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Invited Review Paper

How to incorporate climate change into modelling environmental water outcomes: a review

Andrew John, Rory Nathan, Avril Horne, Michael Stewardson and J. Angus Webb

ABSTRACT

Environmental water represents a key resource in managing freshwater ecosystems against pervasive threats. The impacts of climate change add further pressures to environmental water management, yet anticipating these impacts through modelling approaches remains challenging due to the complexities of the climate, hydrological and ecological systems. In this paper, we review the challenges posed by each of these three areas. Large uncertainties in predicting climatic changes and non-stationarities in hydrological and ecological responses make anticipating impacts difficult. In addition, a legacy of relying on modelling approaches informed by historic dependencies in environmental water science may confound the prediction of ecological responses when extrapolating under novel conditions. We also discuss applying ecohydrological methods to support decision-making and review applications of bottom-up climate impact assessments (specifically eco-engineering decision scaling) to freshwater ecosystems. These approaches offer a promising way of incorporating climatic uncertainty and balancing competing environmental objectives, but some practical challenges remain in their adoption for modelling environmental water outcomes under climate change.

Key words | climate change, ecohydrology, ecological modelling, environmental water, freshwater ecosystems

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INTRODUCTION

Freshwater ecosystems are critically important due to their high biodiversity and the ecosystem services they provide (Dudgeon *et al.* 2006). Managing these effectively is vital to support sustainable development and communities worldwide (Arthington *et al.* 2018). This connection between ecosystem health and human well-being has been outlined in global policy agendas since the early 2000s. The concept is reflected in the *United Nations Sustainable Development Goals*, a programme that prioritises sustainable management of water to support access to clean water and sanitation and promotes ecological conservation (Bhaduri *et al.* 2016). Water for the environment is ultimately a policy that supports the health of rivers and wetlands, the communities that depend on them and hence human quality of life (Acreman 2001). Nonetheless, freshwater ecosystems face significant threats from over-exploitation, pollution, flow modification, habitat degradation and invasive species (Dudgeon *et al.* 2006; Vörösmarty *et al.* 2010). Climate change both adds new threats at local and basin scales and compounds existing pressures through changes to hydrological processes, ecosystem interactions and the way humans respond to changes (Thomas *et al.* 2004;

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IPCC 2014a, 2014b; Reid *et al.* 2019). If the threat of climate change to freshwater ecosystems is not addressed through mitigation or adaptation efforts, we risk extensive environmental, economic and social impacts (Ripple *et al.* 2019).

Globally, governments have responded to flow-related threats to freshwater ecosystems through the provision of *environmental water*. This is broadly the provision, or guarantee, of water of suitable quality, quantity and timing to support ecological objectives and the community values that depend on them (Horne *et al.* 2017a, 2017b).

The earliest examples of environmental water focussed on minimum passing flows downstream of water storages to provide rudimentary habitat for riverine species (Tharme 2003; Acreman & Dunbar 2004). Since then, there have been substantial improvements in our understanding of the importance of preserving the variability and range of streamflow components that are key determinants of the ecosystem structure and function (Poff *et al.* 1997, 2017; Bunn & Arthington 2002). It is now known that ecology responds to individual and combinations of flow events, which are important for specific phenological elements of different species. As such, it is necessary to manage for particular flow sequences, not just regime-averages (Anderson *et al.* 2006; Poff 2018; Wang *et al.* 2018a, 2018b; Horne *et al.* 2019).

There are various legal mechanisms to provide environmental water, including conditions on other water users and storage operators to maintain a particular flow regime within the river, and also the creation of environmental water rights (Horne *et al.* 2017a, 2017b). The creation of environmental water rights provides a volume of environmental water that can be actively managed and released from storage to achieve specific and targeted environmental outcomes (O'Donnell & Garrick 2017; Horne *et al.* 2018). In some jurisdictions, the creation of environmental water rights has involved substantial legal and policy reform across large regional scales and can be measured in the billions of dollars. For example, the Murray–Darling Basin Plan in south-east Australia is securing 2,750 GL of environmental water rights through a 13 billion-dollar ($AUD) public investment in water shares and irrigation efficiency improvements (Hart 2016a, 2016b). Climate change poses a significant threat to these investments and the environmental objectives that they support.

Planning and managing for freshwater ecosystem outcomes under climate change requires new approaches to contemporary water resource and conservation management (Poff 2018; Tonkin *et al.* 2019). Here, we explore the nature of the climate change threat and discuss the shortcomings in our understanding and the uncertainties involved in assessing the impacts of climate change on environmental water planning. We review approaches to modelling freshwater ecosystem responses to climate change and the role of environmental water in adaptation. We discuss challenges in projecting hydrological risk and modelling ecological response using existing methods, including difficulties in representing key uncertainties. We also identify contemporary approaches that have been specifically developed to support vulnerability assessment and decision-making under uncertainty and discuss what is needed to adopt them for use in environmental water management.

PREDICTING THE FUTURE: OUR UNDERSTANDING OF GLOBAL CLIMATE CHANGE

The outputs from general circulation models (GCMs) form the bulk of our understanding of likely climatic changes over the coming century and are commonly used as direct inputs to climate change risk assessments (Foden *et al.* 2018). These global scale and complex models simulate many of the fundamental thermodynamic processes that govern atmospheric circulation. Early GCMs operated at a global scale but were mainly focussed on modelling the atmosphere, with simple parameterisations and assumptions governing large portions of the land and oceans (Edwards 2011; Flato 2011). Their performance and complexity have evolved substantially over the decades, and contemporary GCMs now include representation of land surface processes and feedbacks, detailed ocean circulation and some components of terrestrial carbon and hydrological cycles (IPCC 2013). In the land surface domain, this includes hydrological feedbacks resulting from processes such as land-use change and irrigation (Sridhar & Anderson 2017). Modern models that include these more comprehensive processes are sometimes referred to as earth system models.

However, although they are our best tool, the inherent uncertainties of GCMs hinder their direct application in assessing ecological responses to changing flows (ecohydrological analysis). GCMs usually operate with scenarios

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of anthropogenic greenhouse gas emissions defined by end-of-century radiative forcing targets. However, the uncertainty introduced by the range of available emissions scenarios is considerable. For example, the multi-model ensemble mean results from the fifth Coupled Model Intercomparison Project (CMIP5) show global average temperature increases in 2100 of ~0.4 and 4.0 °C between the lower and upper bounding scenarios of RCP2.6 and RCP8.5 (IPCC 2013). Clearly, the difference in planning and managing freshwater ecosystems for these two contrasting futures would be enormous. Uncertainty in future emissions is compounded with parametric and other elements of structural uncertainty within GCMs. Studies presenting GCMs results such as CMIP5 typically do not explore parametric uncertainty and instead use a single best estimate for all model parameters (Hargreaves 2010). This is partly responsible for the range of estimates of equilibrium climate sensitivity (the long-term global response to a doubling of atmospheric carbon dioxide concentrations) between models of ~1.5–4.5 °C (IPCC 2013; Vial *et al.* 2013). In addition, although there are up to 39 different GCM models available for ensemble representation of the different emissions scenarios, many individual GCMs share common components such as their atmospheric or ocean modules (Knutti *et al.* 2010), and this lack of independence between ensemble members limits our ability to assess the true uncertainties involved. Parametric and structural uncertainty can be explored by using probabilistic projections from simple climate models (Meinshausen *et al.* 2011), but the outputs of this kind of analysis are yet to find their way into climate change risk assessments.

The need to simulate at global scales for decades or centuries into the future means not all processes can be physically resolved. This is especially true for processes of key importance to hydrological assessments, where the typical minimum horizontal resolution of between 50 and 100 km in GCMs means convection and cloud formation must be parameterised with high uncertainty (Stevens & Bony 2013). As a result, GCMs typically feature the 'drizzle rain problem,' where rainfall occurs too regularly and in too small quantities (Stephens *et al.* 2010). One of the reasons this problem persists is that GCM performance is usually evaluated over a range of climate variables and large spatial scales (Gleckler *et al.* 2008; Knutti *et al.* 2010). The inability to fully capture local-scale processes, including hydrological processes such as interception, evapotranspiration and effects of anthropogenic irrigation (Jaksa & Sridhar 2015; Seong *et al.* 2018), combines with difficulties in representing large-scale drivers of natural variability. Many GCMs struggle to represent phenomena such as the El Nino Southern Oscillation (Bellenger *et al.* 2014), which affects precipitation patterns and even temperatures over extremely large scales (Dai & Wigley 2000). This is problematic for assessing the impacts of climate change on environmental outcomes as these are dependent on changes in the timing and duration of wet and dry periods over short (daily) and long (multiyear) time scales.

Given the difficulties in GCM prediction of local or regional precipitation, downscaling techniques assist in transforming information from GCMs to a more local scale. This is typically achieved through one of two approaches: statistical and dynamical downscaling. Statistical downscaling relates larger scale climatic variables that GCMs are known to better replicate to those at local scales, and there are a number of different methodological options for doing so (Fowler *et al.* 2007; Ekström *et al.* 2015). Most of these rely on establishing statistical relationships between climate variables over a training period and assuming the derived large-scale relationships will remain valid despite anticipated climate changes (Wilby 1997; Salvi *et al.* 2016). In dynamical downscaling, regional climate models support GCMs in modelling over a smaller spatial domain, but at increased resolution (Di Luca *et al.* 2015). This can allow the use of convection-permitting models to better understand local impacts, although large uncertainties remain and there are significant computational costs in these approaches (Prein *et al.* 2015). However, within ecohydrological risk assessments, the most common kind of downscaling used is the delta change method (John *et al.* submitted). This involves taking a historical record of precipitation (and other important environmental variables) and changing them by a factor informed by GCM outputs. This factor can change seasonally or be based on quantiles but the simplest and most common approach is to apply a fixed value to the whole time series (Prudhomme & Nick Reynard 2002; Fowler *et al.* 2007; Wang *et al.* 2017). This method is rapid to implement where the observed time series of precipitation

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are already available but has severe limitations in the way variability and hydrological sequences are represented. Natural variability is constrained to the observational period and the sequence of wet and dry periods will always follow the fixed pattern observed in the historical record (Anandhi *et al.* 2011; Johnson & Sharma 2011; Wang *et al.* 2018a, 2018b).

MODELLING VARIABILITY, CHANGE AND THE CONFOUNDING EFFECT OF ENVIRONMENTAL NON-STATIONARITIES

Hydrological impacts under climate change have typically been communicated through changes to the long-term behaviour in various flow statistics (Olden & Poff 2003; Ekström *et al.* 2018), yet freshwater ecosystems function under conditions of a naturally variable climate. Whilst information on changes to long-term average regimes is easy to digest, it ignores the dynamic influence of variability in the system being considered (Anderson *et al.* 2006). It is this variability that determines the structural makeup of ecosystems, how species interact, their individual evolutionary responses to adapting to harsh conditions and the environmental tolerances past which populations may become extinct (Poff *et al.* 1997). This variability may also play a key role in how vulnerable species respond to climate change (Wang *et al.* 2018a, 2018b; Nathan *et al.* 2019). Thus, to manage freshwater ecosystems, we need to consider how different aspects of climate and consequent hydrological variability may change, and be able to report this against the natural range of variability experienced by the species or ecosystem (Horne *et al.* 2019; Nathan *et al.* 2019).

Characterising hydrological variability is made more difficult due to non-stationary changes in climate, hydrology and ecological dynamics. For decades, ecohydrological modelling techniques have relied on the assumption of stationarity for characterising input data and modelling hydrological processes and ecological outcomes (Milly *et al.* 2008). In reality, it is common to calibrate models across non-stationary data records, due to the ongoing changes to land-use, policy and infrastructure development in many places. However, it is now clear that this includes non-stationarity due to climate change and also that climate change can amplify existing sources of non-stationarity within a river basin. Our knowledge of the magnitude and precise nature of these changes is not well-developed. Some hydrological examples of the causes of non-stationarity include shifts in rainfall–runoff regime following extended dry conditions (Saft *et al.* 2015), the construction of diversion weirs and dams (Poff *et al.* 2007), land-use change and urbanisation which affects the distribution of water in the landscape and new policy arrangements or management strategies which alter how regulated systems are governed. Non-stationary ecological factors that increase the modelling challenge include changing species interactions due to local extirpations (Rahel & Olden 2008; Hobbs *et al.* 2009), species invasions (Koncki & Aronson 2015; Hulme 2017) and the slow but measurable evolutionary adaptation in response to environmental change (Hoffmann & Sgró 2011). With environmental water, many assessment methods rely on establishing a reference case based on 'natural' conditions to inform specific flow recommendations for different ecological objectives (Tharme 2003). There would appear to be opportunities to use paleorecords to infer ecological responses to climate variability over long periods of time prior to the historic record (Dawson *et al.* 2011). Such approaches have been used to understand ecosystem changes in wetlands from extreme events (Ralph *et al.* 2011) and climate-induced changes in lake bodies (Saros *et al.* 2012). However, such data are based on a number of assumptions (such as the stationarity of teleconnections) which may weaken its relevance to assessing the impacts of climate change (Power *et al.* 1999). The implications of ecohydrological non-stationarities are that these natural flow regimes inferred from historic hydrological time series are less relevant as reference conditions and predictors of future responses, and by managing for them we may not be achieving the ecological objectives we set out to (Poff 2018).

The outcomes of risk assessments may be misleading if important changes to underlying relationships are not taken into account. These changes can act as feedbacks that amplify or dampen ecological responses to climate change (Mehran *et al.* 2017; Sofaer *et al.* 2018). Examples of the numerous ways that environmental changes may result in different ecological outcomes include changing rainfall–runoff relationships, which may result in more extended drought periods than found in the historic record

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(Saft *et al.* 2016a, 2016b); increasing temperatures that may unlock new habitats allowing invasive species to add pressure to conservation species already suffering from increased thermal stress (Koncki & Aronson 2015; Sofaer *et al.* 2018); and anthropogenic adaptation measures, such as increasing regulation of water resources due to water scarcity, which may adversely impact on ecologically important flow regimes.

The ability to take account of interacting non-stationarities depends on the modelling tools used; different modelling approaches vary substantially in the way that key processes are simulated. For example, it is quite common to simulate land-use change scenarios in conceptual hydrological models by changing model parameters in a lumped or spatially explicit manner (e.g. Bussi *et al.* 2017). Similarly, it is straightforward to simulate changes in how regulated water resource systems are operated or to simulate the influence of individual infrastructure components through model configuration in river systems models (Welsh *et al.* 2013). However, it is not straightforward to model non-stationary rainfall–runoff relationships or ecological dynamics because current tools are generally not designed to accommodate them. Furthermore, the data required to defensibly characterise the changes are considerable (Thyer *et al.* 2006; Saft *et al.* 2016a, 2016b), and there may be difficulties in extrapolating information across sites and through time depending on methods used (Poff 2018). In addition, where these non-stationarities are included in risk assessments, they are usually modelled in isolation, and their combined effect across larger catchment or basin scales remains poorly explored (Ormerod *et al.* 2010; Mantyka-Pringle *et al.* 2014; Mouquet *et al.* 2015).

MODELLING ECOLOGICAL RESPONSES

Even given the difficulties of assessing climate-induced hydrological change, there remains the challenge of modelling its effect on ecology. The methods and tools available to model ecological responses and the behaviour of different target species or communities vary considerably in their complexity. There are broadly three approaches to representing ecological responses to changing environmental conditions: correlative, trait-based and mechanistic (Pacifici *et al.* 2015). Correlative approaches establish statistical relationships between climate or hydrological variables and ecological response, such as seasonal flow thresholds and the presence of fish species inferred from gauged river flows and sampled catch data. However, they assume stationarity in the derived relationships and are most often used to infer equilibrium responses over long time periods. Trait-based approaches use species traits to assess vulnerability. Traits themselves are environmentally sensitive species attributes that can be generalised across sites. Assuming the necessary ecological knowledge is available at a particular site (which can be problematic for remote sites or rare species), traits-based assessments can be rapid to apply (Pacifici *et al.* 2015). Mechanistic approaches differ in that they use physiological relationships, usually at a finer scale, to determine ecological response to change. This can include representation of important phenological interactions and life-history stages that are relevant to target species. The benefits of these process-based simulations may include better representation of compound threats such as land-use and climate change (Mantyka-Pringle *et al.* 2014). Drawbacks of mechanistic approaches include a reliance on extensive data, issues in scaling in space and time and the difficulty of parameterising certain processes (Morin & Thuiller 2009; Cabral *et al.* 2017).

Different model types, which vary in the way that ecological behaviour and response are represented, are used with the approaches above. Key metrics can be used to relate responses directly to environmental variables. These kinds of evaluations indicate the relative changes between impact scenarios but do not attempt to evaluate direct impacts on species. Conversely, species distribution models are used to map habitat for species based on correlative or mechanistic relationships (Keith *et al.* 2008). The key assumption with these models is that habitat is the limiting factor for species occurrence or proliferation. Because colonisation rates and dispersal ability are only seldom considered (Bush & Hoskins 2017), these models mostly concentrate on equilibrium or long-term responses (Chapman *et al.* 2014). However, one limitation of species distribution models is that they generally do not include species interactions in determining habitat (Davis *et al.* 1998; Wiens *et al.* 2009; Beckage *et al.* 2011). This is a very important consideration, as the work suggests that these

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interactions may be more important than direct abiotic effects in mediating climate change impacts (Ockendon *et al.* 2014). Finally, demographic models determine changes in abundance, often using process-based simulations that predict life-history events. These models are able to provide a time series of outputs at different resolutions so that they can be used to evaluate ecological response over short timescales or against environmental events. A key challenge in implementing demographic modelling approaches is that they require extensive data and understanding (Urban *et al.* 2016).

Given the controlling influence of variability in freshwater ecology, modelling approaches and models themselves should be able to accommodate changes in variability (Anderson *et al.* 2006). An important attribute is the need to explicitly consider temporal sequences of individual flow events. These events relate directly to the processes that govern species or ecosystem conditions, such as spawning cues for fish or floodplain inundation for riparian vegetation recruitment (Koster *et al.* 2017). Extreme events too, have large impacts on freshwater ecosystems, some beneficial – such as river avulsions to unlock new habitat (Ralph *et al.* 2011) – and some devastating, such as 'blackwater' (low dissolved oxygen) events resulting in fish kills (Watts *et al.* 2018). Environmental water management has in some places recognised the importance of simulating individual events, notably where watering recommendations are based on specific regime elements such as bankfull or fresh flows. Yet much of the ecological response modelling tools available are still 'states' based using regime-averages (Wheeler *et al.* 2018). Mechanistic modelling approaches are more defensible under non-stationary climatic, hydrological and ecological conditions and should be used where the data and ecological knowledge are available (Horne *et al.* 2019; Tonkin *et al.* 2019). Explicit modelling of the impact of climate extremes and changes in variability is not well-represented in current assessments (Butt *et al.* 2016), but demographic, process-based models are equipped to offer time-varying outputs that respond to specific events. Improving the capability of models to respond to individual events is especially important for active environmental water management, where the aim of water managers is to use specific flow events to modulate threats.

Any prediction of ecological response has large uncertainties because of the inherent complexity in ecosystems over a range of spatial and temporal scales (Levin 1992; Wheatley *et al.* 2017). Indirect impacts of climate, including interactions among different threats, could lead to compounding or non-linear impacts in freshwater ecosystems (Schmitz *et al.* 2003). The call to include species interactions in climate change impact assessments is not new (Davis *et al.* 1998; Harrington *et al.* 1999), as is the call to incorporate non-flow stressors (Jackson *et al.* 2016). There is a trade-off in the level of ecological model complexity and data requirements and how best to align these with the decision-making around environmental water management (Webb *et al.* 2017). Increasing model complexity often does not make outputs better for informing decisions; there is a 'requisite complexity' at which model complexity is optimised for the purpose it is being put to (Webb *et al.* 2017).

USING ECOHYDROLOGICAL METHODS AND MODELS IN DECISION-MAKING

Environmental water management requires information to understand how much water the environment needs and at what times, and where environmental water rights exist how best to make use of this water. Traditional approaches to making these decisions rely on historical data and comparisons of impacted and unimpacted time series, often using static flow indicators as surrogates for ecosystem outcomes. The previous sections have highlighted two key challenges (among others) with this approach: (1) the future is unlikely to unfold the same as the past, and there also remain large uncertainties in our projections of future climate and streamflow; and (2) static based metrics are unlikely to provide adequate information to predict ecological outcomes beyond what has been experienced historically. Continuing to adopt these approaches to make environmental water management decisions under climate change may lead to poor decisions based on spurious modelling results (John *et al.* submitted).

Other areas of water resource management have used so-called 'bottom-up' methods as a means of addressing the uncertainty in predicting future climate outcomes, for example scenario-neutral (Guo *et al.* 2017) or decision scaling methods (Brown *et al.* 2012). These offer a promising future direction for both the modelling of ecological impacts

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associated with climate change and how to base decision-making on concepts that incorporate climatic uncertainty and variability. The premise of these methods is to broadly diagnose system vulnerability to a large range of possible changes, rather than relying on explicit predictions of the future from downscaled GCMs. Often a 'stress test' of the system being considered is used to map vulnerabilities against varying dimensions of environmental and climatic change (Brown *et al.* 2012). Likely impacts of climate change projected from GCMs are then assessed in the context of wider system vulnerabilities, but adaptation plans can be developed to address a range of uncertainty and different threats. This links the benefits of stochastic vulnerability analysis with useful information from climate models. One of the key benefits to these approaches is that they are much more compatible with (and even embrace) the uncertainties driven by climate change.

While there are increasing examples of these methods in water supply applications (e.g. Turner *et al.* 2014; Steinschneider *et al.* 2015; Henley *et al.* 2019), there have been limited ecological applications. Environmental outcomes are incorporated in the high-level decision framework *of eco-engineering decision scaling* (EEDS) which aims specifically to examine trade-offs between stakeholder-defined engineering or resource based outcomes and ecological outcomes (Poff *et al.* 2016). A literature search reveals two recent case studies that have applied EEDS in practice to Andean streams (Rosero-López *et al.* 2019) and a regulated river system in South Korea (Kim *et al.* 2019). These two case studies reveal a number of challenges in applying EEDS in the way that hydrological and ecological information is represented:

- **The hydrological impact of climate change was represented primarily through changes to mean precipitation** (Kim *et al.* 2019). Rosero-López *et al.* (2019) investigated current climate only but hydrological sensitivity is tested against median annual flows. Most other bottom-up assessments (not necessarily using EEDS) that focus on water supply security perturb only two climate or environmental variables, such as mean annual rainfall and temperature (Brown *et al.* 2012; Henley *et al.* 2019). However, in freshwater ecology, the relevant environmental factors that contribute to vulnerabilities may be many due to the known dependence of ecological communities on hydrologic variability and different components of the flow regime.
- **Ecological outcomes were represented using simplified approaches, without a clear representation of sequencing.** In the case of Kim *et al.* (2019), compliance against minimum passing flows was used to represent ecological outcomes within the EEDS approach. The study uses a stochastic approach to generate precipitation time series that does contain multiple sequences, but the representation of ecological responses is not sensitive to them. Rosero-López *et al.* (2019) based analysis on empirical data. Empirical relationships were developed between a ratio of instantaneous flow to median annual flow and data from a taxonomic survey of macroinvertebrates important for fish food and water quality. The data collection method effectively represents the repeated states-based approach to defining flow-ecology relationships (where data are representative of snapshots of the ecological condition through time (Wheeler *et al.* 2018)). One of the drawbacks of repeated-states approaches is an inability to provide temporally specific ecological responses to changing flows, hence limits the recommendation of specific environmental flow sequences (Wheeler *et al.* 2018). The regressions were based on a relatively short monitoring period and, as such, are unlikely to include the full suite of climate conditions that may be experienced in the future (Tonkin *et al.* 2019).
- **There are challenges in the complexity and computational requirement of water resource system models** (Kim *et al.* 2019). Since varying degrees of change are assessed for their impact on the system, the computational cost of modelling can be very large. System responses are then assessed for each combination of these perturbations, and if the effectiveness of management responses is being assessed, the analysis must be repeated for each proposed intervention. This may be confounded if using more complex climate scenarios and perturbations, along with the use of more complex ecological models. There is a need to define a requisite level of modelling complexity that achieves a trade-off in allowing interacting changes and sources of uncertainty to be explored, whilst providing outputs that are detailed enough to inform meaningful management responses.

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- **Methods were adapted in data-scare regions** (Rosero-López *et al.* 2019). The methods that have been discussed in this paper and outlined in the EEDS approach all require significant existing system knowledge and modelling.

These two applications of EEDS demonstrate some of the practical challenges of bringing together the key discipline develops for ecology, hydrology and climate science to inform decision-making around environmental water. While it is acknowledged in the literature that the traditional driving perspectives of ecology and hydrology in environmental water must also be combined with expertise in climatology to understand future risks (Foden *et al.* 2018; John *et al.* submitted), there remains significant challenges in achieving this.

A further advantage of these approaches is that where they are used to make management decisions, stakeholder engagement in defining system objectives is a key step in the methodologies (Marchau *et al.* 2019). The role of stakeholder engagement in environmental flows has only recently gained prominence in the literature (Arthington *et al.* 2018; Anderson *et al.* 2019) and becomes regarded as vital for promoting legitimacy and support for environmental flows programs (O'Donnell *et al.* 2019). Just as the physical processes of hydrology and ecology exhibit non-stationarity, societal values and priorities also change through time. Our paper has focussed primarily on the physical aspects of modelling climate change implications for river ecosystems. We note however that environmental flow management must recognise that it exists within a socio-ecological system that will also change through time.

CONCLUSION – ENVIRONMENTAL WATER IN THE ANTHROPOCENE

The grand experiment humans are performing through anthropogenic greenhouse gas emissions will lead to substantial climatic, societal and ecological changes across the globe. Our ability to predict exactly what these changes will be over long timescales is significantly limited, simply because the world and its ecosystems are extremely complicated. The significant uncertainties in predicting climate change using GCMs will likely persist into the next decades (Maslin & Austin 2012). Our understanding of future risks needs to embrace these uncertainties if we are able to understand the range of possible outcomes and effectively respond in an adaptive manner.

There are three key recommendations to support the future of modelling the impact of climate change to inform environmental water responses:

1. **The uncertainty in future climate changes and its effect on hydrologic regimes should be explicitly considered.** This includes investigating ways hydrological non-stationarities could affect river flows in the system being considered but will allow a more comprehensive understanding of possible hydrological impacts associated with climate change.
2. **Ecological responses to specific flow sequences should be considered in addition to changes to long-term flow regime characteristics.** Methods used to generate hydrological impacts should also be complementary to distinguishing the impact of particular flow sequences. This will assist in the modelling and planning of active management of environmental water.
3. **Modelling ecological responses should be based on a mechanistic understanding of the processes and interactions that govern ecosystem behaviour.** This helps ensure ecological dynamics that are captured in system responses and that environmental water use will have the desired effects.

Much of our current understanding of the relationship between the environment and freshwater ecology is informed by observations over a narrow range of historical conditions that may not be relevant to potential future changes (Poff 2018). We cannot assume that historical relationships used to model ecological response will hold in the future under the novel conditions imposed by climate change. Reliance on current methods that are built on historic dependencies should be abandoned in favour of the process-based understanding of systems. The active use of environmental water supported by process-based ecological models is perhaps one of our most valuable tools for conservation under climate change. Being prepared for changes means firmly establishing adaptive management practices

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in the planning and use of environmental water (Webb *et al.* 2018).

However, developing models that support decision-making around environmental water under climate change is not without challenges. Existing applications of decision scaling methods demonstrate the difficulties in linking best practice across climatology, hydrology, ecology and systems modelling. There remains a significant research need to develop methods that allow the assessment of ecological outcomes under climate change to inform decision-making in a way that accommodates uncertainty.

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