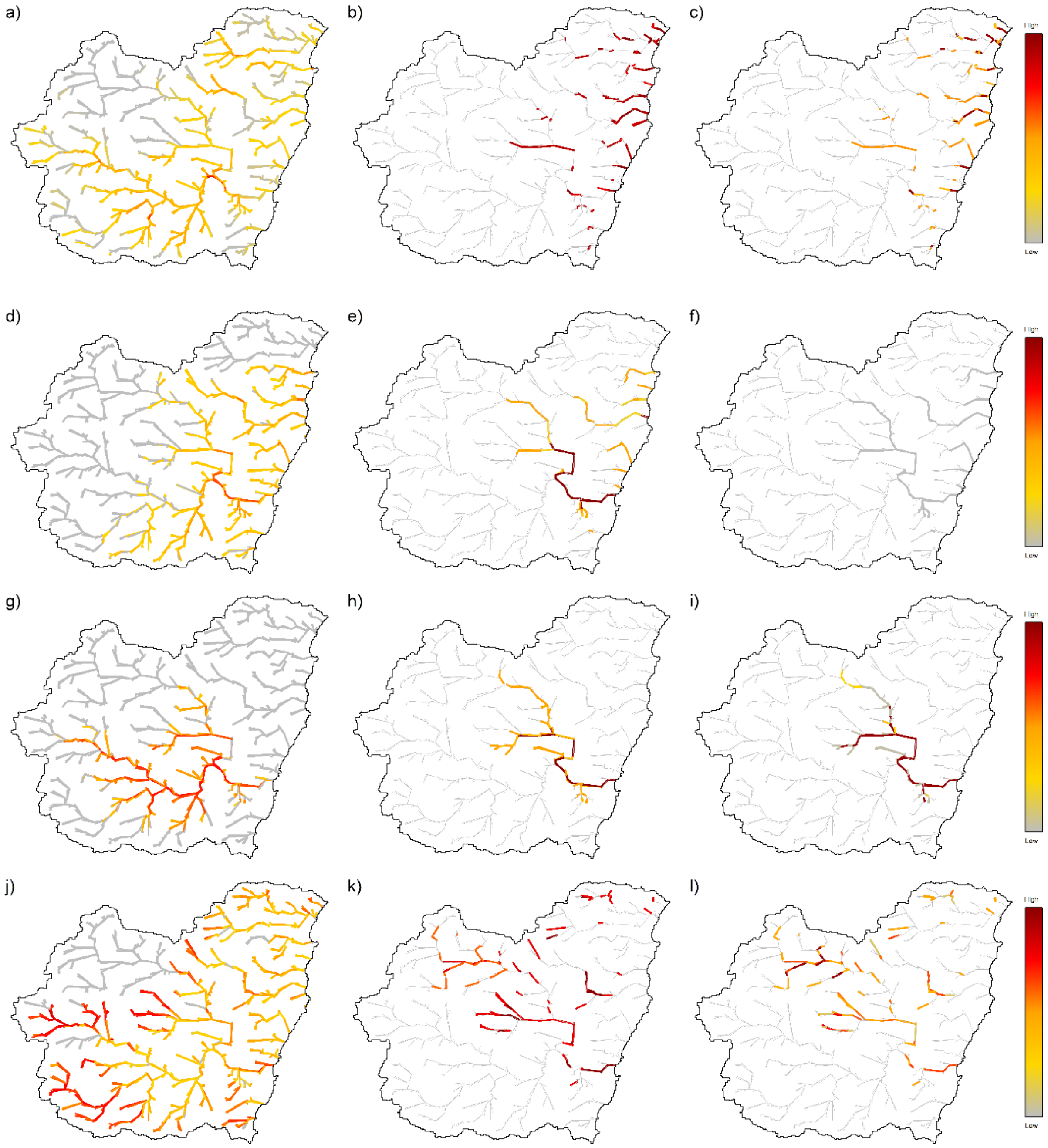

Figure 12 Intensity of threats within the Bega basin from; a) land-use, d) exotic group 1, g) exotic group 2, and j) exotic group 3. The second column shows the in red the most effective reaches in which to act under the current climate (b, e, h and k), and the third column shows the rank of standard deviation between current and future management priorities. Note panel f is grey because there was no difference between the relative priorities of management for exotic group 1 between current and future projections.

The agreement between the priorities for each management combination under current conditions, and future climate scenarios across all coastal catchments is shown in Fig.13. Where at least 20 alternatives of the same management actions were tested, the correlation coefficients (Kendall's Tau) compared the ranks of different options for each future climate scenario. If coefficients clustered close to 1, then current priorities remain high ranking in the future and, notwithstanding the possibility the gains could change, they would at least remain the most effective management action to have taken. Values closer to zero suggest no relationship between current and future priorities, and negative values would imply that the least effective alternatives under current conditions are projected to be the most effective in the future. Fig.13 shows that across multiple basins there is rarely strong agreement on the priority of actions, and although coefficients tend to be higher if larger numbers of barriers are removed, subtle differences such as whether to plan to include management of exotic species can lead to jumps or drops in the correlation of priorities. If however, the resources for a particular management action are known, but instead we prioritise within a single basin then the correlation coefficients tend to cluster much closer to 1 because the trends in changing suitability are much more likely to be common within a basin and there are also typically fewer alternatives. Most actions in medium to small basins were not considered in 20 or more locations and thus relative differences in their priority were not compared.

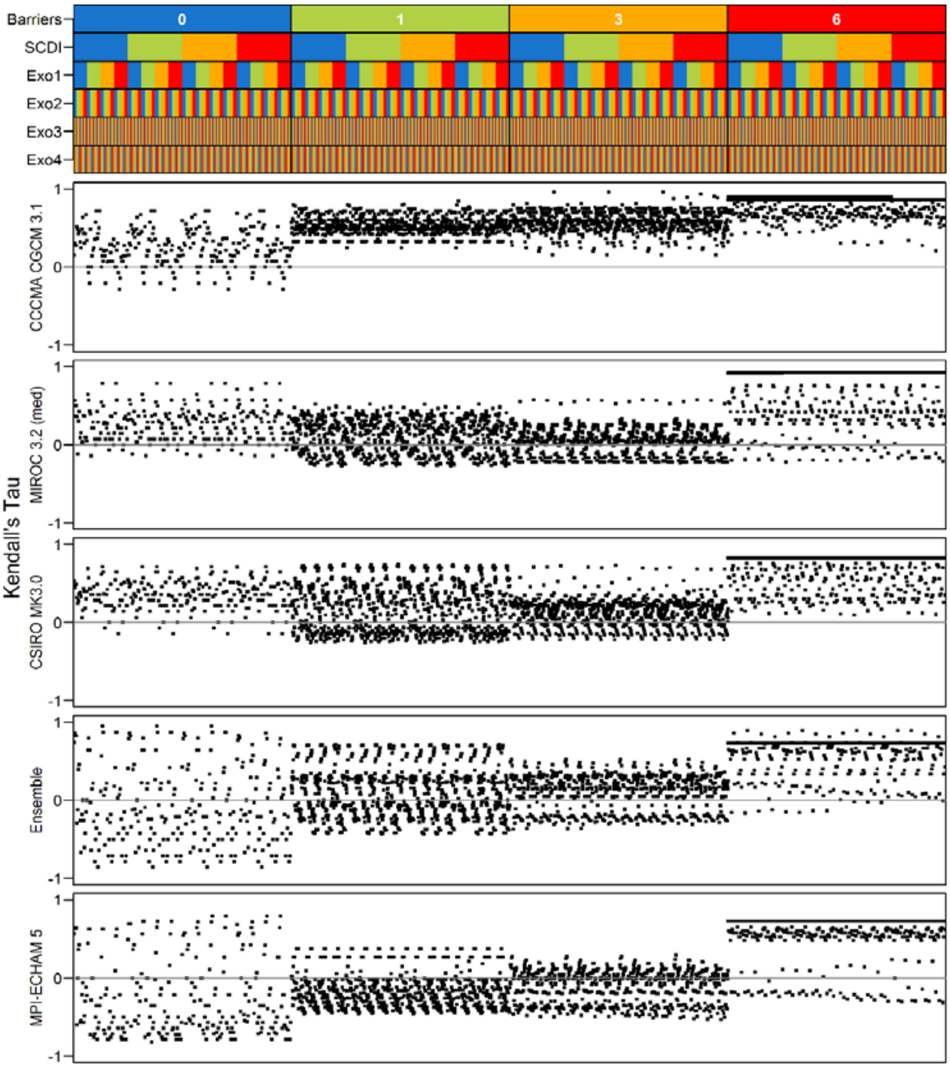

Figure 13– Plot of correlation coefficients of management priorities for the top 20 management actions between current and future climate scenarios (1 indicates complete agreement, and -1 a reversal of relative priorities). Coloured bands indicate the levels of management in barrier removal (0, 1, 3 and 6), restoration for land-use (SCDI) and exotic species control (Exo.1-4).

“No Regrets” Options for Adaptive Management

As all climate scenarios were considered equally likely, identifying what actions would be necessary to achieve a particular improvement for a species or community was difficult to define. The breadth of possibilities for each species clearly demonstrates that positive outcomes are possible under most scenarios, but likewise many actions resulted in negligible benefit. Furthermore, the priorities for which actions should be taken relative to one another could be highly variable depending on which catchment or group is targeted. Therefore, to identify which actions are most likely to result in sustained benefits for species under climate change, a precautionary approach was taken to consider the “worst-case” future scenario for any action.

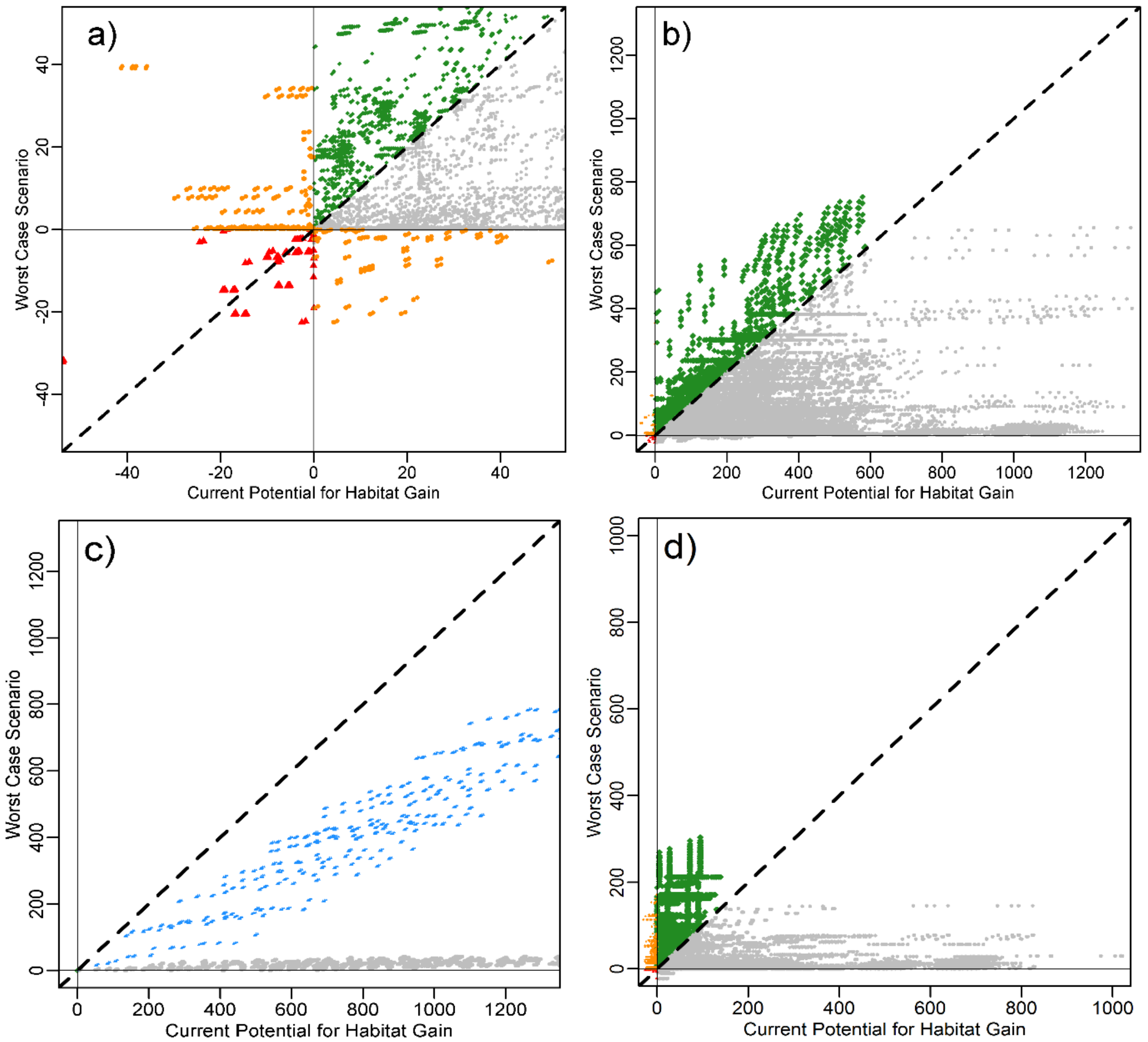

Figure 14 – The potential gains of management actions within the coastal catchments of NSW, for all native fish under current climate conditions, plotted against the “worst-case” minimum outcome from the same management action under different future climate models. Plot a) shows a closer view of the intersection of positive (green and grey) and negative outcomes (red and orange), and b) the full perspective of potential gains (refer to the basin specific reports in Appendix 10 for more detail). Plot c) shows the worst-case outcomes for the Murray-Darling basin in grey, and the best-case i.e.

maximum future gain, in blue. Plot d) shows the current potential and worst-case gains for the climate-sensitive fish subset.

The benefits of management under current conditions were considered against the minimum benefit they achieved in a future scenario i.e. “worst case” (Fig. 14). For the coastal catchments, the majority of possible actions (95%; grey points) had potentially positive gains for fish under current conditions, but those gains were projected to be reduced should particular unfavourable future climate scenarios occur. As shown for *H. compressa*, barrier removal could result in negative outcomes that may reduce the extent of suitable habitat among native species, either under current conditions or a worst-case future scenario (orange), or both (red). Although the proportion of actions that decline overall in Fig. 14 was just 0.1% of the alternatives, changes summed for all native species masked impacts on single species which were affected in 6% of options overall (i.e. cases in which a species suitable habitat declined 10 km or more). Finally but perhaps most importantly from the perspective of climate change adaptation, benefits of many actions could be sustained, or could even increase under the worst case future scenario, offering a “no-regrets” approach (4.7%: green points). Major increases across the community were possible, but were driven primarily multi-barrier removal options, allowing fish, particularly those that are diadromous to access reaches upstream (Fig. 14b). High scores were also most common in the largest catchments as these tended to have the most diverse fish fauna, and therefore more species would potentially benefit from major improvements.

Unfortunately, the projected range declines of many species in the MDB is so severe that even the best-case scenarios do not parallel the gains possible under current conditions (Fig. 14c). An absence of “no-regrets” options in the MDB does not imply fish would not benefit from management actions, but suggests that over time the potential benefits will be undermined by declining environmental suitability as the climate changes. The six coastal catchments that did not include no-regrets options; Tweed, Brunswick, Richmond, Bellinger, Hastings and Port Stephens are distinctly biased north where range contractions for most species start. However, if our focus is on only the climate-sensitive fish subset (Fig. 14d), then it is management actions in catchments in the south that fall short of current expectations (i.e. the Moruya, Snowy, Bega, Towamba and Genoa) (Appendix 8). A further concern is that only 11 of the 27 climate sensitive species receive any significant benefit from *no-regrets* actions. Of the 16 basins that did include *no-regrets* options, the associated cost for particular types of management were significantly different from those actions whose benefits could fall below the current potential. However, differences were inconsistent, and weaker still among the climate-sensitive species subset. In general, the *no-regrets* solutions required less investment than other management options. An exception to this perhaps was the Port Jackson basin in which no-regrets options on average required more barrier removal and higher costs in remediating land-use or control of exotic species in groups 1 and 2. Removal of multiple barriers was also likely to be a safe bet in the Manning and Hawkesbury basins, and acting on land-use in the Brunswick basin. Active management of exotic species, particular those in groups 3 and 4 did not appear to play a key role in ensuring long-term gains.

Discussion

Climate change is predicted to result in significant loss of suitable habitat and/or range shifts for the majority of the species modelled within eastern Australia. Suitable habitat typically follows climate isotherms to higher latitudes and altitudes, although this effect is less pronounced among the crawling and swimming dispersal-groups of taxa. There is particular concern for the long-term survival of many crayfish and frogs, for which limited dispersal capacity means many are prevented from following long-distance shifts and rely on limited altitudinal dispersal to follow suitable environments.

It is important to be aware of the range of assumptions and decisions that support species' climate change projections, and their potential to influence policy and management decisions. The wide spectrum of GCMs and added variation from emissions scenarios meant the majority of variation was inevitably dictated by alternative climate conditions, but nevertheless, changes to dispersal rates, and differences among taxa with different dispersal modes were also influential. Thus dispersal processes will be important to planning spatial objectives for managing the control or conservation of single species. In contrast, the uncertainty for individual taxa does not necessarily translate to greater uncertainty when we prioritise actions for conservation of freshwater biodiversity in general. The balance of uncertainty among climate projections (GCM and RCP), target taxonomic group, as well as dispersal rate and mode have different spatial patterns, but priorities in a number of regions show agreement (Kujala *et al.*, 2013) and these can be targeted for early action before the effects of climate change become more acute.

The uncertainty in species' individual vulnerability has implications for the longevity of management actions that may be taken to remove the burden of multiple stressors, such as those affecting native fish. Changes in habitat suitability and hence outcomes of management were highly variable among alternative future climate scenarios and often highly beneficial actions in one scenario could be reduced in another. Nonetheless, a variety of low-risk options did remain where gains appear to be robust to the variation in projected climates and allow management of current populations to also support adaptation in the future.

Dispersal Capacity and Uncertainty

The dispersal abilities of fish or invertebrates in rivers and streams is important, but also difficult to quantify. Much of the information available through the literature is based on studies and observations at small spatial scales and may not necessarily scale well with the speed at which range boundaries can shift. The probability functions were therefore used to describe the estimated range of rates species within a group could occupy, and meant that while the relative probability of long-distance dispersal was for example, considered to be higher in plants than for frogs, a species dispersal rate could match any value. This study confirms that vulnerability is a function of dispersal rate (see also Reside *et al.*, 2012), and that this is also contingent on dispersal mode i.e. aerial, over-land or in-stream. Variation in the specific spatial context of a species range means a model would not have been able to explain all the differences in vulnerability under climate change, but approximately 11% of variation was explained by dispersal rate and mode. This may be viewed as a relatively minor issue against a backdrop of uncertainty in the climate projections themselves, but under a given future climate scenario, assumptions surrounding dispersal capacity will become critical factors in assessments of which species may be able to adapt autonomously, and which species may decline.

Despite swimming taxa (fish, mussels and molluscs) having the least opportunity to track suitable climate, the crawling taxa like crayfish and frogs were predicted to be most vulnerable to climate change. Species' vulnerability under climate change was projected to increase as the expected dispersal capacity was reduced, and the effect was independent of range size and climatic exposure meaning vulnerable species in these groups must be constrained from adapting for other reasons. Among the taxonomic groups tested, frogs, and crayfish in particular represented a high proportion of what could be described as montane species, and this partially explains why altitude explained another 4% of the variation in species' vulnerability. In general, crayfish and frogs were both projected to be sensitive to climatic shifts; their suitable habitat was fragmented in the landscape, with relatively limited capacity for dispersal capacity, they were assessed as being the most threatened freshwater taxa in the region. Other taxa associated with high altitudes such as galaxiid fishes and some Odonata, were also among the species threatened by climate change, in-line with predictions made by previous studies (James *et al.*, 2013; Bush *et al.*, 2014a).

Taxa with higher dispersal ability such as insects and plants were able to occupy the majority of suitable habitats available to them and respond to environmental changes through range shifts (Hickling *et al.*, 2006; Bush *et al.*, 2013). Contrary to expectations, sensitivity was lowest among groups like fish and molluscs that were most limited in their opportunities for dispersal (Padial et al., 2014). Despite having the least capacity for dispersal, many of the molluscs modelled, in particular freshwater mussels, did not appear to be particularly sensitive to climatic change. However, this may partly reflect a bias in the species data available for modelling, and it is possible many of the more restricted short-range endemics present in NSW could face a degree of threat similar to the crayfish and frogs (Ponder & Colgan, 2002).

The limited climatic response among the swimming taxa probably reflects the generally wide tolerance of many fish species (Sternberg & Kennard, 2013). The biogeographic history of Australian fish has been subject to many periods of aridity and climate extremes, and thus the modern fauna are adapted to tolerate such conditions (Chessman, 2013), or at least able to seek refuge in the landscape (Unmack & Dowling, 2010). That this study found dispersal rate was a less significant influence on the vulnerability of fish, was likely to be at least in part, because faster rates of dispersal did not necessarily allow species to access distant suitable habitats if there was not a connected pathway. The addition of natural and artificial in-stream barriers further reduces the area of accessible suitable habitat to swimming taxa, and would at the very least increase the risk of local population extinctions (Morrongiello et al., 2011; Miles et al., 2013). In addition, while some species of fish may appear to be low risk, climate change may still combine with other threats, and local extinctions may become increasingly important as our awareness of genetically distinct populations and cryptic species in Australia increases (Faulks *et al.*, 2011; Hammer *et al.*, 2014). For example, the climatically available habitat for the 'Bellinger' sub-species of *Tandanus tandanus* (Rourke & Gilligan, 2015) was projected to be heavily reduced in most scenarios. However, *T. tandanus* (*sensu stricto*) was projected to be less sensitive owing to its wider distribution and could mean the sensitivity of the Bellinger sub-species was overestimated.

Uncertainty and Conservation Planning

The uncertainty surrounding dispersal capacity influences spatial conservation priorities because it dictates whether a reserve network should conserve a representative range of habitats and conditions nationally, regionally or even locally. Whilst the results of this study may emphasise the uncertainties and therefore perceived lack of agreement, there are many regions in which improved management could support conservation under all climate change scenarios. Across the multiple climate and emission scenarios and time steps, and across taxonomic groups there were areas of agreement in almost every basin. As might be expected, many of the areas of overlap were clustered around existing protected areas at high altitudes, such as the New England Tablelands (upper Macleay and Manning Basins), Barrington Tops, and the Australian Alpine region around Kosciusko National Park; but further action to protect freshwater habitats is most likely to complement the existing protected area network in five key regions:

1. Lower Murray and Coorong
2. Mid-Murray (border between NSW and NE Victoria)
3. Clyde and Shoalhaven Basins
4. Coastal Port Jackson, Manning and Hastings Basins
5. Lowland Richmond Basin

The regions listed above show that the conservation of freshwater fauna would in fact be best served by protecting more coastal lowlands. Suitable habitats may not necessarily occur at high elevations within a catchment and without the capacity for rapid latitudinal movement this often leaves species vulnerable to change (Bush *et al.*, 2012). The vulnerability of species in the first part of this study did not consider the array of additional anthropogenic threats to freshwater systems, which are typically heavily biased to lowlands and the coast (Turak *et al.*, 2011d). There are therefore fewer opportunities for conserving high-quality habitats in these regions, making early planning all the more vital (Turak *et al.*, 2011a).

The CBD requires that member nations set aside at least 10% of their territory in protected areas to slow the global loss of biodiversity and in 2010, this target was increased to 17% for all terrestrial and inland water areas (CBD, 2010c, a). Within the study region, 10% of the reaches by length are in sub-catchments that are at least 50% covered by protected areas and therefore priorities for a further 7%

outside reserves was considered to protect species vulnerable to climate change. The protection offered by the existing reserve network to freshwater species is a contentious issue as planning has predominantly focused on terrestrial diversity, and does not account for multiple threats other than direct habitat loss (Stein & Nevill, 2011; Turak *et al.*, 2011a). In part this is due to a lack of monitoring and survey data to robustly inform the distribution of many taxa, which was responsible for more uncertainty than climate change when ranking the importance of sites for conservation in this study (see Appendix 5). Whilst conservation planning for particular groups can help guide further work, there is growing evidence that surrogacy among freshwater taxa is low, emphasising the need for conservation managers to be aware of the resulting biases (Heino *et al.*, 2009; Darwall *et al.*, 2011). Differences in sub-catchment priorities were affected by dispersal mode as much as our uncertainty in dispersal rate. This suggests that conservation strategies could be significantly enhanced by including taxa that exhibit a variety of dispersal patterns in the planning process. Although this study demonstrates some of the basics effects of uncertainty on prioritisation, future research could introduce more systematic approaches to account for connectivity, complementarity and adequate representation of desired targets (Linke *et al.*, 2011; Bush *et al.*, 2014g).

Consider findings in the UK where natural habitats are highly fragmented and butterfly and Odonata populations have been responding rapidly to climate over the past 30-40 years. Gillingham *et al.* (2014) found that species were more abundant inside than outside protected areas (PAs) in the historical parts of their British distributions, showing that PAs have retained high conservation value. In addition, they were more abundant inside PAs in regions they have colonised showing that PAs can play a vital role in the conservation of biodiversity as species' ranges become more dynamic. Protected area coverage of high altitudes in the NSW region is relatively high and the Great Eastern Ranges (GER) initiative provides a good example of a coordinated approach that should help species track and adapt to changing climates (Mackey *et al.*, 2010). The GER is, however, predominantly focused on terrestrial conservation and much more could be done to consider the protection it would provide to freshwater biodiversity. Connections do not necessarily require formal protection of a sub-catchment, and can be improved by restoring riparian vegetation that buffers streams from adjacent land use as well as improving the quality of in-stream habitats (Capon *et al.*, 2013). Furthermore, added shading can reduce stream temperatures and if long-stretches can be revegetated, can effectively help to buffer stream temperature of sites downstream (Davies, 2010; Thomas *et al.*, 2015). Benefits from such measures will take time to have an effect, further emphasising the importance to begin taking action in regions where we have agreement across scenarios, and hence most confidence vulnerable species could persist (Bunn *et al.*, 2010).

Managing Multiple Threatening Processes

Reducing the pressure from other non-climatic threats is an often stated strategy for conservation practitioners to mitigate the impacts of climate change on a particular species or ecosystem (Heller & Zavaleta, 2009). This study focused on for four major threatening processes affecting freshwater fish in NSW, testing the potential for management to recover suitable habitat and the congruence between priorities under current and future climate conditions. Within the study region, these primary threats to fish were widespread and likely to be limiting the suitability of habitats across large parts of their potential range. There is therefore a great deal of potential for management of non-climatic threats, or limitations on movement imposed by artificial barriers, to improve long-term habitat suitability for multiple species in a catchment.

Under the current climate, the majority of management actions considered could improve the extent of suitable habitat, although in the long-term, their effectiveness for particular species became highly variable under a range of possible climate change scenarios. Without consensus on which future climate model is likely to be most accurate, establishing quantitative estimates of the possible returns for investing in management actions would have come with enormous uncertainty. Given the highly variable nature of habitat gains for many management actions in the future, relative priority of equivalent actions was compared between current and future climate scenarios. Overall, the priority of different actions was fairly consistent within the MDB, but could be much more variable across the coastal catchments. This was more apparent when management priorities were determined on the basis of only climate sensitive species. This challenges the assumption that management of non-climatic threats should simply continue to follow business-as-usual, because in the long-term those efforts may be wasted because they do not necessarily help mitigate the effects of climate change.

Fortunately a large number of actions were expected to remain as effective in the future as they would under the current climate, regardless of which future scenario was used, and the gains made now may potentially increase under climate change. The *no-regrets* actions are specific to particular locations, but occur in a number of catchments, cover all species to some degree, and are not reliant on particular management actions being taken. Even relatively small and isolated actions within the control of local authorities can therefore be shown to reduce a species' vulnerability to climate change. In addition, although *no-regrets* actions are seemingly risk-free because their gains persist under all future scenarios, by being less conservative and risk-averse, much high gains may be possible. Sophisticated tools and processes are being developed to identify combinations of areas that can balance the potential habitat gains for the current and future against the risks to the species under climate change as a whole, and in relation to adaptive management (Joseph *et al.*, 2009). Naturally, implementing any major management program will consider the perceived ratio of risks and benefits, and other barriers to adoption associated with financial constraints, human water security, and social acceptability (Postel *et al.*, 1996; Moser & Ekstrom, 2010). Whilst catchment management nonetheless requires consultation with a range of stakeholders, this study has shown that many opportunities exist to improve the extent of suitable habitat for threatened fish species. The uncertainty regarding future climates should neither delay action, nor mean that actions be taken without giving some consideration to their value in the future.

Modelling species' responses to threats in addition to climate change is often limited by our understanding of their ecology, and mapping suitable surrogates for the stressors, but consistently indicates species affected by multiple stressors have an elevated risk of extinction (Mantyka-Pringle *et al.*, 2011). When predicting shifts in species' distributions under climate change, model projections assume species' will occupy environmentally suitable habitats. Depending on the species' estimated response to flow-modification, intensity of land-use, and overlaps with exotic taxa, this study estimates that remaining suitable habitat was on average less than half its original potential, and as much as 96% smaller in the case of *Galaxias fuscus*, and 98% for *Gadopsis bispinosus*. The degree to which the non-climatic threats, in particular barriers, affect fish distributions beyond the study limits was unknown, but the landscapes of Victoria and coastal Queensland are also highly transformed, and subject to similar threats as NSW (Drew *et al.*, 2008). Therefore, whilst data were not available to quantify changes in species' vulnerability to climate change for their whole range, the addition of non-climatic threats would reduce the viability of populations in many environmentally suitable areas, leading to further fragmentation, and increasing the risk of local extinction as environmental conditions also begin changing. The range of *Potamolosa richmondia* was one of the few species whose range fell almost entirely within the study area, and non-climatic threats (all except *G. holbrooki*) were predicted to reduce its current suitable range by approximately 14%, of which up to 4.6% could be recovered according to the management actions considered. However, after accounting for connectivity, *P. richmondia* was projected to decline across >60% of its range under all future climate scenarios by 2085. Far from devaluing the contribution management of non-climatic stressors could make to the capacity of a species to adapt, these predictions emphasise the importance of management of non-climatic stressors for species persistence in the landscape. Some reaches and waterbodies may also provide natural refugia because they are decoupled from regional climate changes at fine spatial and temporal scales (Davis *et al.*, 2013). A relatively risk-free option to facilitate climate adaptation would be to construct or modify waterbodies to act as refugia, and complement existing landscape features, particularly where connectivity constraints prevent species from moving to and from other refugia (Robson *et al.*, 2013). More caution may be required before attempting species translocation, but some native fish are already bred in captivity (mostly for commercial purposes) and released so there is a greater understanding of the risk involved. Even diadromous species that have marine life-stages may benefit from this assistance. The *no-regrets* management combinations suggest interventions such as exclusion of exotic species did not need to be spatially extensive. However, some strategies such as riparian restoration, which could both mitigate the impact of human land-use and moderate increases in water temperature (Capon *et al.*, 2013), may have to be extended long distances upstream to accrue the needed improvements (Seavy *et al.*, 2009). Finally, within a formal prioritisation framework, the benefits of management should consider freshwater taxa other than fish. Many frogs for instance are threatened by *G. holbrooki* (Rowe *et al.*, 2008), and modified flow regimes affect freshwater mussels too (Jones & Byrne, 2014). Stream invertebrate composition is monitored to detect diffuse pressures on the ecosystem, but the appropriate means to engineer recovery and improvement is still in its infancy (Turak *et al.*, 2011d).

This study has shown that significant increases in available habitat for fish in NSW will often require improving connectivity across in-stream barriers, either singly or in chains. In addition to preventing upstream movement, dams can alter flow regimes, sediment transport, water temperature, dissolved oxygen and transform the upstream habitat to a more lentic environment that affects fish communities upstream as well as downstream (Gardner *et al.*, 2013). Whilst estimated flow-modification was included as a threat, restoring natural flow regimes via dam removal was not considered among management options because no further information specific to the dams or upstream water storages was available meaning the inferred benefits of any remediation would have been highly uncertain (Hoenke *et al.*, 2014). To keep the barrier functioning, many technical and semi-natural bypass systems are now available that will allow connectivity. These fishways have to be customized to target species, the type and dimensions of the water body, the type of barrier, and the trade-off with the demands of socio-economic function the barrier supports (Thorncraft & Harris, 2000). Although the wider benefits of restored flow may not always be realised (Lamouroux & Olivier, 2015), improving connectivity using fishways could allow species to recolonise habitats from which they had become excluded (Miles *et al.*, 2013), shift their distributions as climate changes (Comte *et al.*, 2012), and move to refugia during harsh climatic extremes (Leigh *et al.*, 2014). Migration of potamodromous species is particularly important when habitats are too small to sustain viable populations and when key-life cycle events are obstructed, such as seasonal reproduction migrations (Miles *et al.*, 2013). Understanding habitat requirements for all life stages is important to define the minimum limit of distribution necessary for a viable population (Brevé *et al.*, 2014).

Restoring connectivity could also produce perverse negative outcomes. A large number of combinations, many more in fact than the no-regrets combinations, resulted in a reduction in suitable habitat for native species when barrier removal allowed exotic species to access upstream habitats. The threat posed by introduced salmonids to most of the galaxiids is so severe that a number of species are not found below particular dams, including six species too rare to be modelled in this study (Raadik, 2013). There are circumstances when dam removal offers an ecologically effective and potentially cost-effective method of restoration (Pittock & Hartmann, 2011). Marks *et al.* (2010) present a rare example in which flow-restoration and management of exotic species could be compared and found eradication of several exotic species was far more cost effective at increasing native fish abundance than returning natural flow. Likewise, examples exist where the introduction of environmental flows to wetlands can benefit native species, but may also promote introduced species (Rayner *et al.*, 2015). This analysis demonstrates that management options to restore connectivity and/or natural flow regimes in particular catchments, may not be undertaken without including other actions to control exotic species (Rahel, 2013).

Assumptions and Further Uncertainties

Climate: Inevitably, one of the biggest sources of variation in predicting species future distributions is the variation among general circulation models (GCMs) in how the Australian climate is expected to change (Hobday & Lough, 2011). Multiple GCMs representing the breadth of possible future conditions were chosen because to ignore such uncertainty may lead to over-confidence when interpreting future projected distributions, with consequences for conservation planning. GCMs represent a range of plausible futures and the IPCC avoids ranking models, and treats each equally (IPCC-TGICA, 2007; Harris *et al.*, 2014). There is some evidence that averages of certain models can perform better for south-east Australia than single GCMs (Fordham *et al.*, 2011; Fordham *et al.*, 2012), which is why we considered the 'Ensemble' GCM in our analysis. Although the inclusion of physical chemical and biological processes in GCMs is highly advanced, projections of future climate remain uncertain, particularly in their spatial and temporal distributions of rainfall and hydrological variables (Mehrotra *et al.*, 2014). Natural cycles of climate variation are also not yet well accounted for because GCMs start from randomly selected pre-industrial states, although it is thought some aspects of near term and decadal variability could be introduced by training models on recent climate time-series (Meehl *et al.*, 2010). There is also a reliance on climate means, whereas we may expect freshwater systems to respond most strongly to climate extremes (Leigh *et al.*, 2014). For Australia, the timing of rainfall and how this then translates to runoff is particularly uncertain, but there is no evidence any particular model is likely to be better at predicting all drivers of change (Chiew *et al.*, 2009; Chiew *et al.*, 2010), although some are certainly worse (Kirono & Kent, 2011). Thus to some extent we must simply live with the uncertainty in climate projections and consider the breadth of possible outcomes (Kirono *et al.*, 2011).

Species Distribution Modelling: The methodological uncertainties surrounding SDMs are part of the rationale for this study (Wiens *et al.*, 2009). If required, models could be improved with a greater numbers of species records (Comte *et al.*, 2012), or more detailed environmental data (Storlie *et al.*, 2013). Correlative modelling techniques like the ones used in this study, using presence-only data, are at risk of over-estimating suitable habitat extent and including errors of commission because the models assume that all suitable climate space is occupied (Pineda & Lobo, 2012). Statistical methods were used to tailor the selection of predictor variables for each species and hence avoid over-fitting, and these were fit to a selection of background points appropriate to the species dispersal potential. Finally, the accuracy with which the current distribution is represented by an SDM is an important consideration, but there is no guarantee that environmental relationships that hold true now will remain so in the future. Consequently, a conservative approach was taken and extrapolation to novel environments was limited (Elith *et al.*, 2010), thereby assuming species are unlikely to persist if the climate shifts outside the range of conditions currently experienced within the region (Elith & Leathwick, 2009).

Dispersal Capacity: Whilst the purpose of this study was to demonstrate that dispersal is important and can be accounted for in vulnerability assessment and conservation planning, there are still many uncertainties associated with predicting the rate at which freshwater species can respond to climatic shifts (Angert *et al.*, 2011). Stream connectivity can be viewed as a property of the landscape (Chaput-Bardy *et al.*, 2009), but this hinges on a species' dispersal capacity, which is a complex function of emigration rates, population density dependence, settlement success, and habitat availability (Baguette *et al.*, 2013; Bocedi *et al.*, 2014). In some cases, if species are able to take advantage of chance long-distance dispersal events, then their rate of dispersal may be underestimated by the group likelihood used in this study. For example models of plant dispersal evolution favoured the use of power-law curves to predict a non-zero probability of propagules dispersing over very long distances in natural landscapes (Hovestadt *et al.*, 2001). Power laws have also shown promise in describing the super-diffusive spread of invasive freshwater plants in Ireland (Kelly *et al.*, 2014), where human activity and animals allow regular "jump-dispersal" events. Taxa such as fish and macroinvertebrates that have evolved in regions with unpredictable habitats are also highly dispersive, ensuring they can locate suitable conditions and persist at least somewhere in the landscape (Warfe *et al.*, 2013).

It is important to understand the dispersal rates used in this study were meant to reflect average movement that may be sustained by a broad range of species in a group, and not the maximum observed distances travelled by single species (Reynolds, 1983; Flenner & Sahlén, 2008). This study assumes that species' populations could maintain a continuous rate of dispersal with no significant delays, despite an understanding that dispersal can be strongly influenced by properties of the intervening matrix (Vogt *et al.*, 2009). Extra barriers or costs to movement were not included for most taxa because for the broad suite of taxa used this would have required further assumptions and would have been difficult to define from the available data on landscape features (Svenning *et al.*, 2014).

Due to the difficulties in determining dispersal rates directly (Macneale *et al.*, 2005), its influence is more often inferred indirectly from analysis of community spatial structure (Grönroos *et al.*, 2013), or from genetic studies of particular species (Hughes *et al.*, 2013). There is a risk that inferences based on survey observations are biased towards common or easily recorded taxa and the mean rate of dispersal may be overestimated (Jacobson & Peres-Neto, 2010). Nonetheless, multiple lines of evidence suggest aquatic insects do fly beyond streams to achieve relatively widespread dispersal (Bohonak & Jenkins, 2003; Alexander *et al.*, 2011) and have done so in response to climate change (Hickling *et al.*, 2005; Hickling *et al.*, 2006). Genetic studies also confirm that animals with flying adults tend to show lower genetic differentiation than species with life cycles restricted to the stream channels (Hughes *et al.*, 2009). Among fish, it is believed that marine larval and juvenile phases probably facilitate extensive gene flow among coastal rivers, particular if the larvae move into the sea rather than remain in estuaries (e.g. Roman Nose Goby

Awaous acritosus (Huey *et al.*, 2014), and Australian grayling *Prototroctes maraena (Schmidt et al., 2011)*). These studies suggest some species may be able to disperse to new basins in the future if larvae are being exchanged among basins. However, in some cases, like *Retropinna semoni*, chemical differences and genetic structuring were very high, because larvae do not appear to move beyond the river mouth (Woods *et al.*, 2010; Hughes *et al.*, 2014). In such cases, the species dispersal options are limited to their own catchments like other non-diadromous species. Crayfish were thought to be capable of terrestrial dispersal, but phylogeographic research looking at crustaceans in eastern Australia (Ponniah & Hughes, 2004; Hurry *et al.*, 2014) have found no significant differences compared to obligate freshwater species, i.e. fish and molluscs, suggesting overland movements have not been a common process (Hughes *et al.*, 2013). Overcoming dispersal limitation by translocation of vulnerable crayfish faces difficulties due to the strong hierarchy of competitive interactions among species (McCormack, 2012), particular if the introduced species are dominant (McCormack, 2014), and building *in-situ* resilience, artificially if necessary, may be more effective (Robson *et al.*, 2013).

Management of Non-Climatic Threats: The difference in scale between a catchment management authority’s influence on their landscape and the global threat of climate change means many practitioners are understandably unsure how best to promote adaptation (Heller & Zavaleta, 2009). Whilst the management of non-climatic stressors to offset anticipated impacts of climate change may be an appealing alternative, caution must clearly be exercised in any expectation of a cause and effect outcome (Tonkin *et al.*, 2014). This study focused on how best to combine multiple management actions and assumed a conservative view that each stressor acted independently of each other which reduces the potential benefits. More detailed information on the synergy between stressors would allow the cumulative impacts of stressors to be better represented (Brown *et al.*, 2014). Restoration of land-use or management of exotic species was prioritised according to where species occurred, once barriers had been considered, and therefore the effect of restoration for many reaches was not considered. Without projections of how land-use is expected to change, the study also assumes that its impact did not vary unless a reach was targeted for restoration. Although environmental improvements may take time to return, restoration of the riparian zone can achieve a wide array of benefits that serve both under current and future climates (Capon *et al.*, 2013). Likewise, the relative impairment of reaches due to flow modification was assumed to remain consistent over time, but evidence now suggests that given enough time restored flow regimes can improve river condition for particular species (Lamouroux & Olivier, 2015). Ultimately, trade-offs on the basis of cost will mean priorities will vary from those presented, but their long-term effectiveness can still be considered in the same way.

Although the NSW Department for Primary Industries has greatly improved the data available to review the location of in-stream barriers, many others may exist that are undocumented (Thorncraft & Harris, 2000; NSW Department of Primary Industries, 2006c). For example, in the near future, higher resolution digital elevation models are likely to help identify many more potential waterfalls that could prevent access to headwaters. In addition, a study by Januchowski-Hartley *et al.* (2014) found that the passability of road culverts by fish was related to natural gradients in topography and stream size, but while the probability of any particular culvert being passable was high, the huge number of culverts in a basin meant that, together, they could pose a greater challenge to migratory fish than dams. Within northern NSW, around 20% of road-crossings were estimated to obstruct fish movement depending on differences in head-loss and water velocity (NSW Department of Primary Industries, 2006a). The study region contained over three thousand road-crossings, but ultimately they were omitted from the barrier dataset because the majority were clearly not restricting movement. Without further information on barrier types, this study also assumed management actions at any location are equivalent when clearly enabling connectivity across large dams presents challenges (Gehrke *et al.*, 2001). Thus completing spatial datasets on barrier locations, design and size that are compatible with other hydrological spatial layrers remain a priority so that systematic cost-benefit analyses can inform which barriers should be removed or have fishways installed (O'Hanley *et al.*, 2013).

Conclusions

Globally, freshwater ecosystems are believed to be suffering due to the combination of multiple stressors (Dudgeon *et al.*, 2006; Ormerod *et al.*, 2010), and whilst short-term conservation management remains essential to slowing the pace of biodiversity loss, preparation for long-term sustainability under future conditions is becoming an increasing priority (Ormerod, 2009; Vörösmarty *et al.*, 2010). However, uncertainty regarding where and how to improve conservation of freshwater biodiversity under current conditions is magnified when considering how ecosystems may respond under climate change (Ormerod, 2009). Climate change adaptation is defined as the adjustment of natural or anthropogenic systems to a changing climate for the purpose of moderating impacts or capitalizing on novel opportunities (IPCC, 2007b). Groves *et al.* (2012) proposed five approaches to climate change adaptation that can be integrated into existing or new biodiversity conservation plans. These included: 1) conserving the geophysical stage; 2) protecting climatic refugia; 3) enhancing regional connectivity; 4) sustaining ecosystem process and function; and 5) capitalizing on opportunities emerging in response to climate change. A major strength of these approaches is that they are generally robust to uncertainty of how climate impacts may manifest in any given location (Game *et al.*, 2011; Kujala *et al.*, 2013; Wright *et al.*, 2015). Results for Part 1 of this project strongly support the second and third strategies. Whilst a highly significant ecological process, our uncertainty regarding species rates of dispersal has relatively little impact when considering how conservation efforts are distributed across the landscape. This is not to say that all the freshwater species modelled were expected to track suitable climates, hence the importance of identifying or even creating refugia for highly vulnerable species like crayfish and frogs (Robson *et al.*, 2011; Robson *et al.*, 2013). Overall, allowing for moderate estimates of dispersal and balancing priorities for taxa from many groups against multiple sources of uncertainties, conservation priorities were most affected by the direction of climate shifts i.e. GCM scenarios.

Whilst many sources of uncertainty exist when predicting the impacts of climate change, conservation planning should make decisions in lieu of the anticipated effects of climate change (Carvalho *et al.*, 2011; Bush *et al.*, 2014g). Given the uncertainty surrounding predictions of climate change, some studies have argued that as much land as possible be set aside in reserves to minimize the distances between new and existing reserves and create porous landscapes (Williams *et al.*, 2005). Whilst reserve systems will remain at the core of conservation approaches for the foreseeable future, many also acknowledge that climate change will exacerbate existing tensions and trade-offs between protecting areas and meeting basic human needs (Heller & Zavaleta, 2009). Systematic approaches can identify new reserves that maximize their effectiveness in the face of climate change (Kujala *et al.*, 2013) but equally, management outside of reserves (the matrix) is often important to preserving quality of aquatic habitat within reserves, and formal protected area status does not necessarily insulate species from threats such as introduced species (Hermoso *et al.*, 2011; Stein & Nevill, 2011). Davis *et al.* (2013) distinguish between aquatic habitats in the arid-zone that are likely to act as climatic refugia (evolutionary refugia) because of decoupling from the regional climate, in addition to others that are important to enhancing regional connectivity or sustaining ecosystem processes and functions. Thus while on the one hand formal protection for any focal sites that fall outside the reserve system is a priority, equally important will be the development of policies that allow the natural functioning of those sites to be supported by actions beyond reserve boundaries.

Even with appropriate policy changes and funding to improve management, it can take decades before policy changes result in visible and widespread improvements. It is difficult to achieve improvements in fragmented landscapes and where additional stressors are not addressed (Armin *et al.*, 2013). Given the wide range of alternative future climates, and the likely recurrence of harsh drought conditions during climate cycles in the coming decades it is difficult to identify common strategies to promote range gains for threatened or climate-sensitive species. Nonetheless, there are instances where catchments have been successfully restored, and their species and habitats are likely to prove more resilient to climatic changes (Floury *et al.*, 2012; Vaughan & Ormerod, 2012). There are of course risks to consider, but delaying action to save species will only make it harder should populations decline, and in the end must be balanced against the high predicted risk of extinction in the long-term. While policymakers are right to recognise the variation between projections of future projected impacts creates a challenging environment, actions that reverse biodiversity declines with the minimum associated uncertainty are available.

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Supplementary Information

Appendix 1 - Sources of species' occurrence data.

Source	Fish	Frogs ∞	Crayfish	Odonata Δ	Hemiptera	Macrophytes	Molluscs	Platypus
New South Wales Department of Primary Industries †	●				●			
New South Wales Office of Environment and Heritage ‡				●		●	●	
Victorian Environmental Protection Authority	●		●	●	●			
Murray Darling Basin Authority (Sustainable Rivers Audit)	●							
South-East Queensland Ecological Health Monitoring Plan	●							
Australian Museum	●	●		●	●		●	●
Queensland Museum	●	●		●	●		●	●
Victorian Biodiversity Atlas	●						●	
Atlas of Living Australia	●		●			●	●	●
Australian Crayfish Project Ω			●					
Australian National Insect Collection (CSIRO) ◊				●	●			
Queen Victoria Art Museum and Gallery				●	●			
Australian Plant Pest Database				●	●			
South Australian Museum				●	●			
Tasmania Museum and Art Gallery				●	●			
Museum and Art Gallery of the Northern Territory				●	●			

† - Courtesy of Dean Gilligan, Fisheries and Ecosystems Research, NSW DPI.

‡ - Freshwater mussel records verified courtesy of Hugh Jones (NSW OEH).

Ω - Courtesy of Rob McCormack, Australian Aquatic Biological.

∞ - Database provided by Dan Rosauer (***reference***).

Δ - Database collated as part of Alex Bush's PhD research (see Bush et al. 2014a).

◊ - Courtesy of Tom Weir, Australian National Insect Collection, CSIRO.

Appendix 2 –Species Distribution Model Evaluation Scores

Appendix 3 – Species Distribution Model Predictor Variables

Appendix 4 – Species Projection Summary Reports

Appendix 5 – Prioritisation of Different Freshwater Groups

It is worth noting the influence of taxonomic groups on selection of conservation priorities. Priorities were inevitably affected by focusing on different groups because patterns of species distributions patterns of species distributions among them were not even (Table A5.1). The overlap among the groups did not follow any discernible pattern based on similar ecology or dispersal mode, but could theoretically reflect similarities in patterns of biogeographic history, or perhaps simply be the results of similar bias in recording efforts. The agreement between groups for example was moderately high based on the Ensemble climate model (RCP 8.5 and 2085) whereas priorities for groups like mussels and frogs are distributed very differently using MIROC 3.2med. Thus GCM has an overarching influence on conservation planning, but this also emphasises the importance of using multiple freshwater taxa to create a representative conservation plan (Darwall et al. 2011).

Table A5.1 Percent overlap of priority conservation reaches between taxonomic groups for two GCMs (Ensemble and CCSR MIROC 3.2med). Species distributions were projected under emissions scenario RCP 8.5 to the year 2085 and scores were discounted to reduce dispersal rate uncertainty. The combined score was summed across taxonomic groups.

	ENSEMBLE									CCSR MIROC 3.2med								
	Mollusc	Mussel	Crayfish	Frogs	Platypus	Plant	Fish	Odonata	Hemiptera	Mollusc	Mussel	Crayfish	Frogs	Platypus	Plant	Fish	Odonata	Hemiptera
Mussel	54									9								
Crayfish	27	11								19	3							
Frogs	28	13	14							3	1	3						
Platypus	43	19	43	24						20	25	11	0					
Plant	37	32	41	24	44					54	16	24	2	26				
Fish	31	22	34	21	47	36				23	8	20	1	18	22			
Odonata	40	18	36	35	63	31	38			52	6	23	2	11	36	18		

Hemiptera	48	23	32	24	36	63	32	28		38	4	28	4	17	50	11	26	
Combined	54	24	38	47	51	66	40	48	66	60	6	26	13	20	63	18	51	62

The difference between priorities before and after discounting reach scores given the uncertainty in dispersal rates was also dependent on which group reaches were ranked. The correlation of reach scores before and after discounting for dispersal rate uncertainty was high (>0.95), suggesting that possible refugia for many climate sensitive taxa were close to their observed occurrences (Table 2). However, reducing the value of reaches that assumed higher rates of dispersal made a particularly big difference to which reaches were important for conserving crayfish and mussels. Circumstances in which there was low overlap before and after discounting was applied, despite high correlation coefficients, reflected agreement in the rank of large numbers of low scoring reaches, whereas the complement of reaches within the top 7% were more variable.

Table 2. Percent overlap of priority conservation reaches (in bold) and correlation coefficients of reach scores (in brackets) under five climate scenarios (based on RCP 8.5 in 2085), between scores with and without discounting for dispersal uncertainty.

	Ensemble	CCCMA CGCM 3.1	CCSR MIROC 32med	CSIRO MK3.0	MPI ECHAM5
Mollusc	**71** (0.90)	**82** (0.93)	**80** (0.86)	**71** (0.94)	**67** (0.90)
Mussel	**71** (0.84)	**38** (0.20)	**28** (0.12)	**43** (0.31)	**69** (0.83)
Crayfish	**52** (0.87)	**30** (0.48)	**37** (0.72)	**61** (0.87)	**37** (0.84)
Frogs	**82** (0.98)	**55** (0.95)	**75** (0.97)	**75** (0.98)	**67** (0.95)
Platypus	**90** (0.99)	**71** (0.99)	**33** (0.93)	**52** (0.99)	**97** (0.99)
Plants	**90** (0.99)	**91** (0.98)	**85** (0.96)	**86** (0.99)	**91** (0.99)
Fish	**83** (0.96)	**74** (0.93)	**80** (0.87)	**81** (0.96)	**77** (0.95)
Odonata	**77** (0.98)	**86** (0.99)	**85** (0.97)	**84** (0.99)	**83** (0.99)
Hemiptera	**94** (0.99)	**92** (0.99)	**91** (0.99)	**89** (0.99)	**95** (0.99)
All taxa	**93** (0.99)	**95** (0.99)	**92** (0.99)	**94** (0.99)	**94** (0.99)

Finally, the agreement between different taxonomic groups on which sub-catchments outside existing protected areas are most important for conservation, shows clear spatial differences among the five GCM projections. Figures A5.1-A5.5 display the degree of overlap between taxonomic groups (maximum is therefore 9) under each GCM across the coastal catchments of NSW and the MDB. The 10% of sub-catchments that more than covered 50% by existing protected areas are scored as 10 (black).

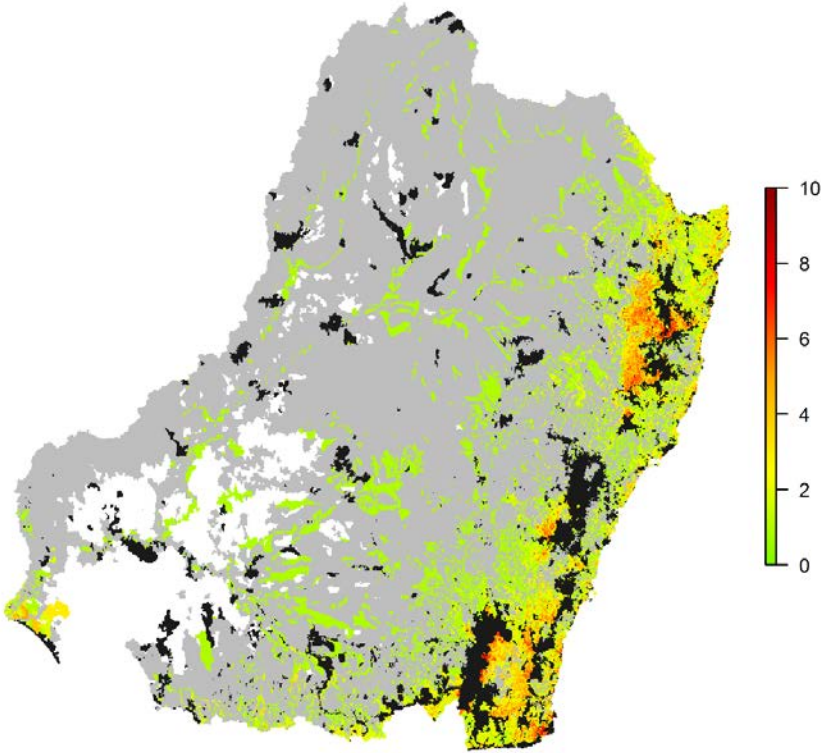

Figure A5.1 Overlap of conservation priorities between nine groups of freshwater taxa to protect 7% of streams outside existing protected areas (10; black). Scores are weighted according to species sensitivity to range shifts using the CCCMA CGCM 3.1 model (RCP 8.5, 2085).

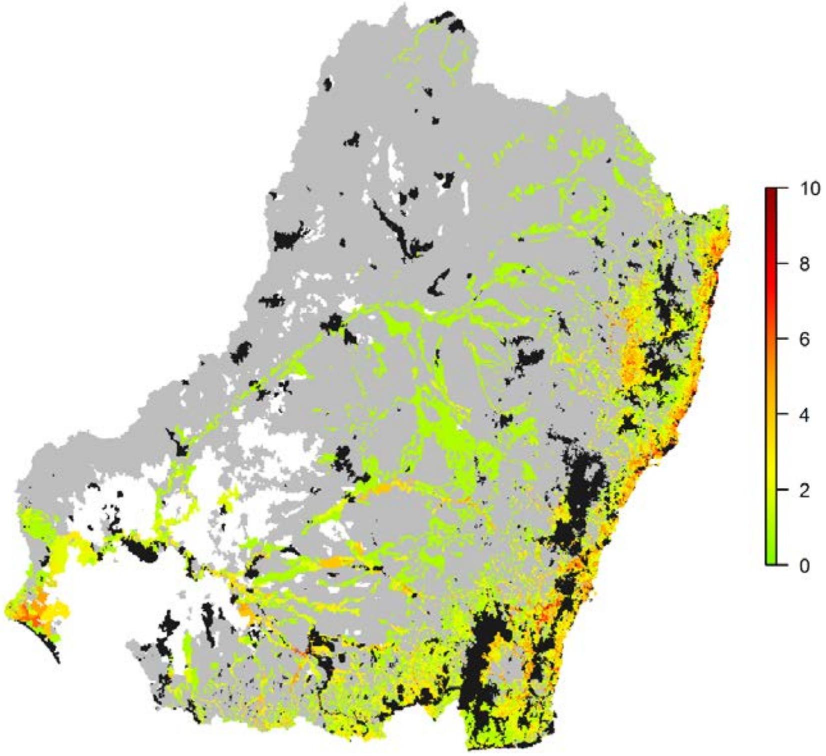

Figure A5.2 Overlap of conservation priorities between nine groups of freshwater taxa to protect 7% of streams outside existing protected areas (10; black). Scores are weighted according to species sensitivity to range shifts using the CSIRO MK 3.0 model (RCP 8.5, 2085).

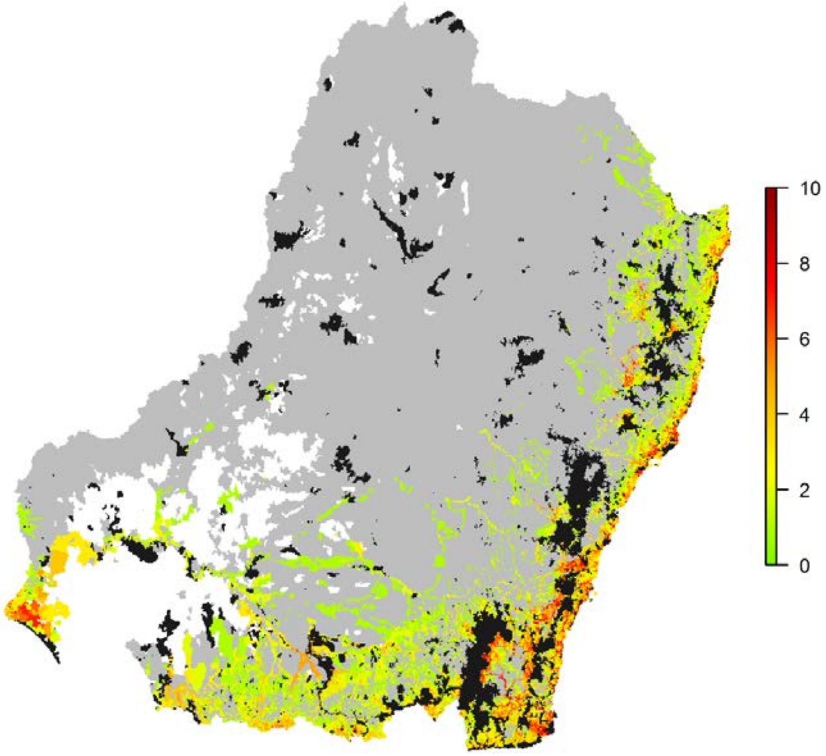

Figure A5.3 Overlap of conservation priorities between nine groups of freshwater taxa to protect 7% of streams outside existing protected areas (10; black). Scores are weighted according to species sensitivity to range shifts using the MPI-ECHAM 5 model (RCP 8.5, 2085).

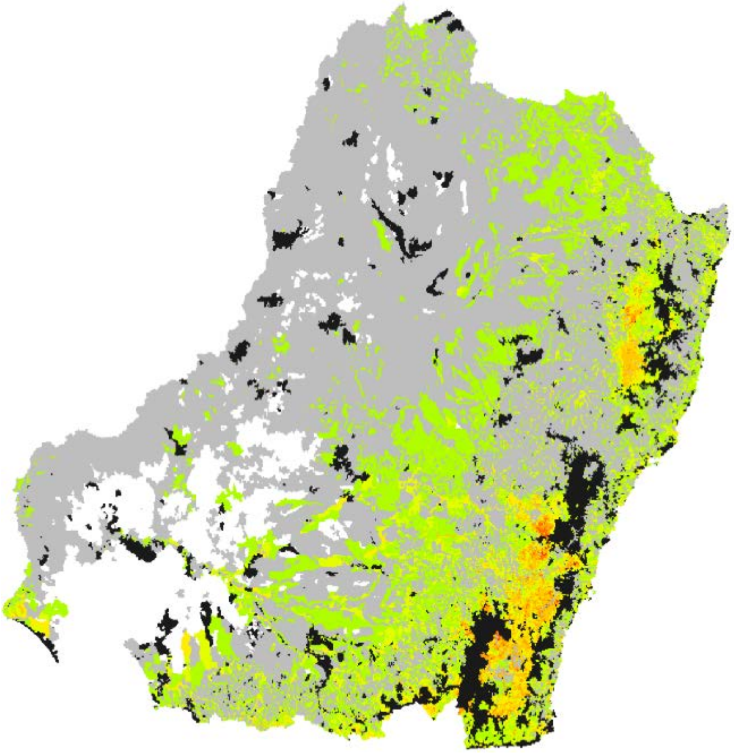

Figure A5.4 Overlap of conservation priorities between nine groups of freshwater taxa to protect 7% of streams outside existing protected areas (10; black). Scores are weighted according to species sensitivity to range shifts using the MIROC 3.2 med model (RCP 8.5, 2085).

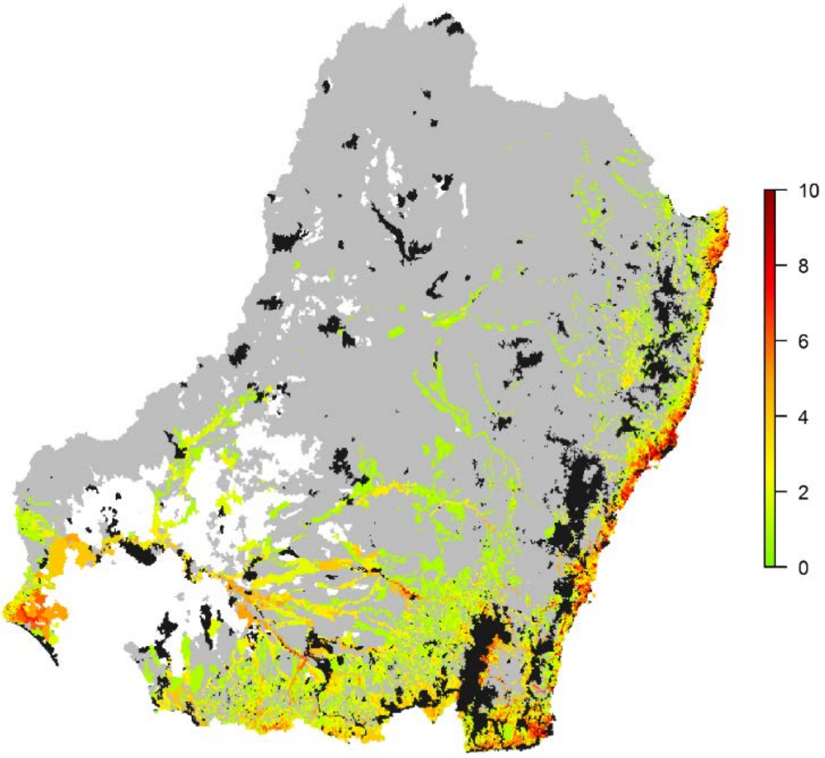

Figure A5.5 Overlap of conservation priorities between nine groups of freshwater taxa to protect 7% of streams outside existing protected areas (10; black). Scores are weighted according to species sensitivity to range shifts using the Ensemble climate model (RCP 8.5, 2085).

Appendix 6 – Fish Threat Constraint Table

Appendix 7 – Impact of Multiple Non-Climatic Stressors

Table 7.1 – Estimated percent reduction to fish species current range (measured as km of suitable habitat) as a result of seven non-climatic stressors, and all combined.

Species	Range	Dams	FRDI	SCDI	Exo1	Exo2	Exo3	Exo4	All
Ambassis agassizii	34293.2	44	3	1	6	28	0	5	62
Anguilla australis	25521.7	0	0	0	0	0	0	0	0
Anguilla reinhardtii	79056.7	0	0	0	0	0	0	0	0
Bidyanus bidyanus	21352	39	29	3	20	11	0	22	67
Craterocephalus amniculus	8152.8	23	3	3	0	36	0	0	51
Craterocephalus fulvus	34809.2	37	13	3	0	34	0	0	59
Craterocephalus marjoriae	6540.6	2	2	2	0	71	0	0	72
Gadopsis bispinosus	13233.3	12	4	5	3	5	98	13	98
Gadopsis marmoratus	29797.8	20	6	7	0	0	0	0	30
Galaxias arcanus	9030.6	51	2	1	0	0	0	0	52
Galaxias brevipinnis	19725.4	0	2	1	0	4	65	4	67
Galaxias fuscus	1924.7	20	1	13	0	0	93	0	96
Galaxias maculatus	13323.8	15	5	1	0	13	39	2	61
Galaxias olidus	86240	10	2	3	0	0	63	14	72
Galaxias oliros	26348.6	34	6	6	0	0	15	8	51
Galaxias rostratus	18045.9	66	28	5	24	16	0	15	82
Gobiomorphus australis	13951.1	9	1	1	1	41	23	1	60
Gobiomorphus coxii	40946.2	0	2	1	0	20	14	0	31
Hypseleotris compressa	21872.7	6	3	2	0	56	0	0	58
Hypseleotris galii	14380.5	3	3	2	0	51	0	0	53
Hypseleotris klunzingeri	78975.2	39	13	5	0	22	0	0	60
Leiopotherapon unicolor	61573	36	2	1	6	39	0	0	63
Maccullochella macquariensis	18786.1	57	28	9	19	0	9	11	78
Maccullochella peelii	65082.9	32	25	6	16	1	1	5	55
Macquaria ambigua	53639.4	26	34	1	20	0	0	5	59
Macquaria australasica	20805.7	39	8	4	0	0	20	7	55
Macquaria colonorum	3756.7	9	6	4	0	0	0	0	14
Macquaria novemaculeata	6844.2	9	3	1	0	0	0	0	12
Melanotaenia duboulayi	15642.3	3	2	1	0	29	0	0	31
Melanotaenia fluviatilis	50641.2	37	25	1	0	5	0	0	57
Mogurnda adspersa	38639.9	47	2	4	0	50	0	13	76
Mordacia mordax	4276.3	0	0	0	0	0	0	0	0
Nannoperca australis	11987.6	39	6	3	0	34	0	20	65
Nannoperca flindersi	3081.5	0	1	2	0	1	0	1	3

Nannoperca oxleyana	383	1	3	7	0	57	0	0	58
Neosilurus hyrtlii	4482.7	55	4	1	2	0	0	0	57
Notesthes robusta	7620.7	10	2	1	0	0	0	0	11
Philypnodon grandiceps	40026.9	22	4	2	0	0	1	2	26
Philypnodon macrostomus	43307	4	2	2	0	0	1	1	7
Potamalosa richmondia	3774.6	6	4	2	0	0	4	1	14
Prototroctes maraena	5865.5	4	1	1	38	0	80	0	89
Pseudaphritis urvillii	4041.6	2	1	1	0	3	32	2	36
Pseudomugil signifer	18031.7	4	2	1	0	28	0	0	30
Retropinna semoniMTV	105077.9	25	10	4	0	48	0	0	63
Retropinna semoniSEC	46258	4	1	2	0	28	0	0	31
Rhadinocentrus ornatus	3751.1	8	2	2	0	59	0	0	62
Tandanus bellinger	3900.9	4	1	1	1	0	0	1	4
Tandanus tandanus	81554.7	27	4	1	34	0	0	14	56

Appendix 8 – Differences in Management Costs of Actions in Worst-Case Scenarios

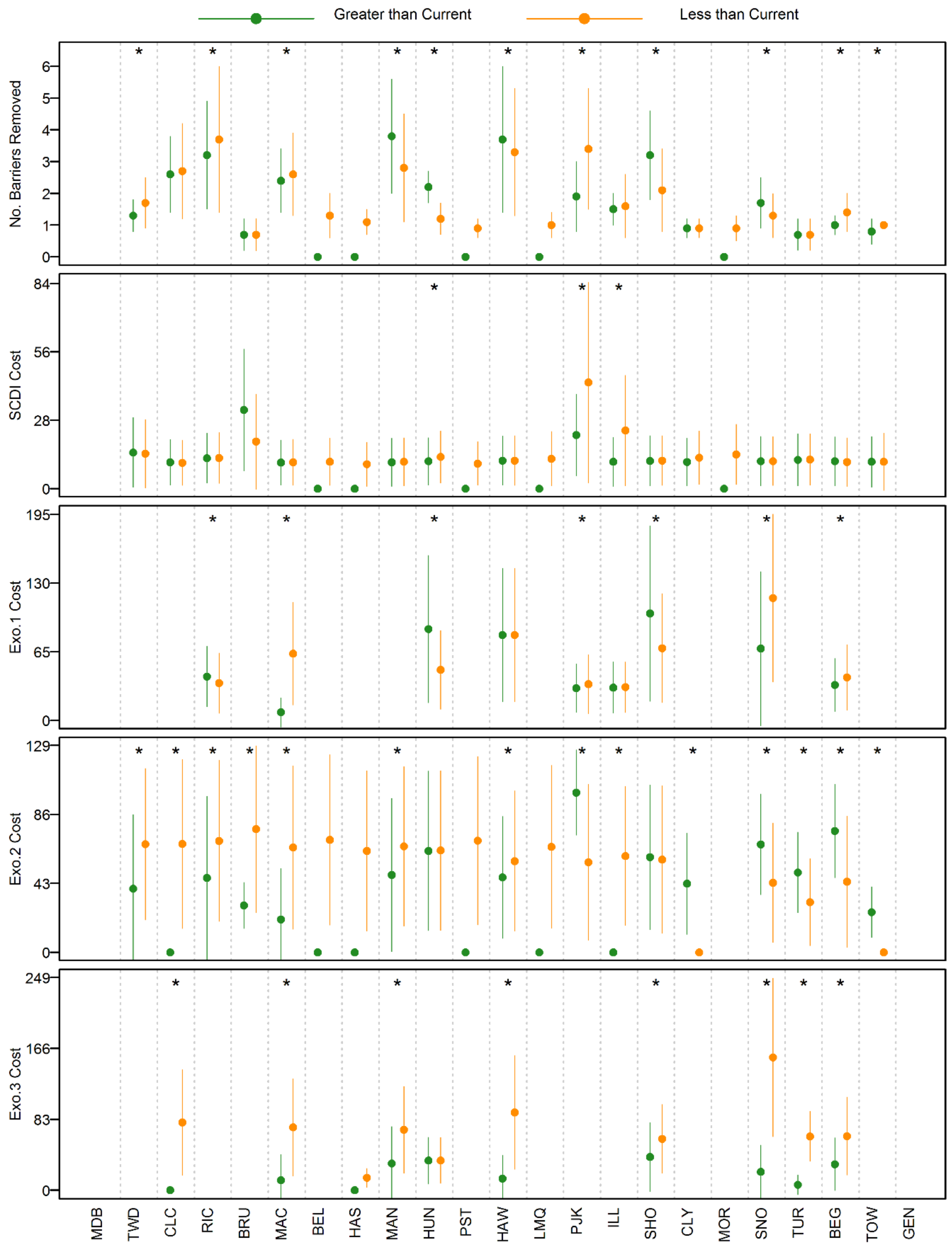

Figure A7.1 – Mean cost of management action in each catchment divided between those whose gains under the worst-case future scenario are greater than or equal to the current (green), or less than the current benefits (orange), based on all native fish species (*n*=49). Stars indicate significant differences (*p*=0.001). Catchment acronyms are matched in Figure Z1.

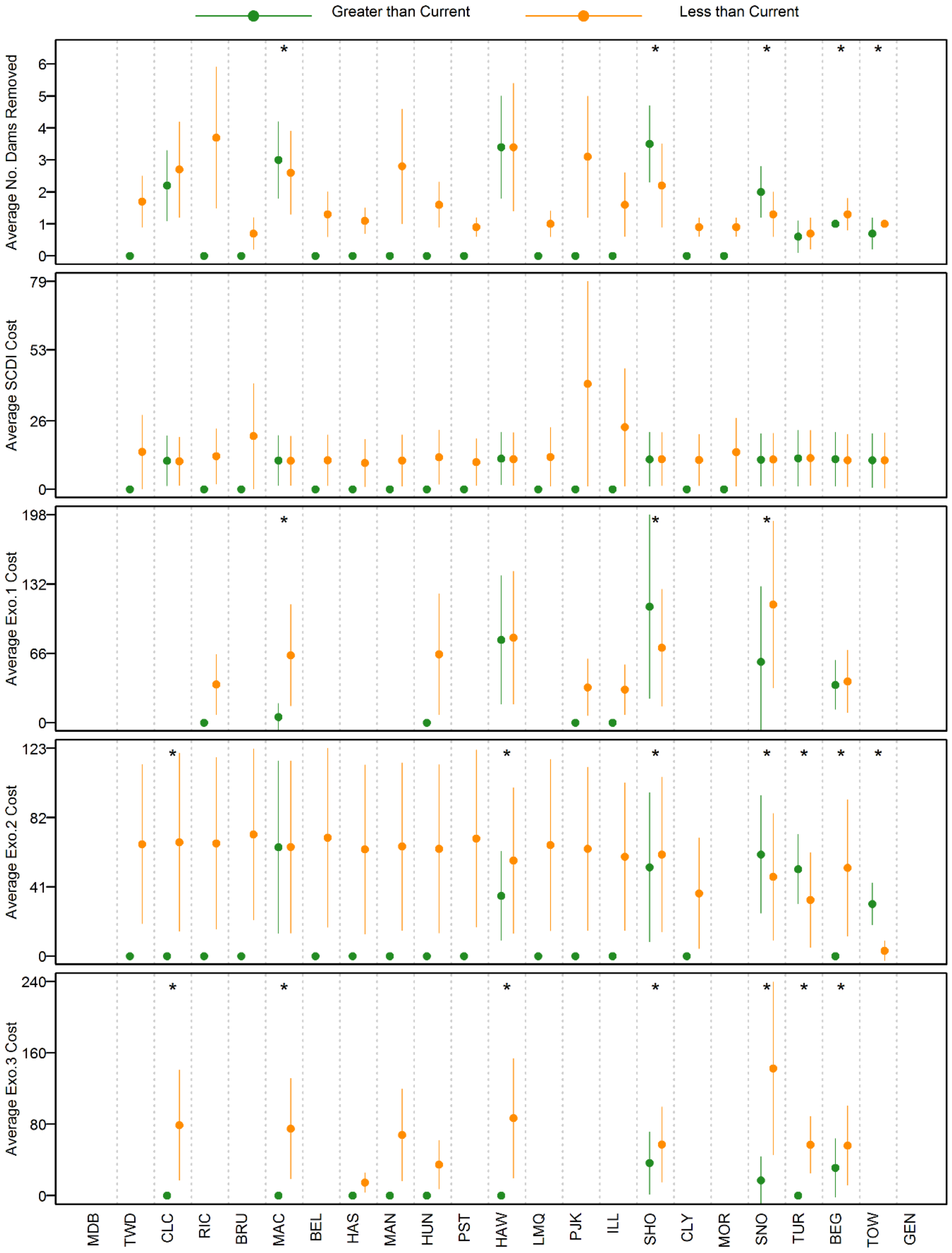

Figure A7.2 – Mean cost of management action in each catchment divided between those whose gains under the worst-case future scenario are greater than or equal to the current (green), or less than the current benefits (orange), based on climate sensitive fish species (*n*=26). Stars indicate significant differences (*p*=0.001). Catchment acronyms are matched in Figure Z1.

Appendix 9 – Correlation Coefficients for Management of Climate-Sensitive and Threatened Species

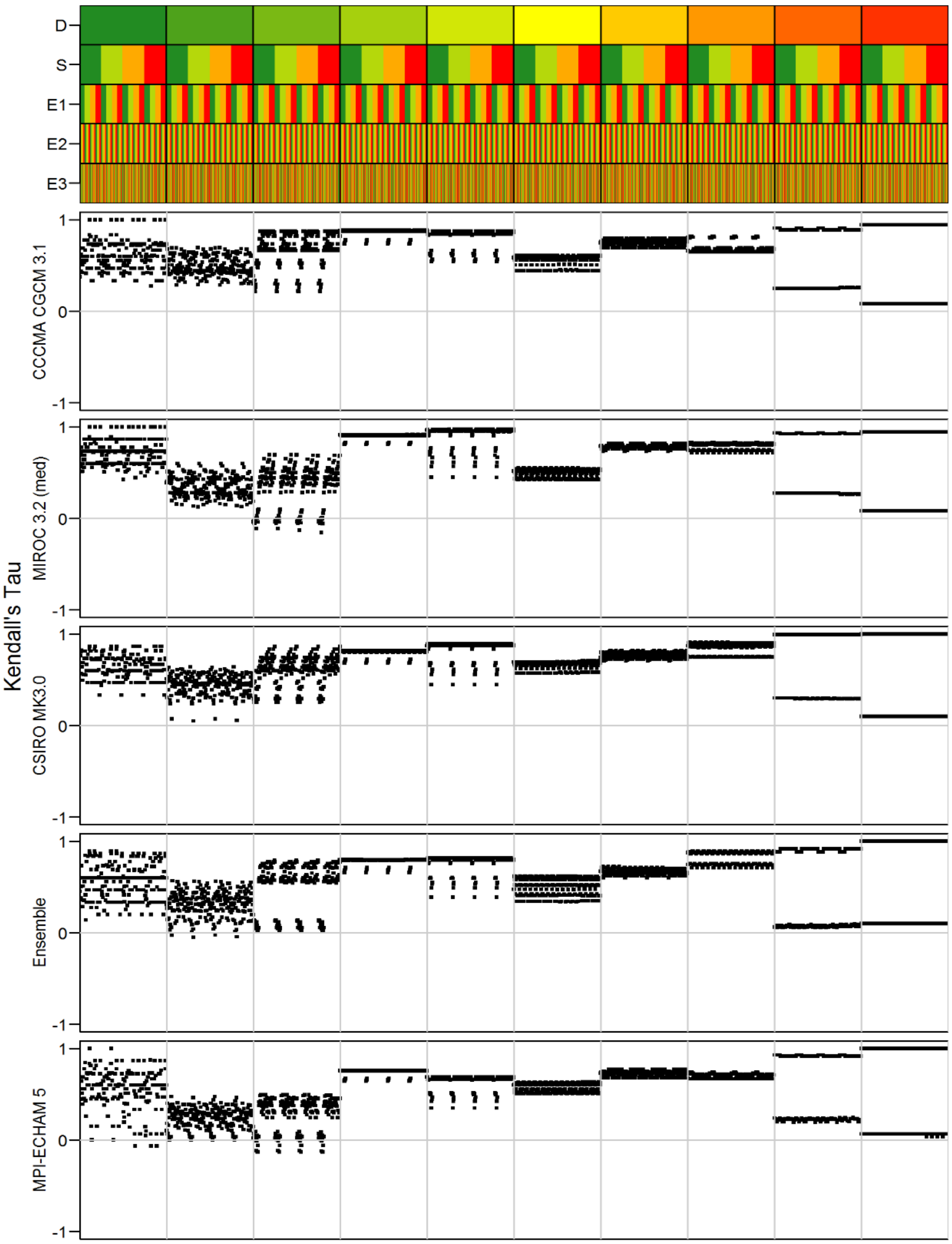

Figure A8.1 – Correlation coefficients of management priorities for the top 20 management actions for climate-sensitive species (*n*=2559) between current and future climate scenarios (1 indicates complete agreement, and -1 a reversal of relative priorities). Coloured shading indicates the number of barriers removed (D, removing 0-10 barriers), and the four levels of restoration for land-use (S) and exotic species control (E1, E2, E3).

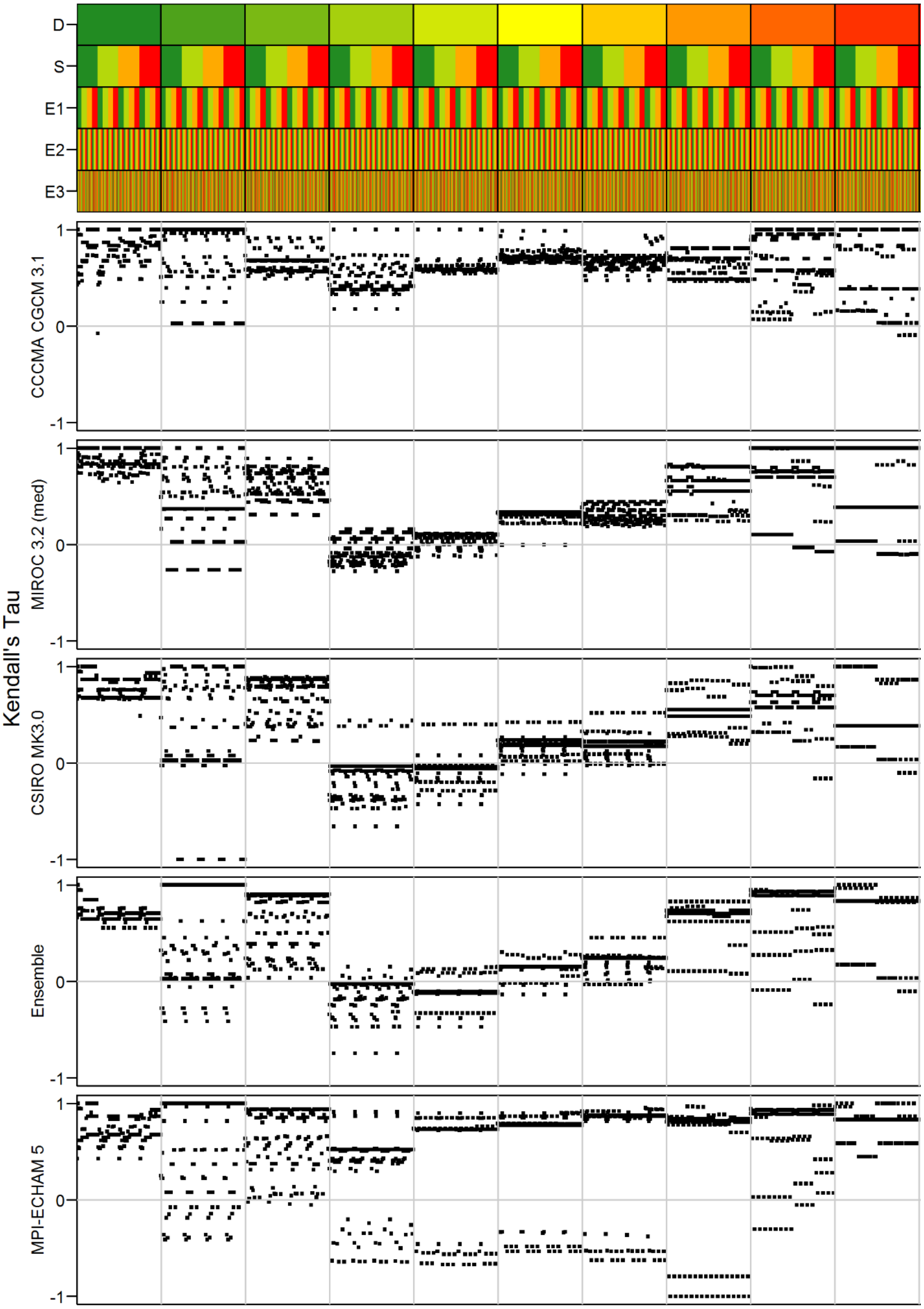

Figure A8.2 – Correlation coefficients of management priorities for the top 20 management actions for listed threatened species (*n*=2559) between current and future climate scenarios (1 indicates complete agreement, and -1 a reversal of relative priorities). Coloured shading indicates the number of barriers removed (D, removing 0-10 barriers), and the four levels of restoration for land-use (S) and exotic species control (E1, E2, E3).

Appendix 10 – Catchment Management Summary Reports