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Predicting water quality and ecological responses

Final Report

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ABSTRACT

Changes to climate are predicted to have effects on freshwater streams. Stream flows are likely to change, with implications for freshwater ecosystems and water quality. Other stressors such as population growth, community preferences and management policies can be expected to interact in various ways with climate change and stream flows, and outcomes for freshwater ecosystems and water quality are uncertain. Managers of freshwater ecosystems and water supplies could benefit from being able to predict the scales of likely changes.

This project has developed and applied a linked modelling framework to assess climate change impacts on water quality regimes and ecological responses. The framework is designed to inform water planning and climate adaptation activities. It integrates quantitative tools, and predicts relationships between future climate, human activities, water quality and ecology, thereby filling a gap left by the considerable research effort so far invested in predicting stream flows.

The modelling framework allows managers to explore potential changes in the water quality and ecology of freshwater systems in response to plausible scenarios for climate change and management adaptations. Although set up for the Upper Murrumbidgee River catchment in southern NSW and ACT, the framework was planned to be transferable to other regions where suitable data are available. The approach and learning from the project appear to have the potential to be broadly applicable.

We selected six climate scenarios representing minor, moderate and major changes in flow characteristics for 1^oC and 2^oC temperature increases. These were combined with four plausible alternative management adaptations that might be used to modify water supply, urban water demand and stream flow regimes in the Upper Murrumbidgee catchment.

The Bayesian Network (BN) model structure we used was developed using both a ‘top down’ and ‘bottom up’ approach. From analyses combined with expert advice, we identified the causal structure linking climate variables to stream flow, water quality attributes, land management and ecological responses (top down). The ‘bottom up’ approach focused on key ecological outcomes and key drivers, and helped produce efficient models. The result was six models for macroinvertebrates, and one for fish. In the macroinvertebrate BN models, nodes were discretised using statistical/empirical derived thresholds using new techniques.

The framework made it possible to explore how ecological communities respond to changes in climate and management activities. Particularly, we focused on the effects of water quality and quantity on ecological responses. The models showed a strong regional response reflecting differences across 18 regions in the catchment. In two regions the management alternatives were predicted to have stronger effects than climate change. In three other regions the predicted response to climate change was stronger. Analyses of water quality suggested minor changes in the probability of water quality exceeding thresholds designed to protect aquatic ecosystems.

The ‘bottom up’ approach limited the framework’s transferability by being specific to the Upper Murrumbidgee catchment data. Indeed, to meet stakeholder questions models need to be specifically tailored. Therefore the report proposes a general model-building framework for transferring the approach, rather than the models, to other regions.

EXECUTIVE SUMMARY

Climate change is predicted to affect Australian freshwater ecosystems by altering the quality and quantity of water in rivers. In turn, this may increase the vulnerability of aquatic fauna to human impacts and management strategies. Managing freshwater ecosystems therefore requires accurate prediction of the changes likely to occur as a result of climate change in combination with impacts such as population growth and management policies. While considerable effort has been invested in predicting stream flow changes under different climate scenarios, we know less about potential water quality and ecological responses.

In particular, we lack ways of integrating existing data to predict relationships between future climate, human activities, water quality and ecology. The key objective of this project was to develop a modelling framework to assess climate change impacts on water quality and, consequently, aquatic fauna (macroinvertebrates and native fish species).

The catchment of the Upper Murrumbidgee River was selected as a case study because it represents many of the issues faced by water management agencies across the country. The area has a growing urban population that requires water and waste water management for the future without compromising ecological values in freshwaters. Climate scenarios were defined and assumptions underpinning each scenario were described (project objective 1). Next, hydrological and water quality models were used to analyse climate scenarios and assess their outcomes (project objective 2). Third, the results of climate scenario analysis were used to assess risks to water quality and ecosystems (project objective 3). Finally, the transferability of the proposed modelling framework was tested for use in the Goulburn Broken catchment (project objective 4).

We used Bayesian Networks (BN) to model the complex interactions between climate impacts, non-climate pressures (such as population growth), adaptation (management) decisions, water quality attributes, and ecological responses defined by macroinvertebrate and fish communities. The model structure was developed using both a ‘top down’ and ‘bottom up’ approach. From expert advice we defined links between climate variables, stream flow, water quality attributes, land management and ecological responses (top down). The resulting model was then simplified by identifying key predictors of ecological response (bottom up) using statistical techniques for macroinvertebrates and expert opinion for fish. The bottom-up approach resulted in an efficient way to reduce the number of predictor variables (nodes) in the BN models. An original 128 initial predictor variables were reduced to between five and nine. The selected predictor variables differed depending on the ecological responses and habitat types in the river (edge and riffle).

This is important because it highlights the specificity and diversity of relationships in freshwater ecosystems. It also highlights the need to be clear about objectives and endpoints for predictive modelling.

Three different measures of macroinvertebrate response (Observed/Expected taxa score, relative abundance of thermophobic taxa and macroinvertebrate assemblage) were used as endpoints in the BN models. Ecological thresholds were empirically estimated using a variety of traditional and novel statistical methods. Estimated thresholds values were similar between the measure of macroinvertebrate response and the method used to identify the threshold. Defined thresholds were used as a novel way to determine states in the BN models. In the case of fish, absence/presence of six native

species was considered, and thresholds were assigned based on expert opinion and literature.

The approach resulted in seven BN models (one model per macroinvertebrate endpoint for each river habitat type, and one fish model for the six native fish species). We used the BN models to explore ecological responses to changes in climate and management activities. Six climate scenarios were selected representing minor, moderate and major changes in stream flow for 1^{o}C and 2^{o}C temperature increases. Climate scenarios were combined with four plausible alternative ways of managing water supply, demand and stream flow, which captured local adaptation initiatives to secure water for the catchment's needs.

The BN modelling indicated that projected water quality changes associated with climate change are small in the Upper Murrumbidgee catchment. The probability that thresholds designed for the protection of aquatic ecosystems would be violated was negligible under most scenarios. However, outcomes varied and models showed strong regional responses to climate and management alternatives. Management had the strongest impact in the Upper Murrumbidgee and Lower Molonglo regions, where the direct impact of adaptation initiatives on water releases (Upper Murrumbidgee) and treated effluent discharge (Lower Molonglo) outweighed climate change effects. In contrast, the Upper Cotter, Yass and Goodradigbee regions showed a stronger response to climate; predicted increased temperatures would affect sensitive macroinvertebrate taxa and vulnerable native fish species. Tolerant macroinvertebrate taxa were unlikely to be affected by climate change or management alternatives.

Varying regional and ecological responses have significant consequences for the prioritisation of adaptation initiatives in response to climate change, suggesting they should be applied specifically, not uniformly. In some regions of the Upper Murrumbidgee catchment, adaptation initiatives appear to have minimal influence, while in others adaptation initiatives may be highly beneficial. In regulated rivers, where future climate and management scenarios involve high water demand and decline in indicators of river health, managing regulation and demand should be a central strategy for protecting freshwater ecosystems. In unregulated but stressed rivers of the region, climate change is likely to amplify current negative effects of catchment management practices. In these regions, a continued focus on improved catchment management will be central to mitigating the effects of climate.

The approach applied in this study may have great value in assessing climate change and management impacts in other regions, though regional differences in stakeholder priorities and management approaches will mean the models must be altered. Managers in the Upper Murrumbidgee catchment are interested in the response of ecosystems and water quality to climate and management actions. Conversely, managers in the Goulburn Broken catchment are more interested in changes to water quality; for them, understanding the implications for river health is a more distant objective. The report proposes a model-building framework for use in other regions.

While this project makes a number of significant advances in understanding the impacts of climate change and management actions on freshwater ecosystems, a number of knowledge gaps remain. Among key areas that should be prioritised for future research are: improved capacity to include extreme climatic events and seasonal changes in flow in models, and addressing the need for experimental data outside of historical climate conditions to which ecosystems have not yet been exposed.

1. INTRODUCTION

This section introduces the study and describes its objectives and the approach it has taken: namely, a major case study, scenario planning and application of Bayesian Networks. The section also outlines the structure of this final report in relation to the contractual reporting obligations.

1.1 Motivation

It is generally accepted that the climate is changing. There are predictions of significant changes to runoff, stream flow (Arnell 2003; CSIRO 2008; Thodsen 2007) and water quality (Delpla et al. 2009; Murdoch, Baron & Miller 2000; Whitehead et al. 2009; Wilson & Weng 2011), which will leave freshwater ecosystems vulnerable.

Australia is particularly exposed to changes in hydrological regimes; communities and ecosystems across the country could be at risk (IPCC 2007; PMSEIC IWG 2007). As a consequence, the future management of freshwater species and ecosystems, particularly those that already are at or near their climate limits, requires predictions of the magnitude of changes in flow and water quality likely to occur as a result of the combined effects of climate change and other stressors such as population growth, community preferences and management policies. The likely success of intervention activities (such as translocation of populations) to secure the future of aquatic species, depends on information about species' physiological tolerances, as well as predictions of future water quality (and how water quality is distributed). This information is also needed to determine Australia's capacity to achieve conservation goals and set policy direction.

At present we cannot determine the extent and implications of changes to water quality that might result from climate change. Neither can we determine the potential ecological consequences of any such changes to water quality. Interactive effects between climate change, water quality, water volumes, human use and biota will further complicate prediction of consequences. We need to be able to make informed decisions about the possible impacts of management adaptation initiatives on the quality of our water resources and freshwater ecosystems. To do that, we need to be able to integrate quantitative tools — to predict the relationship between water quality and climate change.

1.2 Aim and objectives

The primary aim of this project was to develop an integrated modelling framework with which it is possible to make predictions about different climate and adaptation scenarios, and inform water planning and climate adaptation activities. A secondary aim of the project was to apply our modelling framework to evaluate possible impacts of adaptation initiatives.

Towards these aims, we set four key objectives, comprising 12 components.

Objective 1: Estimate the probability, extent and magnitude of water quality changes by linking climate attributes to water quality models. This had five components.

a) Define a set of management scenarios, based on predicted changes in climate patterns (scenarios of precipitation and temperature), land use (including bushfires) and water demands.

b) Predict likely changes to flow regimes under the defined climate scenarios.

c) Predict changes in frequency distributions and probability of threshold exceedance under the defined climate scenarios.

d) Predict exceedance probabilities for water quality attributes designed to protect ecological communities, including the effects of management adaptation initiatives (e.g. for waste water management).

e) Verify the probabilistic water quality models.

Objective 2: Develop a Bayesian Network model to link the projected changes in water quality and quantity and changes in ecosystems, particularly focusing on the probability of adverse biological effects. This had three components.

a) Make an integrated assessment of the relationship between scenarios for climate, land use and water demand, and water quality and ecological response

b) Identify key drivers and ecologically relevant thresholds.

c) Quantitatively and qualitatively calibrate relationships captured as probabilities in the Bayesian Network.

Objective 3: Use the Bayesian Network models to inform management adaptation initiatives. This had three components:

a) Evaluate the consequences of management adaptation initiatives for future water security, and the consequences of waste water management for water quality and ecological response.

b) Evaluate the probability that current water quality regulation will protect ecological communities.

c) Identify priorities (both spatially and in terms of ecological communities) for management adaptation initiatives based on probabilities of adverse effects.

Objective 4: Determine the transferability of the model framework to other regions. This had one component

a) Modify the modelling framework so it can be broadly applicable.

1.3 Approach

We combined and used three research approaches to achieve the project's objectives: a case study; scenario planning; and Bayesian Networks.

1.3.1 Case study: Upper Murrumbidgee catchment

To develop the integrated modelling framework we selected the catchment of the Upper Murrumbidgee River for our case study. The catchment extends from the headwaters of the Murrumbidgee River on the Long Plain in Kosciuszko National Park to the Burrinjuck Dam near Yass. Tributaries of the Murrumbidgee River within the catchment include the Bredbo, Numeralla, Goodradigbee, Cotter and Yass Rivers. Section 2.1 gives a brief overview of the catchment.

This catchment was ideal for this case study because it encapsulates important management issues. The area faces significant water supply challenges to meet the growing demands of Australia's largest inland city, Canberra. Water resource managers are actively considering adaptation initiatives to secure future water supply for human

consumption (ActewAGL 2004) while maintaining and improving the ecological condition of these freshwaters (ACT Government 2004).

Sub-catchments of the Upper Murrumbidgee catchment encompass a range of hydrological and ecological conditions: for example, there are significant areas reserved for conservation, and others areas subject to intensive development. The catchment has also been exposed to the effects of various driving forces (e.g. droughts, bushfires, urbanisation), so it is a region in which we can examine the interactive impacts of these forces, and develop tools and lessons that may be transferable to other areas.

In selecting this catchment for the case study we considered the feasibility of implementing the proposed modelling framework as well as the effectiveness and usability of project outputs. We concluded that:

- the approach we would be developing has the potential to be nationally relevant, because issues active in the Upper Murrumbidgee catchment — stresses from urban development, extreme climate conditions, and infrastructure — are being faced by water resource management agencies across Australia;
- the extensive water quality and biological monitoring data sets and historical information available for the catchment provided a unique opportunity to review projected water quality and climate scenarios in light of historical conditions;
- there was already a foundation of published hydrological and ecological work in the catchment by members of the research team, which would form a sound basis for the modelling framework.

A case study approach can be an effective way to explore and understand complex socio-ecological systems. First, environmental issues are context-dependent; they cannot be investigated without considering the wide range of factors (e.g. biophysical, social, legislative) that affect the phenomenon of interest. Second, a case study approach is well suited for addressing “why” and “how” research questions, and for generating findings that are directly relevant and applicable to the study area.

1.3.2 Scenario planning

Next, we applied ‘scenario planning’ to the case study catchment.

“Scenario” is defined as “a coherent, internally consistent and plausible description of a possible future state of the world” (IPCC 2007). Each scenario represents a story about plausible changes in future conditions, the impacts of these interactive changes on the sustainability of socio-ecological systems, and the consequences of adaptation decisions under each alternative future. Useful scenarios need to be comprehensive, credible, relevant and transparent.

In situations where uncertainty is very high and controllability is very low, scenarios can be helpful support in decision making (Schoemaker 1991). For example, climate change and water use patterns are beyond the ultimate control of water management authorities. In such cases, scenarios can help decision makers formulate hypotheses about the future (i.e. “what-if” questions), and envisage their consequences.

Managers and policy-makers who need to secure human and ecological water requirements are looking ahead to the medium and long term. Their planning must be able to deal with fundamental shifts in drivers (e.g. a step change in climate conditions).

It needs to be robust under various conditions, particularly in environmental policy-making or where there is a long lead time (e.g. building a new infrastructure), and where outcomes may trigger expensive, possibly irreversible consequences (e.g. loss of species). By using scenarios, instead of relying on single snapshots of how the future may unfold, decision makers are better able to look at a series of alternative management policies and assess their outcomes under a set of plausible future conditions.

When used in environmental decision making, scenario planning is an adaptive process through which researchers and stakeholder groups work together to share and develop credible and stakeholder-relevant information about the future of the system.

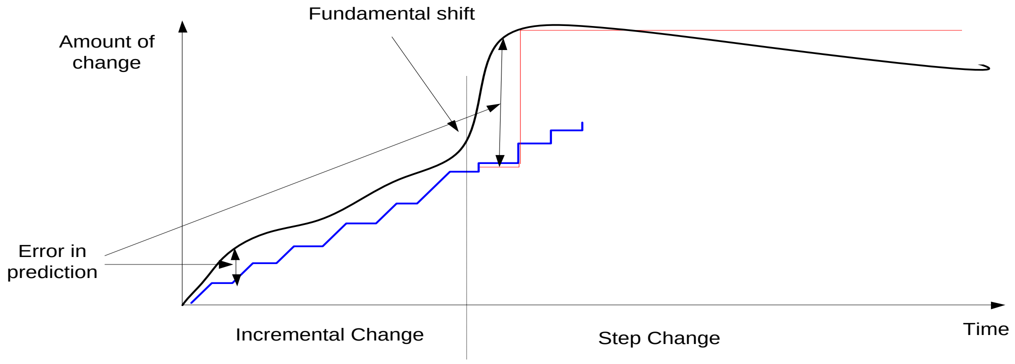
By contrast, forecasting models (no matter how sophisticated they are) assume that past patterns will continue to the future. This approach may be effective for short term planning, and only in case of incremental changes (Figure 1).

(i) Choosing and defining a scenario

According to how a scenario is used to address management questions, there are two types of scenarios: (1) exploratory or normative scenarios, and (2) single or combined.

- Exploratory scenarios postulate changes in the key conditions and then explore the possible outcomes. Exploratory scenarios are used for vulnerability analysis and adaptation analysis, and for impact assessment. Normative scenarios start by postulating a desirable or undesirable future image or system state; then they open up pathways to attain or avoid these states.
- Single scenarios are used to examine one source of future uncertainty (e.g. IPCC carbon emission scenarios) or the combined impacts of two or more uncertainties.

Figure 1 here. Half page



Note: Prediction models assume that past patterns will continue in the future. As long the continuity assumption is valid, the error in prediction is relatively small. When the system behaviour exhibits a fundamental shift, prediction models fail to capture the new patterns.

Figure 1. The performance of forecasting-based approaches in cases of incremental and radical changes in system behaviour

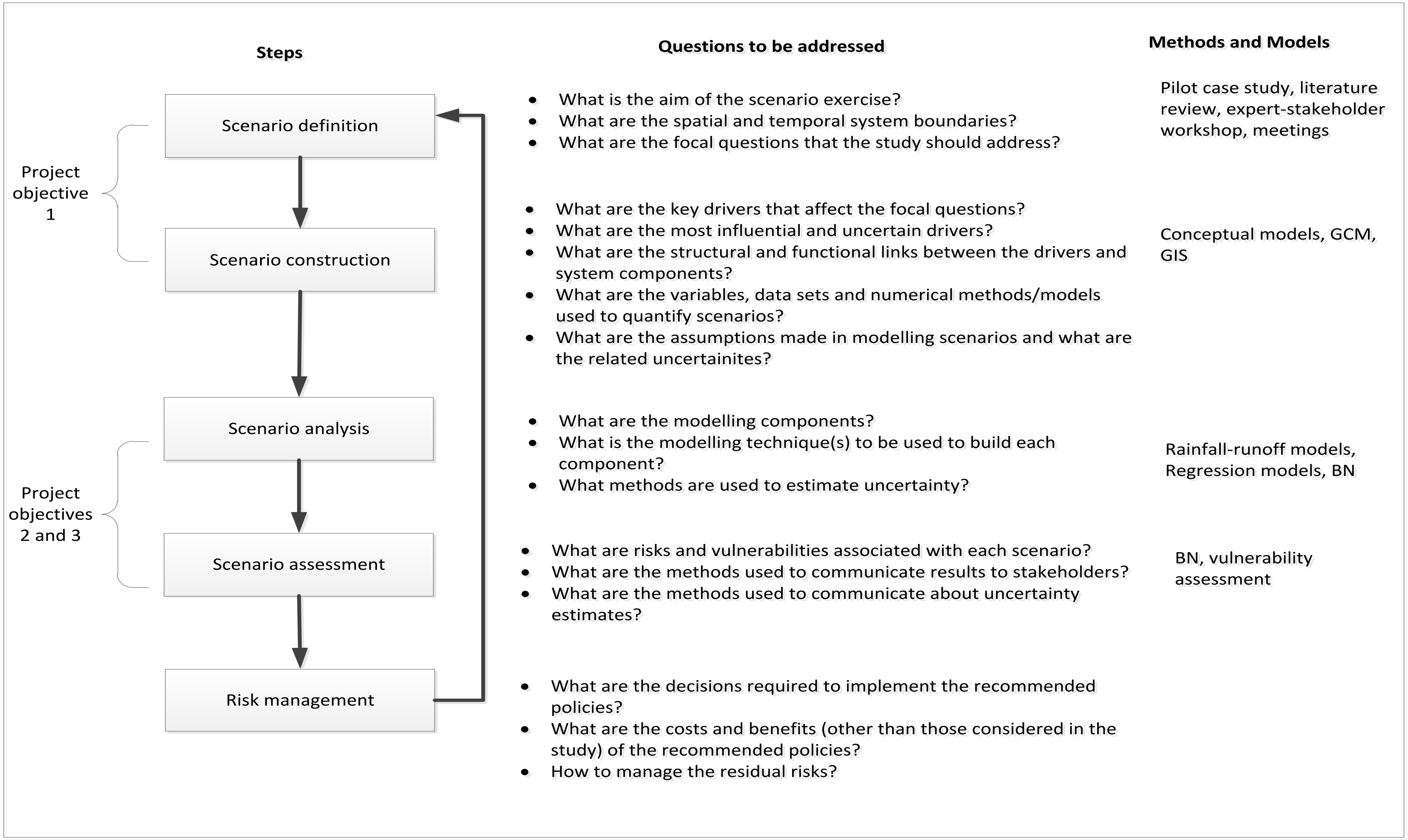


Figure 2. The scenario planning process adopted in the study

In the research literature there are several frameworks for structuring and describing a scenario planning process (e.g. Mahmoud et al. 2009; Van Vuuren et al. 2012). In essence, these frameworks share five main steps or tasks (Figure 2).

1. Scenario definition. In light of the project objective, researchers work with stakeholders and end-users to identify information that is of interest to stakeholders and policy making and relevant to possible scenarios. For example, focus areas and periods of time, and the focal questions about the system's future that scenarios need to address.
2. Scenario construction. Researchers, stakeholders and end-users work together to identify the drivers (natural and human-induced) that are thought to affect the focal questions, directly or indirectly. They determine which drivers are the most uncertain and the most influential (i.e. this is qualitative scenario input). In quantitative modelling approaches, each driver is then explored to find or construct numerical variables and mathematical relationships that quantitatively describe changes in this driver (i.e. this is numerical scenario input).
3. Scenario analysis. Researchers develop analytical methods and models so they can make quantitative assessment of the scenarios. They use them to test the models' outputs, estimate uncertainties, and define indicators that will help them in reporting results.
4. Scenario assessment. Researchers interpret and synthesise the outputs of the scenarios and their implications for end-users. This might include, for instance, identifying the most influential threats, vulnerable spots, and leverage points for adaptation interventions. A critical part of this phase is deciding how to communicate results and uncertainty estimates to decision makers (e.g. vulnerability maps, risk matrices).
5. Risk management. This phase includes adoption and implementation of recommended policies, and goes beyond the scientific inquiry (in terms of time and scope). Whereas researchers can still provide advice on risk management issues, this phase is mainly the responsibility of decision makers.

1.3.3 Bayesian Networks

Bayesian Networks, the third approach used in this project, provided the high level integration framework model.

Bayesian Networks (hereafter BNs) are directed graphical models that use statistical inference techniques first proposed by Bayes (Heckerman, Geiger & Chickering 1995; Morawski 1989; Olson, Willers & Wagner 1990). BNs model complex interactions within ecosystems by calculating the relative probabilities of competing hypotheses, given a particular set of conditions (Ludwig 1996). From these calculations, researchers can identify the most probable hypothesis (Taylor et al. 1996).

In a BN model, each variable is represented by a "node" (Charniak 1991). For ecological studies nodes can represent either predictor variables — such as management regimes, climate scenarios and environmental disturbance factors — or response variables, such as changes in macroinvertebrate abundance or richness (McNay et al. 2006). Directional arrows link the nodes and indicate causal relationships between them (Morawski 1989; Olson, Willers & Wagner 1990).

Relationships between different variables represent conditional dependencies. For example, researchers can model change in one variable and its resulting independent and interactive effects on other nodes in the BN (Stewart-Koster et al. 2010).

BNs are being increasingly used to model ecological systems (Allan et al. 2012; Borsuk, Stow & Reckhow 2003; McCann, Marcot & Ellis 2006; Ticehurst et al. 2007) as well as to assist decision making within water resource management (Castelletti & Soncini-Sessa 2007; Chan et al. 2010; Molina et al. 2010). BNs have also been used as a modelling framework (Borsuk, Stow & Reckhow 2004; Varis & Kuikka 1997), or coupled with other types of models (c.f. Liedloff & Smith 2010) to model ecological responses.

BNs are very useful in modelling predictive changes in ecosystem health because they do not require design, sampling, randomisation or replication of data sets (Reckhow 1990), nor sampling within assumed temporal or spatial scales (Ellison 1996; Smith et al. 2007). This means several data sets can effectively be combined within the one model.

Bayesian statistics are also robust when using small sample sizes (Gazey & Staley 1986; Ter Braak & Etienne 2003), and incomplete data sets (Walton & Meidinger 2006), including those collected from populations that may have been previously affected by human activities (McNay et al. 2006). This is because the expectations maximisation method used in Bayesian learning can cope with missing observations regardless of whether they are random or not (Heckerman, Geiger & Chickering 1995). This means that inference can be gained even from field data collected from uncontrolled environments with few replicates (Ellison 1996; Smith et al. 2007).

BNs also allow researchers to attach probabilities to interactions, and so risks and uncertainties can be better estimated than in models which are limited to only expected values (Reckhow 1999; Uusitalo 2007).

In this project, we used BNs as the high level integration framework to elicit, capture and express our knowledge across the hydrological, water quality and ecological domains. This had the advantage of building on pre-existing well developed (and tested) disciplinary component models (such as hydrological models) and allowing them to be linked to less well developed models (such as ecological response models).

The visual nature of BNs means results are presented in a highly interpretable format for managers, which is an advantage in environmental decision-making (Crome, Thomas & Moore 1996; Taylor et al. 1996). BNs also enable scenario testing, which is very useful for identifying major levers, such as water quality or management impacts, which drive changes within the ecosystem (Hart & Pollino 2008). This made them the obvious choice to use in this project.

1.4 Final report structure

Combining the three research approaches we were able to achieve our four main objectives. We tell the story in a series of separate sections of this report, even though the three research approaches were integrated during the study. Figure 3 illustrates the structure of the study and Table 1 maps each of the objectives of the study to sections of the report.

Section 2 sets the stage by introducing the case study area — the Upper Murrumbidgee River and its catchment, its water resources and ecological situation and the legislation

and regulations that govern water planning. In Section 2, we outline the information we used in defining and constructing the four management adaptation alternatives tested in the project. These four scenarios were the first building block for subsequent analysis and modelling.

Section 3 describes the methods we used to predict the flow regime changes within the catchment, including the selection of climate scenarios that we used in the study. The predicted changes to flow regimes were analysed using hydro-ecological indicators of change and compared against previously published effects of river regulation. Section 3 describes the linking of flow regime changes to water quality, and the water quality modelling approach. In this section we also give the predicted probabilities of exceedance for water quality attributes designed to protect ecological communities.

Section 4 describes the approach we used to investigate the relationships between environmental predictors and ecological responses. Specifically, we identified the important indicators (predictors) of ecological response (changes to macroinvertebrate or fish communities); and we quantified threshold values (change points) in environmental predictors for ecological responses (macroinvertebrates). We used this information subsequently to structure the integrated BN model and discretise the main model nodes.

Section 5 outlines how we constructed the Bayesian Network that links the theoretical future changes in water quality and quantity with changes in the ecosystems. The structure of the models is described, as is the discretisation of each node in the network. In this section we also assess the main uncertainties and limitations of the networks.

Section 6 describes how we used the BNs to inform the four management adaptation alternatives. This section describes a series of 'story lines' that demonstrate the use of the BN models to explore scenarios both of climate change and of management. We evaluate the probability that current (2000s) water quality guidelines will protect ecological communities in the future. We explore the likely consequences to water quality and aquatic ecology of applying particular management adaptation initiatives for future water security and waste water management. Then we put the management adaptation alternatives into an order of priority based on the probabilities that they could have adverse effects.

Section 7 assesses the transferability of the model framework to the Goulburn Broken catchment — a large area also inland in eastern Australia and to the south-west of the case study catchment. We identify how the model could be revised to make it broadly applicable. The section presents a framework for a linked modelling approach and presents the key findings from this project which will guide future use of the framework.

Finally, Section 8 describes the top three knowledge gaps we identified during the project.

Detailed results are presented in a number of Appendixes.

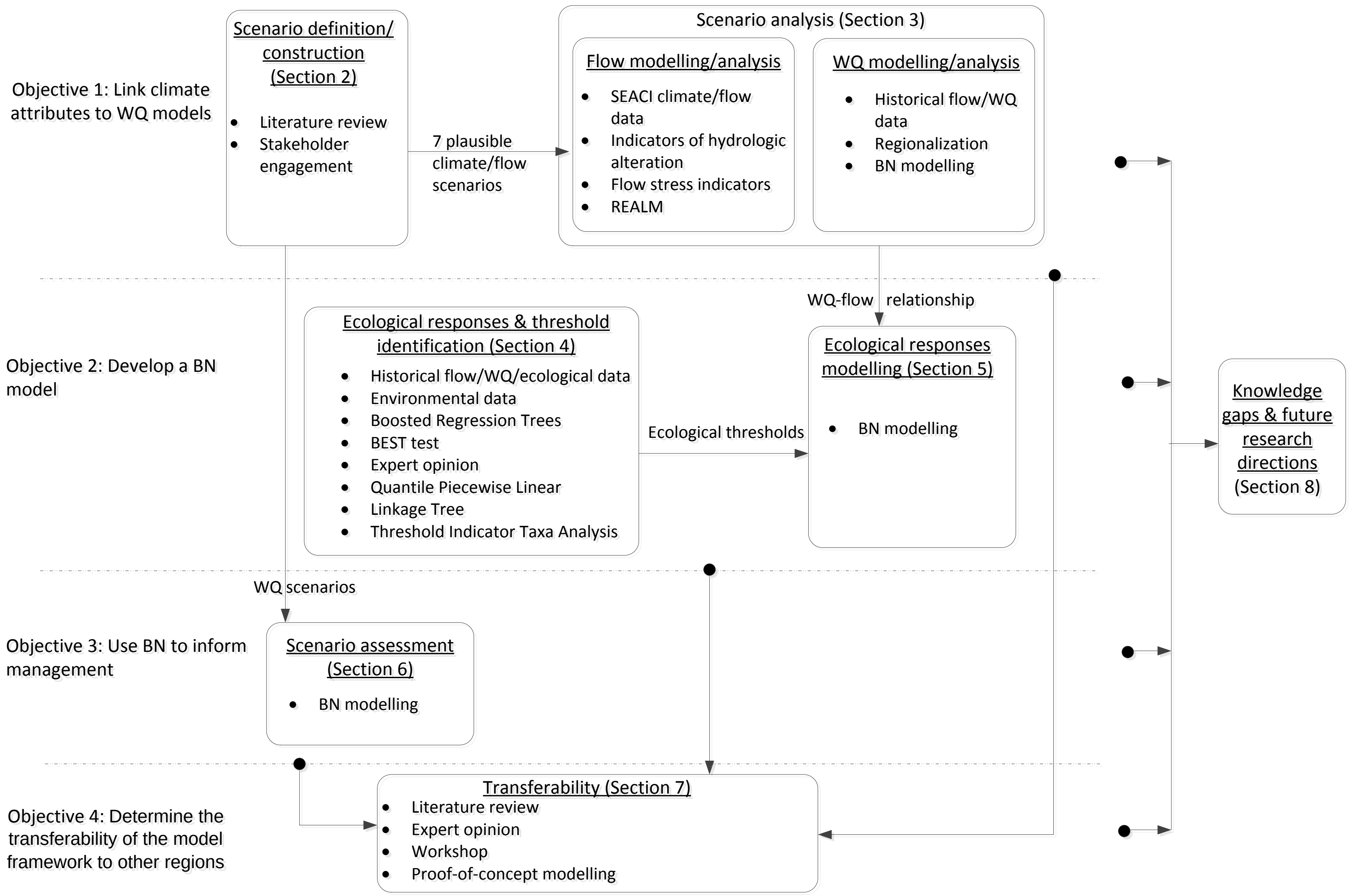


Figure 3. Project objectives, the framework of key data or methods developed and used to address these along with report structure

Table 1. Study objectives and components mapped to sections of the report

Objective/component	Report section
Objective 1: Estimate the probability, extent and magnitude of water quality changes by linking climate attributes to water quality models.	Section 2 and Section 3
Define a set of management scenarios, based on predicted changes in climate patterns (scenarios of precipitation and temperature), land use (including bushfires) and water demands.	Section 2
Predict likely changes to flow regimes under the defined climate scenarios.	Section 3
Predict changes in frequency distributions and probability of threshold exceedance under the defined climate scenarios.	Section 3
Predict exceedance probabilities for water quality attributes designed to protect ecological communities, including the effects of management adaptation initiatives (e.g. for waste water management).	Section 3
Verify the probabilistic water quality models.	Section 3
Objective 2: Develop a Bayesian Network model to link the projected changes in water quality and quantity and changes in ecosystems, particularly focusing on the probability of adverse biological effects.	Section 4 and Section 5
Make an integrated assessment of the relationship between scenarios for climate, land use and water demand, and water quality and ecological response.	Section 4
Identify key drivers and ecologically relevant thresholds.	Section 4
Quantitatively and qualitatively calibrate relationships captured as probabilities in the Bayesian Network.	Section 5
Objective 3: Use the Bayesian Network models to inform management adaptation initiatives.	Section 6
Evaluate the consequences of management adaptation initiatives for future water security, and the consequences of waste water management for water quality and ecological response.	Section 6
Evaluate the probability that current water quality regulation will protect ecological communities.	Section 3 /Section 4
Identify priorities (both spatially and in terms of ecological communities) for management adaptation initiatives based on probabilities of adverse effects.	Section 6
Objective 4: Determine the transferability of the model framework to other regions.	Section 7
Modify the modelling framework so it can be broadly applicable.	Section 7

2. SCENARIO DEFINITION AND CONSTRUCTION

This section outlines geographical, ecological and hydrological characteristics of the catchment of the Upper Murrumbidgee River — the physical region that is the context for this project. It presents relevant information about the region's hydro-climate and water resources (supply and demand), bushfires, factors in human demand for water, ecosystem protection and environmental flows. The information helped us decide on four theoretical 'management adaptation alternatives' for use in modelling future stream flows and contributed to the development of conceptual models that underpin the BN modelling.

2.1 Upper Murrumbidgee River catchment

The Murrumbidgee River is the third longest river in the Murray-Darling Basin (MDB), and in Australia. It rises in the Kosciuszko National Park on the Long Plain, and flows south-east, then north, and then westward for 1485 km in total (GA 2012), to its confluence with the River Murray near Balranald in southern NSW.

The Upper Murrumbidgee River extends from the source, above Tantangara Dam in the Snowy Mountains, to the Burrinjuck Dam wall. The catchment of the Upper Murrumbidgee River includes the northern slopes of the Snowy Mountains, and the Southern Tablelands region from Bredbo in the Monaro Plains of NSW to undulating country near Yass in NSW, and the whole of the Australian Capital Territory (ACT) (Figure 4). The catchment is approximately 13,144 km^2 in area (Gilmore 2008).

Vegetation varies across the catchment. It is largely undisturbed by human activity in the river's steep rock gorges and in the Brindabella ranges and the Snowy Mountains above Tantangara Reservoir. On the broad river valley flats the vegetation has been mostly cleared and replaced by non-native species. Soils on the steep slopes are often shallow and stony. On the broader slopes of the foothills, the soils are deeper with clay-rich subsoils and deep river terraces near the river (UMDR 2011).

Major streams and tributaries of the Upper Murrumbidgee itself include the Naas River, Paddys River and Cotter River (all entirely within ACT), Molonglo River and Ginninderra Creek (which cross the NSW–ACT borders) and the Numeralla River, Bredbo River, Goodradigbee River and Yass River (all entirely in NSW).

2.1.1 Hydro-climate and water resources

Summers in the catchment are warm and winters are cold, and sites at higher altitudes are generally cooler (MCMA 1998). Table 2 summarises climate data from stations in the catchment.

The average annual precipitation (including snow) for the Upper Murrumbidgee catchment as a whole varies between 1000 and 500 mm (depending on topography and position), and there is no particular wet or dry season (Newham 2002; Gilmore 2008). The Snowy Mountains ranges have a rain-shadow effect on rainfall in the Cooma–Bredbo area (Newham 2002). Records at several locations (except Michelago) near Canberra show a small decline in rainfall and a decrease in inter-annual variability after the mid to late 1980s (ACT Government 2004). However, rainfall in 2010–12 has been above average in the region (BoM 2013). Annual average potential evaporation exceeds annual average rainfall across the entire catchment (Newham 2002).

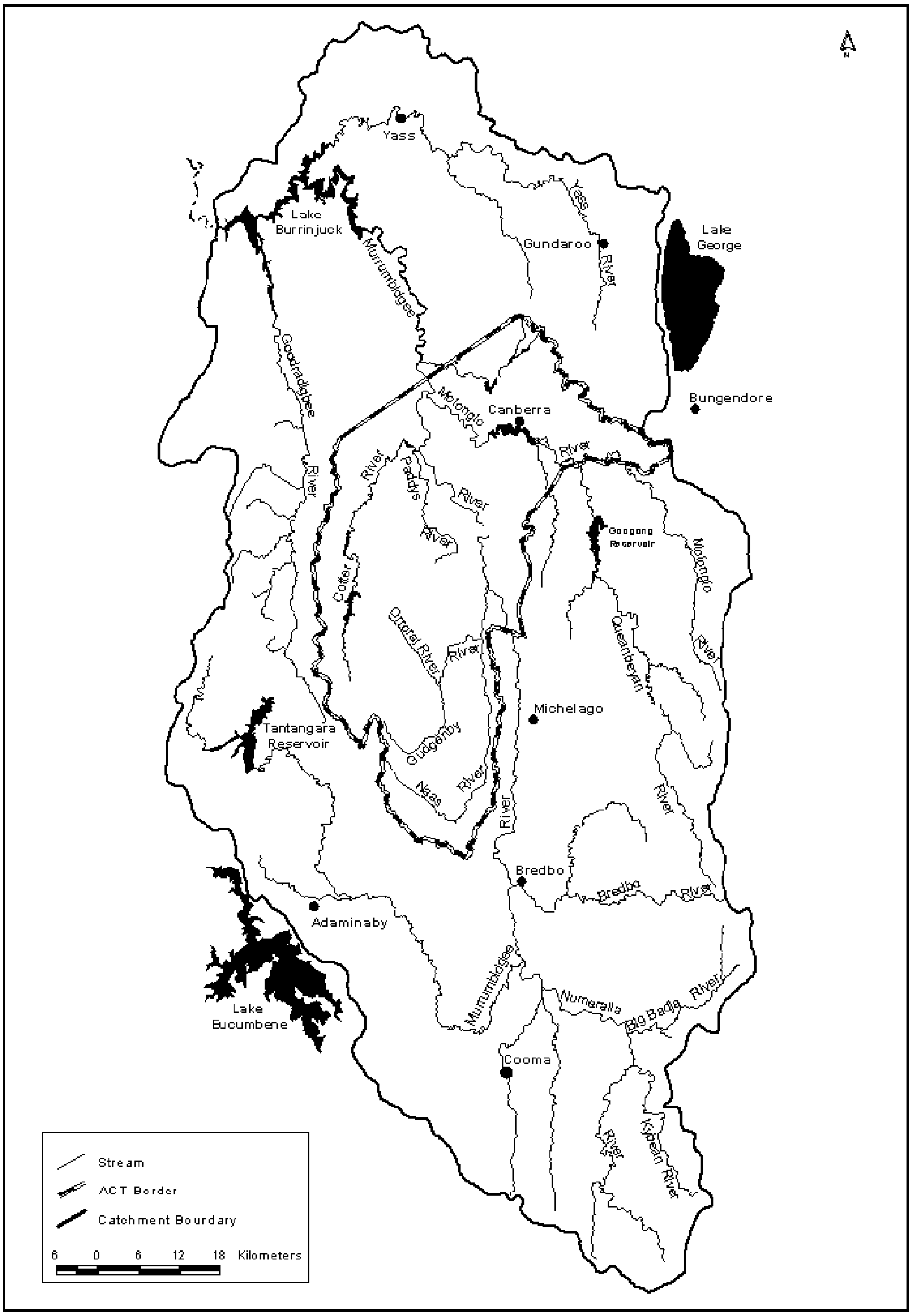


Figure 4. A map of the Upper Murrumbidgee catchment (Linternmans, 2002 © Australian Capital Territory)

Table 2. Average climate data from weather stations throughout the Upper Murrumbidgee catchment (Australian Bureau of Meteorology (BoM))

	Rainfall (mean annual)			Max Temp		Min Temp	
Location, BoM station no. (latitude longitude)	**Rainfall (mm)**	**No. of days of rain**	**No. of days of rain ≥ 25mm**	**Mean daily max temp (°C)**	**Mean annual no. of days ≥ 40°C**	**Mean daily min temp (°C)**	**Mean no. of days ≤ 0°C**
Burrinjuck Dam, 073007 (148.6°E 35.0°S)	925.8	111.1	8.6	20.6	0.5	9.2	7.8
Gudgenby, 070172 (148.9°E 35.75°S)	767.3	94.1	3.7	17.4	0.0	3.0	114.0
Yass – Linton Hostel, 070091 (148.9°E 34.83°S)	651.7	94.1	4.8	20.7	0.3	7.2	46.1
Canberra Airport, 070014 (149.20°E 35.31°S)	616.4	106.2	4.6	19.7	0.1	6.5	59.7
Queanbeyan Bowling Club, 070072 (149.23°E 35.36°S)	594.6	75.6	4.8	20.6	N/A	6.4	N/A
Cooma Lambie St, 070023 (149.1°E 36.23°S)	502.3	91.4	3.0	19.6	N/A	4.7	N/A

The Upper Murrumbidgee catchment provides water resources for the towns of the ACT and surrounding areas — nearly half a million people: Canberra (359,000 inhabitants), Queanbeyan (41,500 inhabitants), Yass (15,000 inhabitants), and Cooma (10,500 inhabitants) (ABS 2011). Most of the ACT's reticulated water supply (60%) is obtained from the Corin, Bendora and Cotter reservoirs in the Cotter River catchment (484 km^2), with supplementary supply from the Googong Dam on the Queanbeyan River, and direct abstraction from the Murrumbidgee River. The catchment's surface- and groundwater resources also provide for irrigated horticulture, such as vineyards, and livestock operations.

Groundwater resources are relatively low-yielding and held in fractured rock such as the Upper Murrumbidgee Fractured Rock aquifer (Goode & Daamen 2008). Recharge is largely by rainfall, and extraction is managed via licensed bores. Within the ACT, groundwater extraction does not exceed the determined sustainable yield. Across the 14 ACT groundwater management areas, from Upper Murrumbidgee to Cotter, a total of 7248 ML is available for use, with water quality similar to that in nearby surface waters (ACT SoER 2011).

2.1.2 Water quality

Water quality in the Upper Murrumbidgee catchment is influenced by geology, climate, hydrology, human activities (agriculture, urban development, urban waste), and bushfires. The quality of water flowing in the Snowy Mountains into Tantangara Reservoir is generally good, with low to moderate total phosphorus concentrations, extremely low total nitrogen concentrations, low turbidity and low concentrations of

dissolved salts (Barlow et al. 2005; SCC 2010). As the Murrumbidgee River flows downstream there is a gradual decline in water quality as the non-point source catchment inputs of sediment, nutrients and salts increase (SCC 2010).

(i) Salinity

Generally, the rivers of the upper catchment have very low salinity. The Yass River catchment and the urbanised Cooma region are exceptions to this (DLWC 1995), with extensive areas subject to dryland salinity (DLWC 1995; Marcot et al. 2001; Yass Shire Council 2012). There are also several creeks in the ACT in which electrical conductivity (EC) values of 300–500 µS/cm have been recorded (DLWC 1995).

(ii) Nutrients and algal blooms

Most streams in the Upper Murrumbidgee catchment generally exceed the ANZECC/ARMCANZ (2000) guidelines in their concentrations of total phosphorus (NSW Government 2010). Even though phosphorus concentrations in Tantangara Reservoir are low to moderate, they are higher than expected for a headwater dam with no agriculture or human settlement upstream. Concentrations remain good upstream of Cooma, but are 2–4 times the concentrations considered low enough to reduce the risk of algal blooms. Most of the upper catchment rivers and lakes in the ACT are in the 'good to fair' category for phosphorus concentration. Exceptions are the Murrumbidgee River downstream of the ACT and the Yass River, where high phosphorus concentrations can occur (DLWC 1995). The Cooma Sewage Treatment Plant is a point source of elevated nitrogen levels (generally above ANZECC/ARMCANZ guidelines) in Cooma Creek, a tributary of the Upper Murrumbidgee (Tuft et al. 2007).

The first serious algal blooms reported for the Upper Murrumbidgee catchment occurred between April and June 1994. *Anabaena* populations of between 1000 and 100,000 cells/mL were recorded from Cooma to Burrinjuck for a week or more at a time (DLWC 1995). Algal blooms now are regularly reported in the summer in Lake Tuggeranong, Googong Dam and Lake Burley Griffin. Most of the ponded waters more than 2 m deep in the Upper Murrumbidgee catchment (including several stormwater-control ponds) stratify in summer, resulting in concentrations of dissolved oxygen declining near the sediment bed, which causes nutrient release from the sediments (DLWC 1995; SCC 2010). Examples where such de-oxygenation has been recorded include Tantangara and Googong Reservoirs and Lake Burley Griffin (DLWC 1995).

(iii) Turbidity

The lower reaches of the major tributaries of the Upper Murrumbidgee River are generally more turbid than their feeder streams, though the waters are generally considered to be of good quality (DLWC 1995). Major flood events in the catchment contribute to periods of elevated turbidity. Bushfires, including the major fire in January 2003 in the Cotter and Goodradigbee catchments, together with drought, 2002–2010, have also contributed to an increase in turbidity after rainfall events in the catchment, as a result of a decrease in ground cover (NSW Government 2010).

(iv) Water quality trends

In the NSW Government 'State of the Catchments' 2010 report, water quality trends for temperature, salinity (EC) and turbidity were identified at two sites in the Upper Murrumbidgee catchment: in the Murrumbidgee River at Mittagang Crossing (due north of Cooma) and in the Goodradigbee River at Wee Jasper (as it enters the Burrinjuck

Reservoir), using 30–40 years of data. Water temperature was stable at both sites. Turbidity was stable at the Murrumbidgee site but increasing at the Goodradigbee site. Salinity was stable at the Goodradigbee site and decreasing at the Murrumbidgee site (NSW Government 2010).

2.1.3 Ecological conditions

The general ecological condition of the Upper Murrumbidgee catchment has been classified as poor (Davies et al. 2008), but there is considerable variation in conditions across the catchment, both from place to place and at various times.

Ecological conditions vary throughout the Upper Murrumbidgee catchment because of the non-uniformly distributed impacts of human activities on water quality (e.g. salinity and sediment concentrations influenced by agricultural work) and flow regimes affected by river regulation (Bowman & Keyzer 2010; Harrison, Norris & Wilkinson 2008; Harrison, Wright & Nichols 2011; RSoER 2009). For instance, salinisation is recognised as a potential stressor for ecosystems in the Yass River and Molonglo River catchments (Bowman & Keyzer 2010; RSoER 2009); while regulation by dams affects ecosystems along the Cotter River (Chester & Norris 2006; Nichols et al. 2006) and the Murrumbidgee River downstream of Tantangara (SCC 2010).

Assemblages of macroinvertebrates in the Upper Murrumbidgee catchment have been found to be in poor to very poor condition in areas of the catchment affected by agricultural, urban or river-regulation pressures (Davies et al. 2008; Harrison, Wright & Nichols 2011). In contrast, macroinvertebrate assemblages within undisturbed areas of the catchment in the Namadgi and Kosciuszko National Parks are in ‘reference’ condition (‘reference’ condition means ‘near-natural’, or not affected by human influences) (Harrison et al. 2011b). Streambed algae (‘periphyton’) also can be affected by river regulation, which in turn can alter macroinvertebrate communities (Chester & Norris 2006; Nichols et al. 2006).

Ecological condition within the Upper Murrumbidgee catchment also changes in response to extreme events such as floods, drought and bushfires. For example, at times of extreme low flow during drought in the regulated Cotter River system, both an increase in periphyton cover and a decrease in macroinvertebrate taxonomic richness were observed (Chester & Norris 2006; White et al. 2012).

The fish community of the Upper Murrumbidgee catchment is severely degraded — i.e. reduced in species, numbers and proportion in relation to pest fish (Davies et al. 2008; Gilligan 2005). Within the catchment there are only 12 native fish species present (Lintermans 2002). The catchment also has nine alien fish species, and seven of these species have established reproducing populations (Lintermans 2002). Introduced fish species dominate the total number of individuals and total fish biomass (Gilligan 2005).

2.1.4 Legislation and regulations

Federal and state-level legislation, regulations and guidelines, as well as local-government policies, govern water resource management in the ACT region and Upper Murrumbidgee catchment. The team identified management policies to be examined during this study by reviewing water planning documents and information about the underpinning legislation and regulations (see the summary in Table 3).

Table 3. Legislation and regulations that govern water resource management in the Upper Murrumbidgee Catchment

	Territory (ACT)	State (NSW)	Commonwealth
Legislation/ Statutory Instruments	• ACT Water Resources Act 1998, 2007 • ACT Environment Protection Act 1997 • Nature Conservation Act 1980 • ACT Planning and Development Act 2008 • Public Health Act	• Water Act 1912 • Water Management Act 2000 • Water Management (General) Regulation 2011 • Environment Planning and Assessment Act 1979 • Snowy Hydro Corporatisation Act 1997	• Water Act 2007 • Murray-Darling Basin Plan • Water Quality and Salinity management Plan • Commonwealth Environment Protection and Biodiversity Conservation Act 1999
Guidelines	• Environmental Flow Guidelines (2006, 2011) • Environment Protection Regulation (2005)- Environment quality guidelines • ACT Government People Place Prosperity		• National Water Initiative • Australian Guidelines for Water Recycling • AUS/NZ Guidelines for Fresh and Marine Water Quality 2000
Strategy	• ACT Natural Resource Management Plan (Bush Capital Legacy) • Think Water, Act Water strategy 2004 • National Capital Plan • Canberra Spatial Plan	• NSW 2021 • NSW Implementation Plan for the National Water Initiative 2006 • NSW Natural Resources Monitoring, Evaluation and Reporting Strategy 2010-2015 • NSW State Groundwater Policy Framework Document • NSW Diffuse Source Water Pollution Strategy 2009 • NSW Algal management Strategy	• Water for the Future • Murray Darling Water CAP • Basin Salinity Management Strategy 2001-2015 • National Water Quality Management Strategy
Policies/ Programs	• Future water options for the ACT region: implementation plan • Future sewerage options review • Water Efficiency (Incentive) Programs • Canberra Integrated Urban Waterways • Urban Water Sensitive Design • Water Management Plan for ACT Sportsgrounds	• Water Compliance Policy • Murrumbidgee Catchment Action Plan Murrumbidgee monitoring programs • Snowy Water Licence October 2011 • Snowy Water Inquiry Outcomes Implementation Deed 2002	

2.2 Scenario definition: water resource management scenarios

As explained in Section 1 of the report, this project aimed to predict impacts of climate change on water quality and freshwater ecosystems by exploring scenarios of future climate, water demand and human activities, via a modelling framework using the Upper Murrumbidgee catchment as a case study or context.

Therefore the first stage of this project was to identify features of the Upper Murrumbidgee catchment and its waters (quantity and quality of flows; water demand) and ecosystems that needed to be considered when developing the modelling framework. The project team:

- ran meetings and a workshop with the project stakeholders, end-users (ACTEW and ACT Government), and field experts to identify the concerns and issues of interest with respect to water quality and ecological systems in the Upper Murrumbidgee catchment (Dyer et al. 2011);
- conducted desktop analysis of available literature relevant to the study area and issues to be addressed in the project;
- inquired into data sets and models available for use in constructing scenarios.

From the findings from these activities, three categories of driving factors ('drivers') emerged: (1) climate conditions, (2) urban water demand, and (3) adaptation policies. (From here on, these three 'categories' are termed 'dimensions'.) Changes in these three dimensions in the Upper Murrumbidgee catchment could affect water quality in the future.

To test that possibility, the team identified a series of related questions that the study should try to answer.

(i) In relation to hydro-climate, what would be the impacts of changes in precipitation and temperature on water quality attributes?
   - How would changes in stream flow regimes affect water quality attributes?
   - How would change in temperature affect water quality attributes?
   - How would the occurrence of bushfire events alter runoff and stream flow regimes?

(ii) In relation to urban water demand, what would be the impacts of changes in water demand on water quality attributes?

(iii) In relation to adaptation policies, what would be the combined impacts of a set of management adaptation alternatives (i.e. supply, demand management, and aquatic protection measures) on stream flow regimes, and on biological and chemical water quality attributes?

The next stage was to construct a range of scenarios that would combine likely climate conditions and possible intensities of water demand by the region's residents.

2.3 Scenario construction

Having identified key drivers influencing water quality and dependent ecosystems in the catchment, the second step in scenario construction was to conceptualise how these

drivers affect the system, and then to select and define the variables, data sets, and analytical models and methods that we would use to quantify scenarios.

We recognise that climate is a driver that interacts with other driving factors, including population growth, community preferences and management policies. Therefore, overall in this project we have taken a systems-based approach which assesses future impacts by considering the complex interactions among:

(1) direct and indirect climate impacts on chemical and biological water quality attributes,

(2) impacts of non-climate pressures, such as population growth, and

(3) impacts of multi-scale adaptation decisions.

Below, we discuss the dimensions we identified above, which we later quantified when producing the water resource management scenarios — which, in this report, are called 'management adaptation alternatives'. (Appendix B, Table B1 characterises each (single) scenario in terms of its underlying drivers, assumptions, input/model/output, and uncertainty.)

2.3.1 Hydro-climate and bushfires

(i) Climate as a factor modifying water quality and ecosystems

Climate is one of many drivers that influence water regimes. Yet there is almost a consensus among researchers and policy makers that future climate is the 'biggest unknown'. Climate variations (including climate change and variability) are the most substantial and challenging factors to be considered in water planning and biodiversity protection. Climate exerts direct and indirect pressures on (almost) every aspect of the water system: supply, demand, water quality and ecological systems. It is the crucial 'switch' for most disturbances (e.g. fire, drought, flood, pollution), and it also serves as a catalyst for other impacts.

Nevertheless, there is much uncertainty about the magnitude and nature of future changes in the climate regime (average, inter-annual variability and extreme events for rainfall and evaporation) as well as consequent impacts. For example, climate change imposes severe risks to the hydrological cycle and other related natural phenomena whose effects extend far beyond human experience and knowledge. Some actual or expected climate impacts include:

- droughts — which may become more frequent and severe;
- dry soil conditions — which may follow decreased rainfall (especially in autumn) combined with increased evaporation;
- less stream flow — which may result, disproportionately, if there is decline in rainfall;
- bushfires, runoff, sediment — increase in temperature, especially during extreme weather and dry catchment conditions, may increase the frequency and intensity of bushfires along with the severity of their impacts on runoff and sediment regimes;
- risks to freshwater ecosystems — increased frequency and duration of heatwaves may pose severe risks to freshwater ecosystems;
- increased water demand — higher air temperatures, combined with decreased rainfall, may lead to increase in water demand and exert pressure on the storage reservoirs, especially in the ACT where domestic irrigation constitutes most of the ACT water demand.

The South Eastern Australian Climate Initiative (SEACI) has found that recent drought (2002–10) in the southern Murray-Darling Basin and Victoria is unprecedented by other recorded droughts since 1900, in:

- being largely contained to the southern Australian region;
- having lower year-to-year rainfall variability, with no wet years over the dry spell;
- the maximum rainfall decline being observed in autumn, and there were also losses in winter and spring as in previous droughts;
- air temperatures steadily rising.

The drought's impact on flow in the Murrumbidgee River is shown in Figure 5, recorded at Mt McDonald, near where the Murrumbidgee flows out of the ACT towards Yass.

Findings from SEACI and other programs suggest that[1]:

- observed changes may be (at least partially) linked to climate change;
- a shift in hydro-climatic conditions is expected, most likely towards conditions being drier and warmer than the long-term historical average;
- given that natural variability will still contribute to climate conditions, some wet periods can be expected in the short term.

(ii) Quantifying climate for the scenarios

To assess climate impacts on stream flow regimes, and other dependent systems (socio-economic and bio-physical), we obtained daily hydro-climate data (historical and projections) from SEACI[2]. SEACI uses daily scaling methods to scale historic data to outputs from 15 different global climate models (GCMs). SEACI uses two global warming scenarios: 1°C or 2°C increases in global average surface air temperature (Chiew et al. 2009); and it uses SIMHYD rainfall–runoff modelling to generate daily runoff series.

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1 For current knowledge on climate and related research programs for the ACT, see Webb 2009, 2011

2 SEACI data are available as 0.05o grid cells. Shape files for the Upper Murrumbidgee catchment were used to aggregate data at sub-catchment scale.

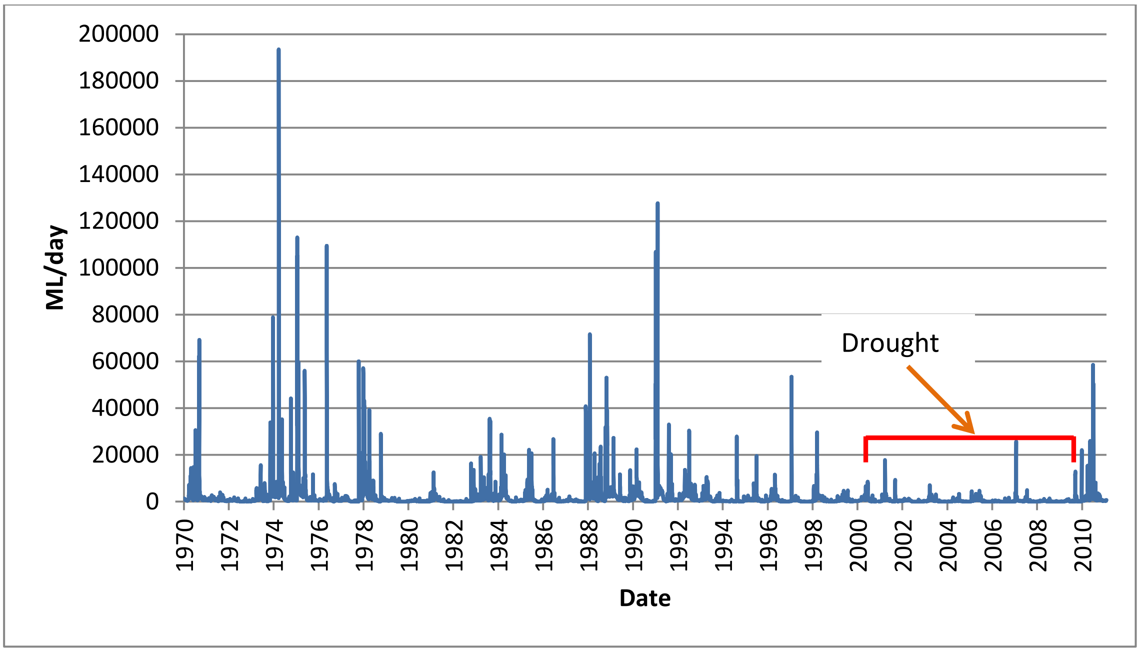


Figure 5. Daily discharge volume (ML/day) for the Murrumbidgee at Mt McDonald (1970–2011), showing the low discharge during the drought of 2002–10

(iii) Bushfire effects on water quality and stream flow

In the future, it is anticipated that risks of bushfires will increase if the region's climate becomes drier and warmer. Hennessy et al. (2005) investigated how the changes in climate conditions will affect the likelihood of fire occurrences in South-eastern Australia by 2020 and 2050. They found an increased risk of fire at most sites (including the Upper Murrumbidgee catchment). They suggested there will be an increase in the average number of days when the Forest Fire Danger Index (FFDI)[3] is "very high" or "extreme" (reference year 1990: 4% to 25% by 2020, 15–70% by 2050). Using improved climate projections, Lucas et al. (2007) updated the findings from the 2005 study to include FFDI of "very extreme" and "catastrophic". They predicted an increase in the average number of "very extreme" and "catastrophic" days by 2020 and 2050.

There has been a history of devastating fires in the Upper Murrumbidgee catchment. Pryor (1939) reported that severe bushfires had occurred in the area since the European settlement and in the period 1860–80. Table 4 lists major fires (area burnt >2600 ha) reported in the Cotter catchment since the beginning of the twentieth century (Carey et al. 2003).

[3] McArthur Forest Fire Danger Index (FFDI) is used throughout Eastern Australia to indicate the fire risk where (McArthur, 1967): (FFDI>25%) indicates "very high" risk, (FFDI>50%) indicates "extreme" risk, (FFDI>75%) indicates "very extreme", and (FFDI>100%) is catastrophic.

Table 1: Major fires (area burnt >2,600ha) reported in the Cotter Catchment since the beginning of the 20th Century. Source: Carey et al. (2003)

Year	Fire description
1920	Severe and extensive fire across the centre of the ACT and into NSW.
1926	Lighter winds allowed it to be contained at a fire trail along the Tidbinbilla range.
1939	Very hot days with low humidity and high winds exacerbated the fire, resulting in spot-fires up to 24kms ahead of the fire-front. The fire affected central parts of the ACT and the north-western ACT/NSW border.
1951-52	Resulted from lightning strikes and power line failures, and a severe fire weather year. About 20,000ha of grasslands were burnt in the ACT and NSW.
1979	Affected a large area crossing the north-eastern ACT/NSW border.
1983	Followed a severe drought and burnt a larger proportion of the southern part of the ACT.
1985	Burnt areas in NSW just across the north-eastern border of the ACT.
2001	Affected a small region, close to the Canberra area.
2003	The extremely dry conditions in the catchments, combined with strong wind and lightning strikes, caused unprecedented bushfires. As a result, almost 98% of the Cotter catchment was burnt out. In the few months after the fire, intense rainfall events washed out fire debris and sediment into the reservoirs, which caused major water quality problems.

The frequency and intensity of bushfires are influenced by a set of interactive factors that vary over short and long terms (Finkele et al. 2006). These factors include (but are not limited to): vegetation cover, soil dryness, fuel load, extreme weather conditions (e.g. low humidity, high wind speed, and temperature), and random ignition events (e.g. human activities).

Bushfire is another important uncertainty affecting predictions of catchment and river health. A bushfire event causes immediate and long term impacts on the catchment and stream flow quality and quantity, which are highly dependent on the severity and extent of the fire and the proximity of the fire to the streams. Observed impacts include:

- an initial increase in runoff caused by a loss of vegetation (it may be combined with extreme weather events such as were experienced after the 2003 bushfire) which may lead to increase in stream flow (Fernandez et al. 2006; Watson et al. 1999);
- an initial reduction in water quality caused by large amounts of fire ash and debris being input to the streams (Minshall 2003);
- a short-term or medium-term degradation of water quality caused by increased erosion and inputs of sediment (Wilkinson et al. 2006);
- a long-term reduction in stream flow caused by vegetation regrowth (Langford 1976; Watson et al. 1999).

ActewAGL and ANU (FIRESCAPE) have modelled the likelihood of future fires in the Cotter River catchment. ActewAGL compared two modelling approaches and found the results were broadly in agreement (ActewAGL 2011b).

To incorporate bushfire effects within the modelling framework we used ACTEW's available fire triggering and vegetation modelling to capture the hydrological effects. This approach generates a stochastic fire sequence when particular hydro-climatic conditions are met (e.g. inflows are below a certain threshold). The fire model is coupled with a vegetation model to simulate the impacts of a fire event on land cover and catchment yield. The direct effects of fire on water quality and ecological characteristics can be captured by including the fire frequency in the Bayesian Network. This then directly links to water quality and ecological attributes.

2.3.2 Water demand

As the team identified in the pilot project, urban-style demand for water is also a key driving factor in water management in the case study catchment. Size of population and water use per head are major components of water demand. In this catchment the demand for water that need not be potable is larger than the demand for potable water.

Here we discuss aspects of population growth and water use in relation to scenario construction. We defer discussion of demand management and effluent treatment capacity to the next subsection on management adaptation alternatives.

Human demand for water is a major driver in relation to water security and river health. First, water demand puts direct (physical) pressure on water availability and storage levels. Second, the quantity and quality of effluent discharged after use of the water affects river health. Third, water demand puts (societal and political) pressure on government and water authorities to increase supply, either by acquiring new water sources (e.g. build a new dam) or by extending the existing capacity (e.g. increase the capacity of the Cotter dam). Given that socio-political impacts are external to the scope of this project, we focused on quantifying the impacts of water demand in terms of water availability and effluent discharge, rather than defining specific policy outcomes.

To calculate water demand and effluent discharge, it was essential to: (1) predict future population size, (2) estimate per capita water use, (3) account for reduction in demand in response to demand management, and (4) consider the capacity of the treatment facility.

(i) Population scenario

Population growth puts increasing pressure on water security. In their seminal book "Limits to Growth", Meadows & Randers (1972) warned that population growth would exceed the carrying capacity of the planet, causing an over-degradation of natural resources and collapse of dependent ecological and socio-economic systems. For the ACT, El Sawah (2010) examined the impacts of structural changes (such as efficient water fixtures and appliances), behavioural changes (such as shower time), and population growth on water demand and water security. Findings suggested that population is the "elephant in the room" which may eventually push water demand beyond sustainable limits.

In 2008, the Australian Bureau of Statistics published three population growth scenarios for the ACT and surrounding region: high (Series A), medium (Series B), and low (Series C) (ABS 2008). These projections included the planned new developments at Tralee and Googong in NSW, near Queanbeyan. In the last three years, observed population growth rates have exceeded the ABS's high growth projections. In water planning, ACTEW has based its water supply strategy on the high growth scenario. There is also a possibility that ACT will eventually supply water to a number of nearby towns in NSW, such as

Yass and Murrumbateman, to provide a secure water supply to these growing regional centres (see Figure 6).

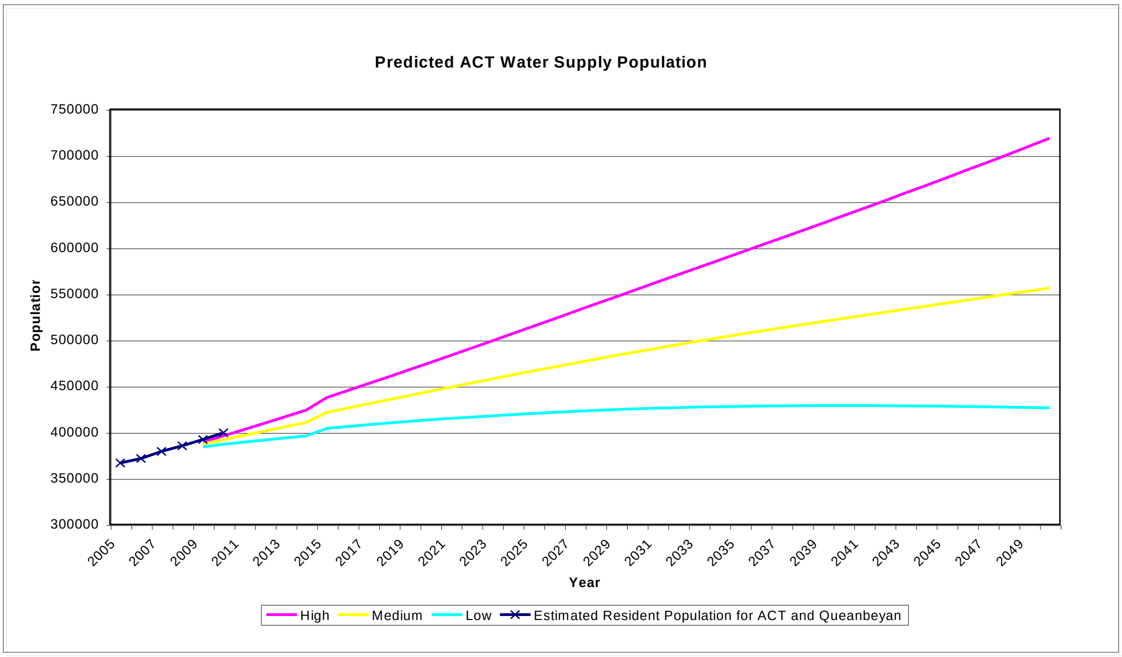


Figure 6. Projected population for the ACT region water supply (ACTEW, 2011) xxx = estimated resident population for ACT and Queanbeyan, pink line (top) = high growth; yellow line (middle) = medium; teal line (lowest) = low.

To construct a population scenario, we assumed that:

1. population growth will follow the high level growth scenario as defined by the ABS.
2. in year 2015, the ACT will commence to supply water to neighbouring towns. In the scenario, the population served across borders is:
   - zero until 2015, and
   - equivalent to an additional 1.6% growth of the ACT population (Figure 6).

(ii) Per capita water use scenario

After Perth and Adelaide, the ACT has the third largest per capita consumption in Australia (SEWPaC 2006). In the ACT, demand is mainly driven by household consumption. As shown in Figure 7, 54% of water demand comes from the household sector. About 43% of household consumption (i.e. approximately 24% of total consumption) is used for irrigation, reflecting the culture of English-style lawns which is dominant in the region (Head & Muir 2007).

As a part of demand modelling, ACTEW has developed a regression model to estimate “unrestricted” per capita water use as a linear function of Canberra Airport rainfall and evaporation. The modelling approach separates out the impacts of water restrictions (and other demand management policies) on consumption levels, and estimates “only climate-driven” per capita water use. For this, consumption data from January 1993 to November 2002 are used to calibrate the model’s parameters. Figure 8 shows there was a shift in water demand after water restrictions were introduced in late 2002. (Since

November 2011, in the present wetter conditions, ACTEW has asked ACT residents to observe Permanent Water Conservation Measures, instead of water restrictions.)

(iii) Land-use in relation to water demand and effluent

Projected land-use changes in the Upper Murrumbidgee catchment include an expansion of urban areas and an increase in peri-urban and rural residential development. Changes in land-use can also have a significant effect on the quality and quantity of water derived from an area, as part of overall water supply.

With the increase in peri-urban and rural residential development, there is typically an increase in the number of 'farm dams' constructed to provide stock and domestic water supply. Farm dams capture rainfall and runoff. Depending on the number and density of dams in a catchment, they can reduce the runoff to water supply rivers and storages.

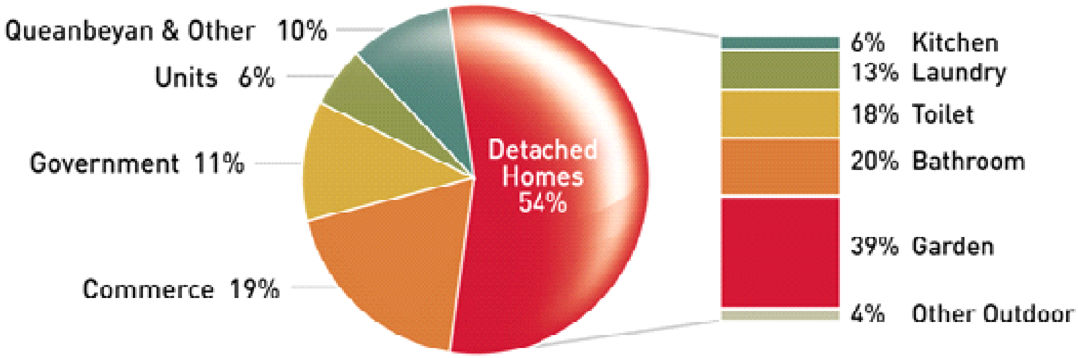


Figure 7. Distribution of the ACT consumption by category (Printed with permission from ACTEW Corporation)

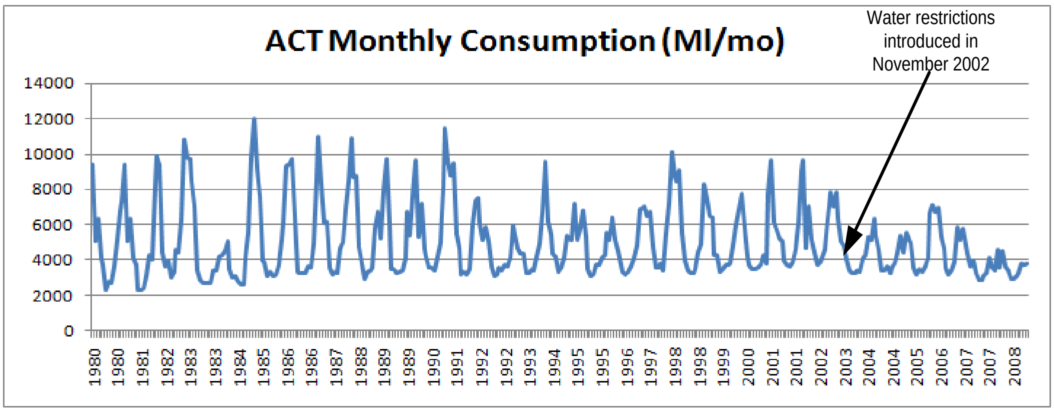


Figure 8. A shift in water demand after water restrictions were imposed in November 2002

Schreider et al. (2002) suggested that farm dams have very small effects (<5%) on yield. Also, Starr (2006) suggested that, in future, it is unlikely that the combination of circumstances will recur that has produced farm dams in the Googong catchment. Therefore we chose not to include changes in the numbers of farm dams as part of the modelled management adaptation alternatives.

Increasing peri-urban development and settlement also increases the use of On Site Systems of Sewage Management (OSSMs — which are individual property's processing facilities such as septic tanks or aerated water treatment systems, plus an effluent disposal area). In a review of existing water quality in the western Palerang area it was concluded that OSSMs represented the main risk to water quality for the region (Holloway, Masterman-Smith & Plumb 2011). Further risks were posed by road crossings over waterways, and erosion of gullies and streambanks.

The water quality attributes of greatest concern were identified as suspended solids (turbidity), followed by nitrate, pathogens, phosphate and other chemicals. There is little information available for the Upper Murrumbidgee catchment on projected population increases that would result in an increase in the numbers of OSSMs and thus the magnitude of the future risks for the region. Given that Starr (2006) suggests that it is unlikely that the combination of circumstances will recur that have produced farm dams in Googong catchment, we have considered that it is also unlikely that there will be a similar expansion in the number of OSSMs, and we chose not to include changes in the number of OSSMs in the modelled management adaptation alternatives.

2.3.3 Adaptation policies

The Intergovernmental Panel on Climate Change (IPCC 2001) defines adaptation as an ‘‘adjustment in ecological, social, or economic systems in response to actual or expected climatic stimuli and their effects or impacts”.

In this project, we focused on adaptation policies. Adaptation policies are context-specific and subject to the legislative and regulatory framework of the water system (summarised in Table 3 for the case study catchment). They emerged from the pilot study (scenario definition stage of the project) as a key dimension or set of driving factors that could affect water quality and ecological responses with future climate change.

The discussion below indicates matters the team considered in devising four combinations of ‘management adaptation alternatives’ (or ‘conditions’), for testing in this project (listed in Figure 10).

To identify adaptation decisions, we reviewed available literature (e.g. policy/regulatory documents, peer-reviewed publications, reports) and conducted a series of meetings with the project stakeholders. Based on the findings, we grouped policies into three categories (see Figure 9):

- supply management,
- demand management (including approaches for conservation and source substitution),
- water quality and aquatic systems protection.

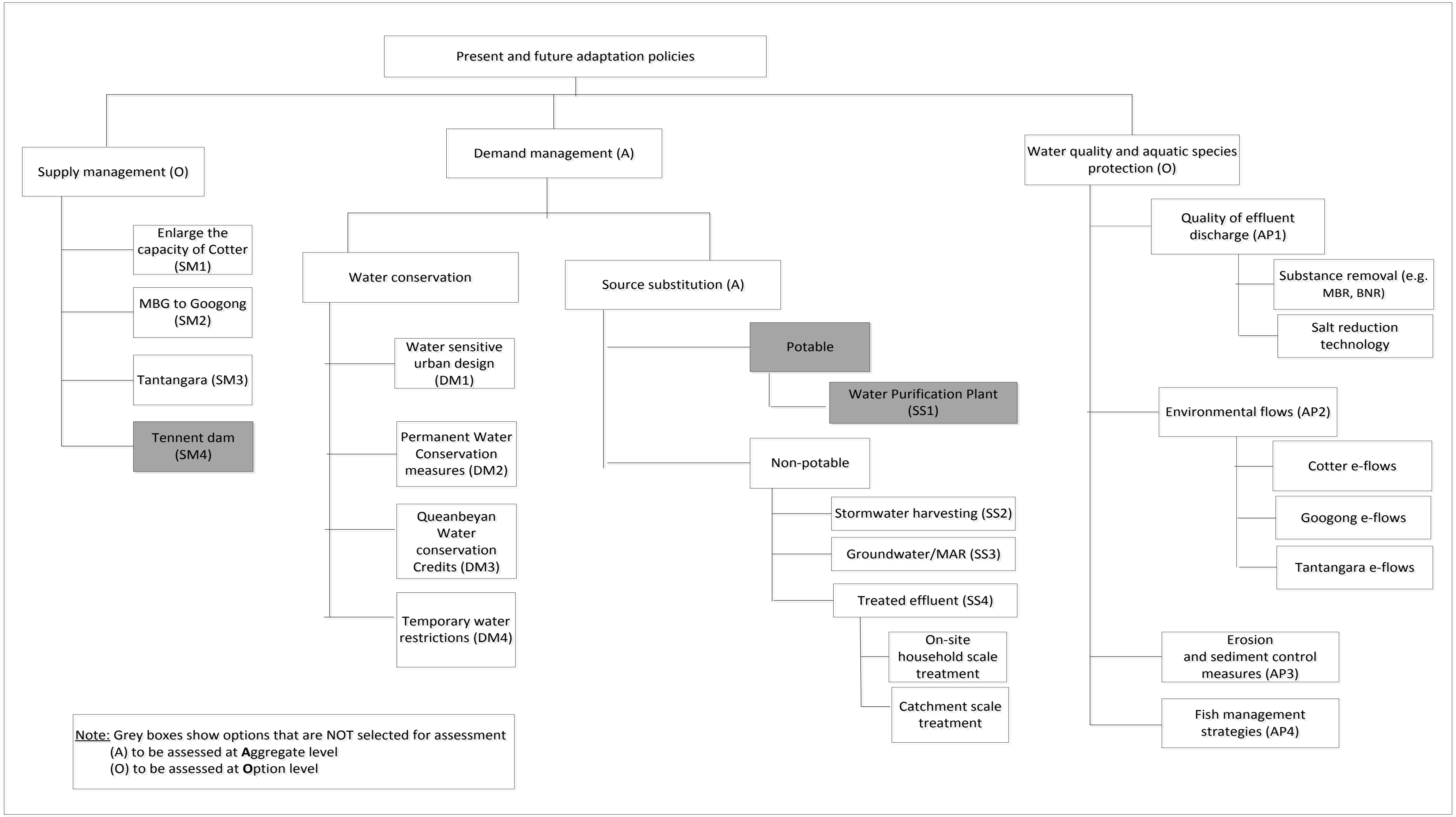


Figure 9. Categories of present & future adaptation policies

(i) Supply management

Supply management refers to the installation of infrastructure (e.g. dams, pipelines, pumping stations) to acquire water from new sources (e.g. build a new dam) or extend the capacity of existing sources (e.g. enlarge the capacity of a dam). Thanks to the expansive water supply infrastructure in the Upper Murrumbidgee catchment, the quality of the water supply to ACT and the region has been sufficiently high since 1967, with the exception of intermittent water supply shortage (Dovers et al. 2008).

After the drought of 2002–10, along with the looming impacts of climate change and a growing population, ACTEW pursued a number of new options for water supply (summarised in Appendix B, Table B2), including:

- enlarge the Cotter Dam (ECD) capacity from 4 GL to 78 GL,
- pump water from the Murrumbidgee River to Googong Dam (M2G),
- purchase water from NSW irrigators to be delivered from Tantangara Reservoir,
- construct the Tennent Dam.

Now, in 2013, ECD and M2G are already being implemented, and the purchase of water from NSW irrigators (the Tantangara option) is expected to begin by 2014. (It is included in our set of ‘management adaptation alternatives’ in Figure 10.) At present, Tennent Dam is rated “low” on the water planning agenda because of its high economic, social, and environmental implications. Hence, Tennent was not considered for assessment in this study.

(ii) Demand management: water conservation and source substitution

Since the drought and bushfire experience of the last few years, the ACT Government has set targets of 12% reduction in per capita consumption by 2013 and 25% reductions by 2023 (ACT Government 2004). To achieve the target, a number of policies and programs have been set up, such as water sensitive urban design for stormwater management, and permanent conservation measures for demand management. In Appendix B, Table B3 is a summary of water conservation programs.

Source substitution (Appendix B, Table B4) includes options to supply demand for potable or non-potable water from alternative sources such as rainwater tanks or grey water capture. ACTEW has explored the option of treating sewage water to produce high quality drinking water (rigorously via a Water Purification Plant (WPP)). There are many factors that will determine the viability of implementing this option, such as high energy and economic costs and poor community acceptance of the idea of drinking treated water. So, at the present time, a WPP is seen as a “ready card” in case of dramatic change in circumstances (e.g. severe reductions in catchment yields).

There are two levels of non-potable water: household scale (e.g. rainfall tanks, shower buckets), and catchment scale. At a catchment scale, the current sewerage network in Canberra includes the Lower Molonglo Water Quality Control Centre (LWMQCC), in west Belconnen, which is the main sewage treatment plant for Canberra and the surrounding area. The Fyshwick Sewage Treatment Plant (FTP) stores industrial and domestic sewage before its controlled release into the sewer main which flows either to the LMWQCC or to the North Canberra Water Reuse Scheme.

We considered two approaches for constructing demand management scenarios. In the bottom-up approach, water saving per option would be used to estimate total water saving for a set of options. The alternative approach would be to identify whether reduction targets have been met or not, regardless of "how" they are achieved. The bottom-up approach allows for analysing and comparing the costs and benefits (socio-economic) for a suite of options. Although this was not relevant to the aim of this project, we modelled demand management as achieving (or not achieving) a target.

ACTEW's baseline scenario postulates that demand reduction targets will be met. The underpinning assumption is that there has been a permanent shift towards water-wise behavioural patterns. This assumption overlooks the changes in the factors that influence decision making, and therefore, their impacts on water use patterns (e.g. perceptions of climate change and weather conditions). Therefore, in this project we challenged that assumption and examined "what-if" demand reduction target are not met.

(iii) Water quality and aquatic ecosystems protection

(a) Quality of effluent discharge

The LMWQCC discharges treated effluent to the Molonglo River; after a very short distance the Molonglo River enters the Murrumbidgee River at a flow rate of about 29 GL/year (or 80 ML/day). The discharged flow from the treatment plant has crucial environmental, social and economic value, because:

1. it significantly contributes to flows in the Murrumbidgee River, especially during drought periods (about 15% of river flow during 2002–08);
2. it contributes to flows in the River Murray and the Murray-Darling Basin, and therefore to water use downstream (for irrigation and urban supply);
3. its water quality contributes to the quality of water in the Murrumbidgee River and River Murray, and therefore affects dependent systems (both ecological and agricultural);
4. it constitutes a component of the ACT's net water use under federal legislation and agreements.

The Canberra Sewerage Strategy 2010–2060 (ActewAGL 2011) has explored a range of future options for upgrading the sewerage system at the Fyshwick treatment plant and LMWQCC. Options were identified and assessed (qualitatively) based on a range of criteria (economic, social, and environmental). For this project's scenarios we examined, selected, and grouped options that were most relevant for their impacts on water quality and ecosystems. For this purpose, we grouped options (according to their function) into:

- technologies for substance removal, such as Membrane bioreactor and Biological Nutrient Removal,
- technologies to reduce salt, such as reverse osmosis.

(iv) Environmental flows/releases

River flow is a driver that limits the distribution and abundance of river species and regulates the ecological integrity of flowing water systems (Poff et al. 1997). Infrastructure such as dams or weirs affects the natural flow of water downstream. Water released from reservoirs to flow downstream as 'environmental flows' helps to restore the ecological processes and biodiversity of water dependent ecosystems.

Within the ACT, the Environmental Flow Guidelines (ACT Government 2006) established by the *Water Resources Act 1998*, define volumes and timing of environmental flows, and set abstraction limits for streams, rivers, lakes and aquifers. Environmental flows are adapted to the type of water system (natural, water supply, modified, or created) and for special situations such as drought periods. For instance environmental flows specified for water supply catchments (the Cotter and Googong catchments) specify a minimal requirement for healthy aquatic ecosystems to ensure that both water supply and conservation objectives can be met. Conversely, environmental flows in natural ecosystems, such as those within Namadgi National Park and Tidbinbilla Nature Reserve, are designed to protect the base flow [4] and also protect most of the volume of flood flows that are necessary to maintain the channel form[5] (ACT Government 2006).

Within the scenario planning for this project we considered that there were three regulated ‘sub’ systems in the Upper Murrumbidgee catchment for which environmental flows were relevant: Tantangara, Cotter and Googong Reservoirs.

Tantangara: Releases from Tantangara are regulated by the Water Administration Ministerial Corporation (WAMC) by advice of the Snowy Scientific Committee (SSC) under the Snowy Water license. In 2011, the SSC made environmental flow recommendations for release from Tantangara Dam to the Upper Murrumbidgee River. Environmental flow recommendations for the Murrumbidgee River between Tantangara Dam and the ACT border were established by an Expert Panel (Pendlebury et al. 1997) and reviewed by the SSC (2010).

Generic ecological objectives for the environmental flows are defined by the Snowy Water Inquiry Outcomes Implementation Deed (SWIOID 2002, Annexure Two), as:

a) to protect endangered / threatened species,

b) to maintain natural habitats, and

c) to maintain wilderness and national parks values.

The SSC has advised on environmental water releases recently to support breeding of Macquarie Perch.

Cotter River and ECD: Dams on the Cotter River are classified as water supply ecosystems (ACT Government 2006) and these storages are managed to ensure that there is adequate supply of water for consumption while maintaining the ecological health of the rivers. The ecological objectives for the Cotter River include maintaining:

a) populations of Macquarie Perch between Bendora Dam and Cotter Dam;

b) populations of Two-spined Blackfish between Corin Dam and Cotter Dam; and

c) healthy ecosystems in the Cotter River catchment between Corin Dam and Cotter Dam.

Googong Dam: The Environmental Flow Guidelines (ACT Government 2006) specify the environmental flows in NSW immediately downstream of Googong Dam. These flows are under the direct control of the ACT through regulation of releases. At the time of development of the Environmental Flows Guidelines (ACT Government 2006) NSW had

[4] Base flow: the minimal volume of water that the stream needs to support the fish, plants and insects and protect water quality.
[5] Channel maintenance flows ensure the river maintains its natural channel form.

not established environmental flow requirements in streams upstream and downstream of the ACT. For releases from Googong Dam, environmental flow requirements for water supply ecosystems are applied. The ecological objective below Googong Dam is to maintain healthy aquatic ecosystems (a base flow below Googong of 10 ML/day or inflow, whichever is less, and 4 ML/day during drought periods).

(v) Environmental water quality

Water in the deeper layers of stratified reservoirs can have a much lower temperature and oxygen content than surface waters. If released as an environmental flow, this 'cold water' may severely disrupt spawning, migrations, and reproductive activity of animals downstream. While most of the focus on releasing deeper water is on the cold water pollution it creates (Preece 2004), deeper water may also contain higher concentrations of metals and nutrients than the surface waters.

Corin, Bendora, Cotter, Googong and Tantagara Dams all have the capacity to release water from a variety of depths (through multi-level off-take towers) and so inflow and release water temperature can be matched. However, Scrivener Dam which contains Lake Burley Griffin on the Molonglo River, which is managed by the Commonwealth, does not have a multi-level off-take (ACT_Government 2006). In the case of the Cotter River, water releases from all the reservoirs can be sourced from multi-level off-take towers to minimise potential impacts of cold water pollution on the aquatic ecosystem, particularly on native species as Macquarie Perch (ACTEW 2010).

2.4 Output scenarios

Based on information in the discussion above, for the case study catchment, the team:

(i) defined a range of plausible scenarios for future climate and stream flow conditions,

(ii) combined those scenarios with several scenarios of future water management and water use situations.

For all four alternatives we assumed that the region's population will go on expanding steadily, and that the region's long history of bushfires will continue. With those two assumptions constant, we varied (as 'do' or 'do not')

(i) control of human water demand in the ACT,

(ii) supply of water available, for humans, and

(iii) boosts to the flow regime, for river ecosystems.

The result was the proposed four plausible (flow-driven) 'conditions' or 'management adaptation alternatives', shown in Figure 10.

The four 'management adaptation alternatives' in Figure 10 could be paraphrased as:

C1, Manage in the future within the currently available limits of water supply;

C2, Maximise the future water available for both humans and the flow regime;

C3, Moderately increase the future water available to meet human demand, but do not adjust the flow regime;

C4, Manage within the currently available limits of water supply for humans, but boost the flow regime.

At the next stage of the project the team quantitatively examined how these alternatives would affect stream flow in the 30 climate scenarios we had devised (see Appendix Table C1). Section 6 illustrates how we assessed their impacts on water quality and aquatic protection options.

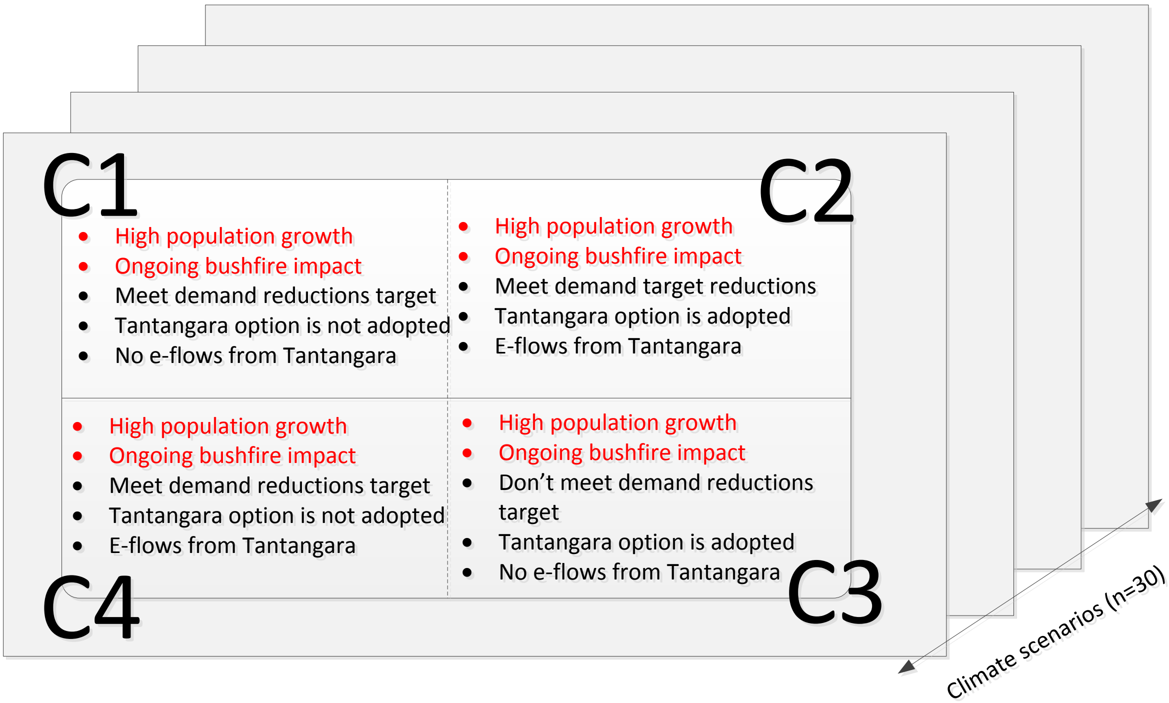


Figure 10. Four plausible (flow-driven) management adaptation alternatives to be examined under a range of climate scenarios.

3. SCENARIO ANALYSIS

This section describes the methods used to predict the flow regime changes within the catchment, including the selection of the climate scenarios used during the remainder of the study. The predicted flow regime changes are analysed using hydro-ecological indicators of change and compared with the effects of river regulation. The linking of flow regime changes to water quality is described, including the water quality modelling approaches. The section concludes with predictions of exceedance probabilities for water quality attributes designed to protect ecological communities.

3.1 Flow modelling

The first step in linking the climate attributes to water quality and ecological response models is to generate flow regimes for each of the climate and management scenarios of interest. Flow regimes were generated for all possible climate scenarios and those displaying minor, moderate and major changes in ecologically relevant attributes of the flow regimes were selected for further analysis.

3.1.1 Climate and "natural" flow data

As described in Section 2, historical climate data and future projections were obtained from the SEACI database. Historical daily rainfall and potential evapo-transpiration (PET) data extend from 1895 to 2008. We used climate projections that represent outputs from 15 Global Climate Models (GCMs) for the A1B emission scenario at 1^{o}C and 2^{o}C increases in atmospheric temperature, which yielded a total of 30 scenarios (see Appendix Table C1). SEACI uses an empirical daily scaling method to downscale climate predictors from catchment scale rainfall and PET. The scaling method considers changes in the future mean seasonal rainfall, PET and daily rainfall distribution.

To generate runoff time series we used the lumped conceptual daily rainfall–runoff model, SIMHYD, with a Muskingum routing, to estimate daily runoff as gridded data at (~5 km × 5 km) resolution. The model was calibrated against 1975–2006 daily stream flow data (Chiew et al. 2009).

We aggregated the SEACI flow estimates for all cells within each catchment. This gave an estimate of the input to flows at each selected site in the region. To convert these inputs to flows, we aggregated the input flows to each selected site. Comparison of the aggregated flow estimates with observed flows at gauged sites showed no significant pattern, indicating that there was no need for the addition of a routing model. This implies that at the scales being considered in this study, the routing of water represented in the SEACI flow estimates (5 km x 5 km grid cell) dominates over the routing through the Upper Murrumbidgee catchment.

Flow estimates were produced for all sites under "natural" conditions (i.e. assuming no dams or regulation were present in the catchment). Further, groundwater–surface-water interactions add complexity to the routing of flows through transmission losses and the addition of baseflow to the river. This can lead to an error in the volume of flows as well as the temporal distribution of flows. To estimate the uncertainty in the flows at ungauged locations we compared the resulting flow values with observed flows at gauged sites to assess the accuracy of the modelled flows.

The high correlations between observed and modelled flows (e.g. Figure 11) indicate that for most catchments, the SEACI gridded data reproduce the temporal pattern of flow

at the outlet of the study catchments. In the case of gauge 410033, deconvolution (Figure 12) shows that a lag-route routing method was able to capture the difference between the observed and aggregated SEACI flow values, using a time constant of 0.7 days, though there was considerable uncertainty in this value; the time constant obtained for gauge 410050 (about 20 km downstream of gauge 410033) was significantly higher, at 1.2 days. The high residuals at negative lags indicate the presence of timing errors in the SEACI modelled flows, most likely a result of errors in the input rainfall data.

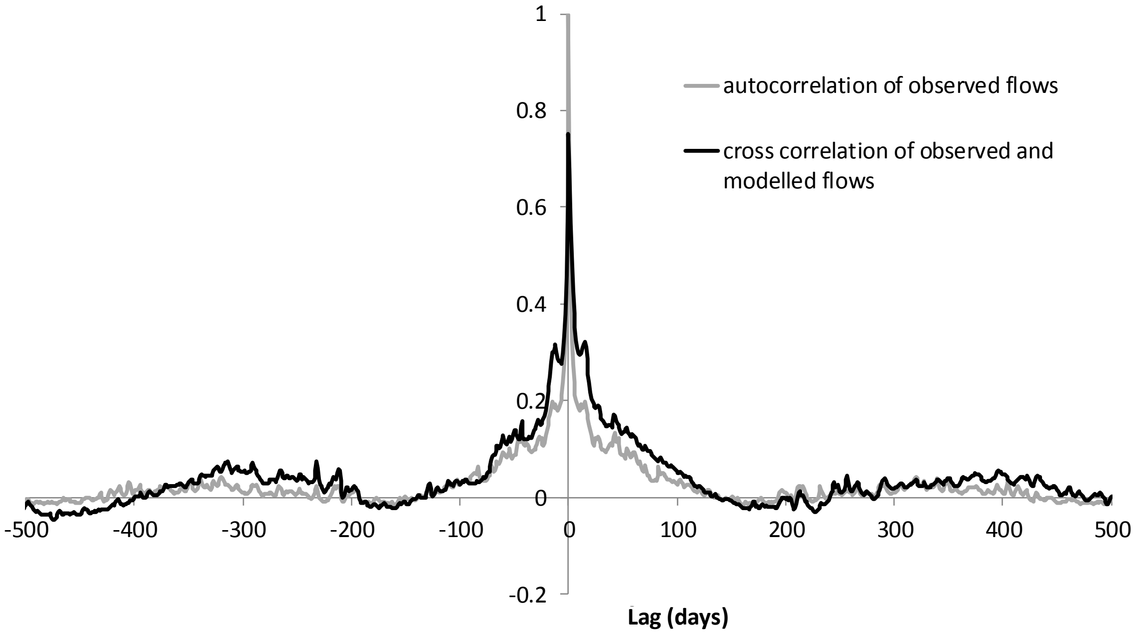


Figure 11. Cross correlation analysis for gauge 410033

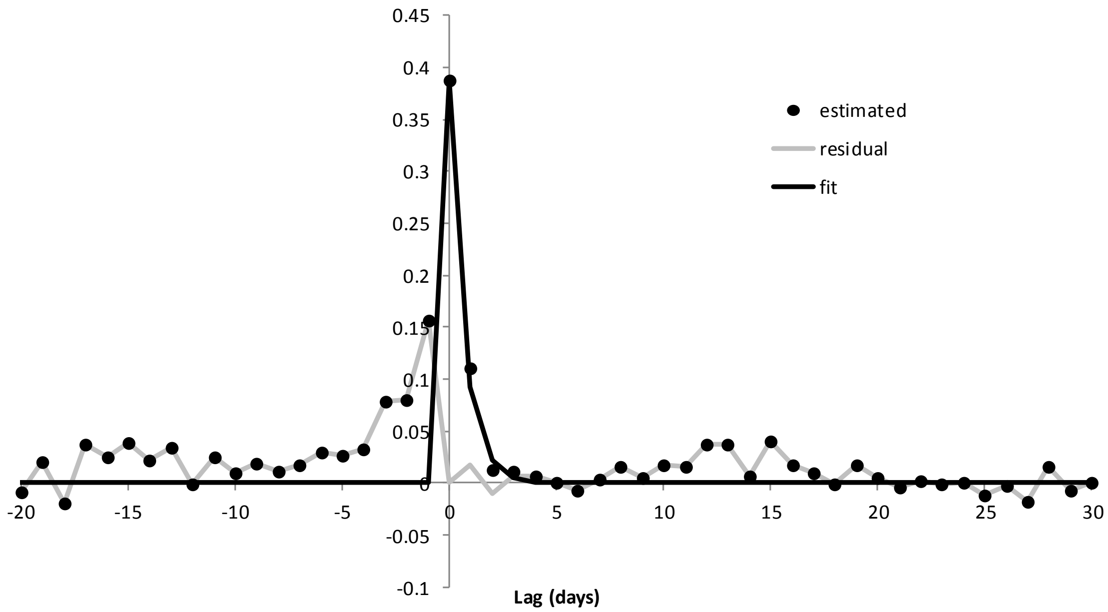


Figure 12. Estimate of routing impulse response function for gauge 410033

There was a significant error in the magnitude of the flow at the gauged sites, with a mean over-estimation by a factor of 2 (median multiplicative factor = 1.43), and a standard deviation of 1.87 (error in mean = 0.62). This was not surprising, as the SEACI modelled flows are regionally calibrated, and not specifically calibrated to the gauges assessed. There was an implication that flows tend to be over-estimated across these gauged sites (with respect to the observed flows, which will also have associated uncertainty). However, generalising these results to the entire region studied is problematic. The indication is that the expected uncertainty (1σ) in the magnitude of flows will be a factor of 2 (actual flow is expected to be between half and double the SEACI modelled values).

3.1.2 Climate and regulated flow data

In this project, we used outputs from the ACT REALM model supplied by ACTEW Water to provide flow data for the regulated rivers. REsource ALlocation Model (REALM) is a simulation-based generalised framework used to represent water supply (e.g. pipes, reservoirs) and demand (e.g. urban centres, irrigation areas) systems. REALM is used to design and test the effects of various supply and demand management options, such as building new supply infrastructure and/or water efficiency measures, and operating rules.

The ACT REALM model includes the four reservoirs that supply water to the ACT: enlarged Cotter (78 GL), Bendora (11.5 GL), Corin (71 GL) on the Cotter River, and Googong (121 GL) on the Queanbeyan River. It also includes a pipeline to extract and transport water from the Murrumbidgee River to Googong Dam, plus the ability to extract and treat Murrumbidgee water at Stromlo Treatment Plant before it is supplied to Canberra. At each simulation time step REALM calculates Canberra's and Queanbeyan's urban potable water demand, inflows (e.g. catchment runoff), and outflows (e.g. spills, environmental flows).

3.1.3 Climate scenario selection

The selection of climate scenarios was directed at those most likely to capture the range of possible future changes for 2030. Our initial focus was on those scenarios likely to generate the most extreme changes and potentially produce adverse water quality and ecological effects, and as a consequence scenarios for 2070 were also considered. To select the scenarios we used a suite of ecologically relevant hydrologic parameters from the Index of Hydrological Alteration (Richter et al. 1996) to determine the magnitude of the flow regime changes.

The Index of Hydrological Alteration (IHA) (Richter et al. 1996) is commonly used across the northern hemisphere to assess the eco-hydrological effects of alteration in flow regimes caused by regulation (e.g. dams, diversions) and climate conditions (Suen 2010). It comprises 32 hydrologic parameters which characterise the intra- and inter-annual variation in flows, according to five key biologically-relevant components of flow regimes (Richter et al. 1996; Table 5).

The full suite of IHA parameters was calculated for all sites for all 30 climate scenarios. Non-parametric inter-annual metrics, including median, 25th percentile, and 75th percentile, were calculated for each of the IHA parameters. Using the "natural" or "pre-dam" data set as a baseline scenario, climate scenarios were assessed in terms of the degree of alteration by calculating the absolute percentage change of these inter-annual metrics.

Table 5. Indicators of Hydrological Alteration (Richter et al. 1996, 1997)

IHA Statistics Group	Regime characteristics	Parameters	Ecological relevance (Black et al. 2005)
Group 1. Magnitude of monthly water conditions	Magnitude Timing	Mean value for each calendar month	Habitat availability
Group 2. Magnitude and duration of annual extreme water conditions	Magnitude Duration	Annual Minima: 1,3,7, 30, 90 day means. Annual Maxima: 1,3,7,30, 90 day means.	Structuring river channel morphology and physical habitat conditions
Group 3. Timing of annual extreme water conditions	Timing	Date of each annual 1-day maximum. Date of each annual 1-day minimum.	Compatibility with life cycles
Group 4. Frequency and duration of high and low pulses	Magnitude Frequency Duration	Number of high pulses each year. Number of low pulses each year. Mean duration of high pulses. Mean duration of low pulses.	Frequency and duration of anaerobic stress for plants
Group 5. Rate and frequency of water condition changes	Frequency Rate of change	Means of all positive differences between consecutive daily means. Means of all negative differences between consecutive daily values. Number of rises. Number of falls.	Entrapment on islands and floodplains

The degree of alteration was determined using the following classes (Richter et al. 1998): no or minor (<30%), moderate (30%–70%), and major (>70%). For the Group 3 metrics, the alteration was classed slightly differently because the indicator was measured in term of absolute difference in days; then the classes were: no or minor change (<30 days) or major change (>30days).

Climate scenarios were ranked and compared in terms of the number of parameters for which there was minor to significant change (assuming that parameters are equally weighted) (full analysis is provided in Appendix C). As a result, six climate scenarios were identified for further analysis. These selected scenarios encompass those which represent major, moderate and minor alterations to the stream flow, where _1 and _2 refer to 1^{o}C or 2^{o}C:

Major change: CSIRO_1 and CSIRO_2

Moderate change: INMCM_1 and INMCM_2

Minor change: NCAR_PCM_1 and NCAR_PCM_2.

3.2 Prediction of flow regime changes for the defined climate scenarios

The flow data generated for the climate enabled a comparison of the effects of river regulation and projected climate change on ecologically relevant attributes of the flow regimes. We used two approaches for measuring the effects of climate on the flow regimes, to assess the severity and extent of human alteration to flow regimes compared with those that may be caused by plausible climate conditions:

1. the Index of Hydrological Alteration (IHA, Richter et al. 1996),
2. Flow Stress Indicators (FSI) (SKM 2005).

These approaches and the results obtained are described in detail in the following sections. Supporting results and analysis are in Appendices D, E and F.

3.2.1 IHA analysis

(i) Parameters selection

Some IHA parameters may be highly correlated and have the potential to bias the end result. Non parametric Kendall's Tau correlation (Kendall 1938) was used to exclude correlated parameters (>0.8) while retaining those that showed the highest degree of alteration.

To select the parameters that represent the highest degree of alteration, we gave each parameter a score using the following rule: 0 (if minor change, <30%), 1 (if moderate change, 30%–70%), and 2 (if major change, >70%). For Group 3 the rules were: 0 (if minor change, <30 days), and 1 (if major change, >30 days).The maximum scores for each group across all six selected climate scenarios are shown in Table 6.

We selected parameters that scored 50% or more of the maximum available points across all regions. The scoring selection method was combined with the Kendall correlation analysis using step-wise selection (full analysis is in Appendix D) in order to make the final selection of an indicator which exhibited significant alteration along with minimal correlation:

Group 1: Monthly mean flows in February and March,

Group 2: Annual 30-day minima,

Group 4: Frequency of high and low pulses, and duration of low pulses.

To facilitate comparison across sites, the selected parameters were combined into one indicator using Euclidean Distance (NWC 2012),

$$IHA.EDj = \frac{1}{100}\sqrt{\left[\frac{1}{n}\sum_{n}(I_i)^2\right]} \quad \forall\, j = 1 \ldots m$$

where

IHA.EDj is the combined score for the IHA parameters at site *j,*

I_i is the absolute percentage change in a given IHA parameter i,
n is the number of parameters,
m is the number of sites.

Table 6. Maximum points available for each group, across all selected climate scenarios, for the purposes of indicator selection

Group	Metrics of interest	Maximum points
1 & 2	Min, Q25, Med, Q75, Max	2 points * 6 scenarios * 5 metrics = **60 points**
3	Med	1 point * 6 scenarios * 1 metric = **6 points**
4 & 5	Q25, Med, Q75	2 points * 6 scenarios * 3 metrics = **36 points**

This provides a measure of similarity in hydrological conditions at a site under a given scenario, and conditions at the same site under the 'natural' conditions. An IHA-ED takes value between 0 and 1, where

IHA.EDj of value close to 0 means that the hydro-ecological conditions at this particular site are similar or closely similar to natural conditions (i.e. minor alteration);

IHA.EDj of value close to 0.5 means that the hydro-ecological conditions at this particular site are moderately similar to natural conditions (i.e. moderate alteration);

IHA.EDj of value close to 1 means that that the hydro-ecological conditions at this particular site are significantly divergent from natural conditions (i.e. major alteration).

Throughout the analysis, the individual site results were aggregated to region level by averaging, using the list of stations in Table 7. These average results were then used to formulate the final results shown throughout Appendices E to F.

Table 7. Flow stations for each region used in the analysis of the flow regime changes

Region	Station numbers
Bredbo	76 and 42
Cooma	81 and 2262
Cotter	700, 2234, 994, 702 and 701
Ginninderra	9064 and 991
Goodradigdee	2136 and 2129
Gudgenby	995 and 996
Lower Molonglo	2191
Mid Molonglo	999, 2188, 997, 998 and 987
Mid Murrumbidgee	50, 141, 704, 705, 777, 990, 993, 2048, 2079 and 9151
Numeralla	62 and 2154
Paddys	2010 and 9015
Queanbeyan	703, 2106, 2235 and 2286
Tuggeranong	9058
Upper Molonglo	208 and 2242
Upper Murrumbidgee	33, 706, 2940 and 9173
Yass	2298, 2300, 26, 90, 2436 and 85

3.2.2 *Flow Stress Indicators analysis*

The IHA method and its derivatives (such as the DHRAM, Black et al. 2005) are emerging as the most commonly used method for assessing hydrologic alteration internationally. However, in Australia, a suite of variance corrected Flow Stress Indicators (FSI) are emerging as the preferred method (Davies et al. 2010; SKM 2005; Slijkerman, Kaye & Dyer 2007). The selection of a different suite of indicators of hydrological change for Australian conditions is in response to the high variability of

Australian hydrology (Finlayson & McMahon 1988) and the adaptation of Australia's aquatic biota to high variability. For example, it is assumed that extracting 20% of the water from a river is likely to have fewer adverse effects on the aquatic biota of a river that is naturally highly variable in flows, than in a river that has consistent flows. Thus, variance corrected indicators determine if the modified flow conditions fall within the range of natural flow conditions in the river. The FSI comprise ten ecologically relevant measures of hydrological change (Table 8; full details of the calculations are in Appendix I). As with the IHA, to facilitate comparison across sites, the selected parameters were combined into one indicator by averaging component indicator scores,

$$HIj = \sqrt{\left[\frac{1}{n}\sum_{n}(FSI_i) \quad \right]} \;\; \forall\, j = 1 \ldots m$$

where
HIj is the combined score for the FSI parameters at site *j*,
FSI is the FSI parameters,
n is the number of parameters,
m is the number of sites.

Flow Stress Indicators are defined mathematically to values between 0 and 1 (Appendix I) where 1 represents no change (equivalent to natural) and 0 a complete change.

To facilitate comparisons between the IHA.ED and the FSI scores, the IHA.ED was converted to IHA.ED* such that values of 1 indicate no change and a value of 0 equals a complete change using the formula *ED** = 1 – *IHA.ED* .

3.2.3 Results and findings

We used the IHA/FSI analysis to compare three sets of data to the "natural" conditions within each region:

for all regions, the flow times series generated from the six selected climate scenarios: CSIRO_1, CSIRO_2, INMCM_1, INMCM_2, NCAR_PCM_1, and NCAR_PCM_2;

for regulated sites, the 'post-dam' or regulated conditions assuming that dams are in place and they stop the flow from going downstream;

for regulated sites, the combined impacts of the six climate scenarios and the four management adaptation alternatives defined in Figure 10 (Section 2.4), called C1, C2, C3 and C4 (or SM1, SM2, SM3 and SM4).

Table 8. Flow Stress Indicators (FSI) (from NRMSouth 2009; SKM 2005)

Component indicator	Description	Specific ecological relevance
Mean annual flow	The change in overall volume of water carried by a waterway each year	No — difficult to link to a specific response
Flow duration	The change in overall flow regime. Considers all points of the flow duration curve to be of equal ecological relevance.	No — difficult to link to a specific response
Variation	Changes in variability (CV)	Biota respond to changes in water level throughout the year
Seasonal amplitude	Reflects changes in depth of flooding and in-stream hydraulics. Reflects changes to the magnitude of flows in 'low flow' and 'high flow' periods.	Changes in water level are drivers of vegetation response — influencing community composition, structure and zonation patterns. Riverine and floodplain productivity responds to floods and low flows and the timing of these defines the nature of the response. Typically aquatic flora and fauna are adapted to the natural patterns of high and low flows (e.g. some fish species rely on spring floods for breeding).
Seasonal period	Reflects changes to the timing of 'low flow' and 'high flow' periods	Typically aquatic flora and fauna are adapted to the natural patterns of high and low flows (e.g. some fish species rely on spring floods for breeding).
High flows	The change in the magnitude of high flows. Reflects changes to maximum depths and velocities. Reflects changes to disturbance events.	High flows play an important role in sediment transport and primary production within a stream.
High flow spells	Changes to flooding (magnitude, duration and frequency). Changes in the number, duration and interval of 'spells' (periods that the flow is above a threshold value),	Duration and frequency of high flows influences plant responses both on the floodplain and in-stream.
Low flows	Changes to the magnitude of low flows.	Low flows are a natural feature of Australian rivers and are considered to be a time of stress for biota. Increasing the magnitude of low flows will reduce the wetted area and thus potential habitat availability for aquatic biota.
Low flow spells	Changes in the number, duration and interval of 'spells' (periods that the flow is below a threshold value)	The duration and frequency of low flows indicates the amount of time aquatic biota are subject to periods of stress through reduced habitat availability and potentially poor water quality.
Zero flow	Reflects changes in the ephemeral nature of streams	The change to the duration of zero flows will reflect changes to the nature of ephemeral streams. Related to the degree of drying of the channel (longitudinal connection) and thus availability of habitat. Increasing the duration of zero flows may ultimately result in a change from aquatic to terrestrial biota.

We structured the analysis around two parts:

1. using results from the combined indicators of IHA.ED and FSI to identify the overall extent of alteration in ecologically relevant hydrological conditions for each of the three data sets described above;
2. using results from the individual IHA and FSI parameters to identify which flow components/characteristics are most vulnerable under which scenario.

We address each of these parts respectively, below.

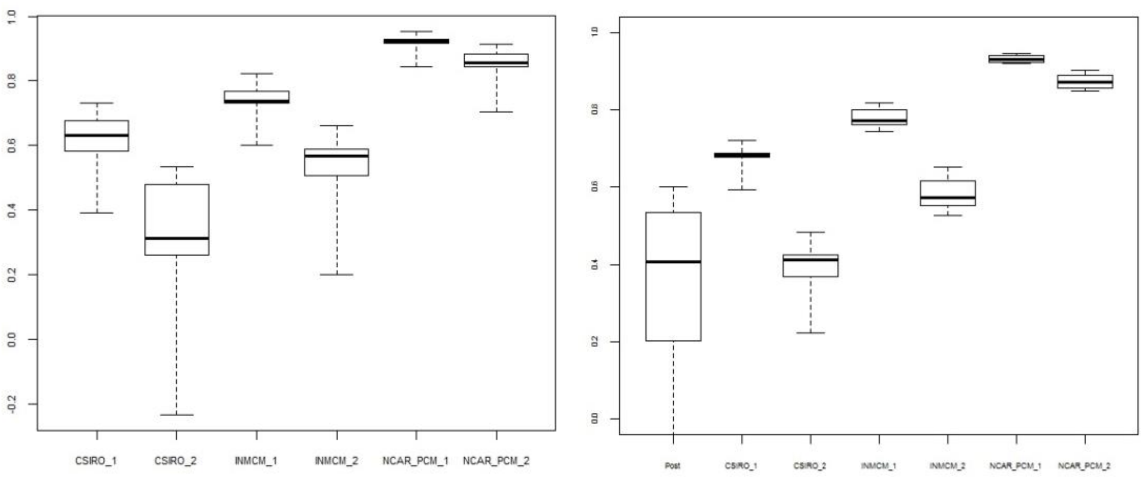
(i) Overall extent of alteration

(a) Regulation and climate impacts

Distinct differences between the three sets of scenarios outlined above are evident in the IHA.ED* and FSI results (Figure 13 and Figure 14). The regulated conditions clearly produce a much higher level of hydrological alteration than most of the climate scenarios (Figure 13). For unregulated sites, the most severe hydrological alteration occurs under the climate scenario CSIRO_2, which shows a level of hydrological alteration similar to that resulting from regulation, but with a much narrower range of values. The NCAR_PCM scenarios have a less significant impact on the hydrological outcome than the other scenario types (Figure 14).

(b) Management, regulation and climate impacts

Individual climate and regulation scenarios were compared to the combined scenarios which included climate and management adaptation alternatives (Figure 15). The combinations of climate change and management alternatives have much greater impact on the hydrology across the region, because the IHA.ED* values are much lower, and many lie below zero, signifying major hydrological alteration. There is little difference between the IHA.ED* results between management alternatives except for alternative C3 (Figure 15(c)) which appears to show the most impact, in particular under the CSIRO_2 climate condition where the median IHA.ED* value is almost –1.



(a) (b)

Figure 13. IHA.ED* results across (a) each climate condition for the unregulated sites, (b) the regulation conditions and each climate condition for regulated sites. Note: a scenario that exhibits no hydrological alteration from the 'natural' state will produce an IHA.ED* value of 1, whilst an IHA.ED* value of zero or less represents major hydrological alteration.

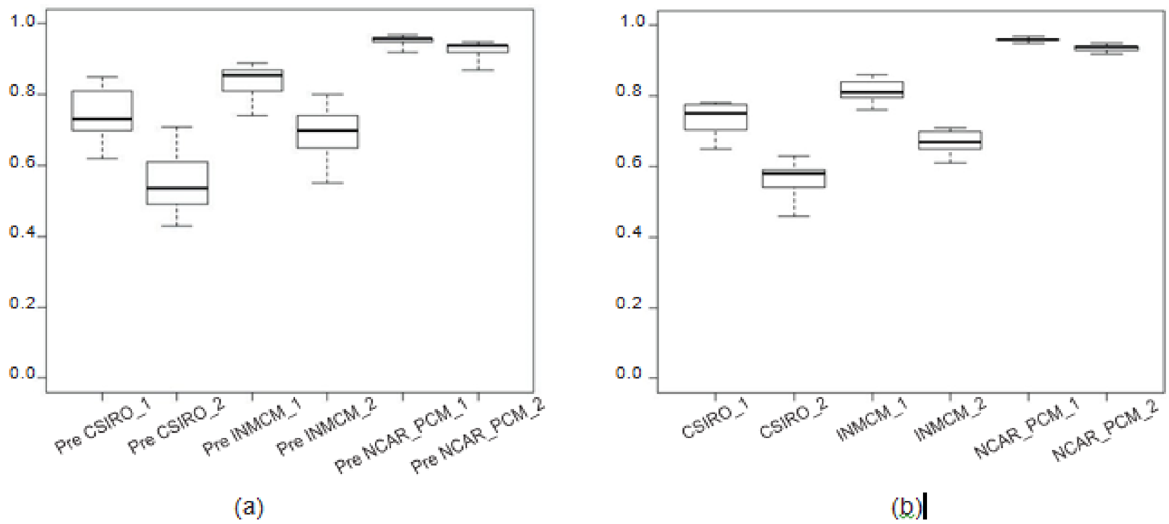


Figure 14. FSI results across (a) each climate condition for the unregulated sites, (b) the regulation conditions and each climate condition for regulated sites

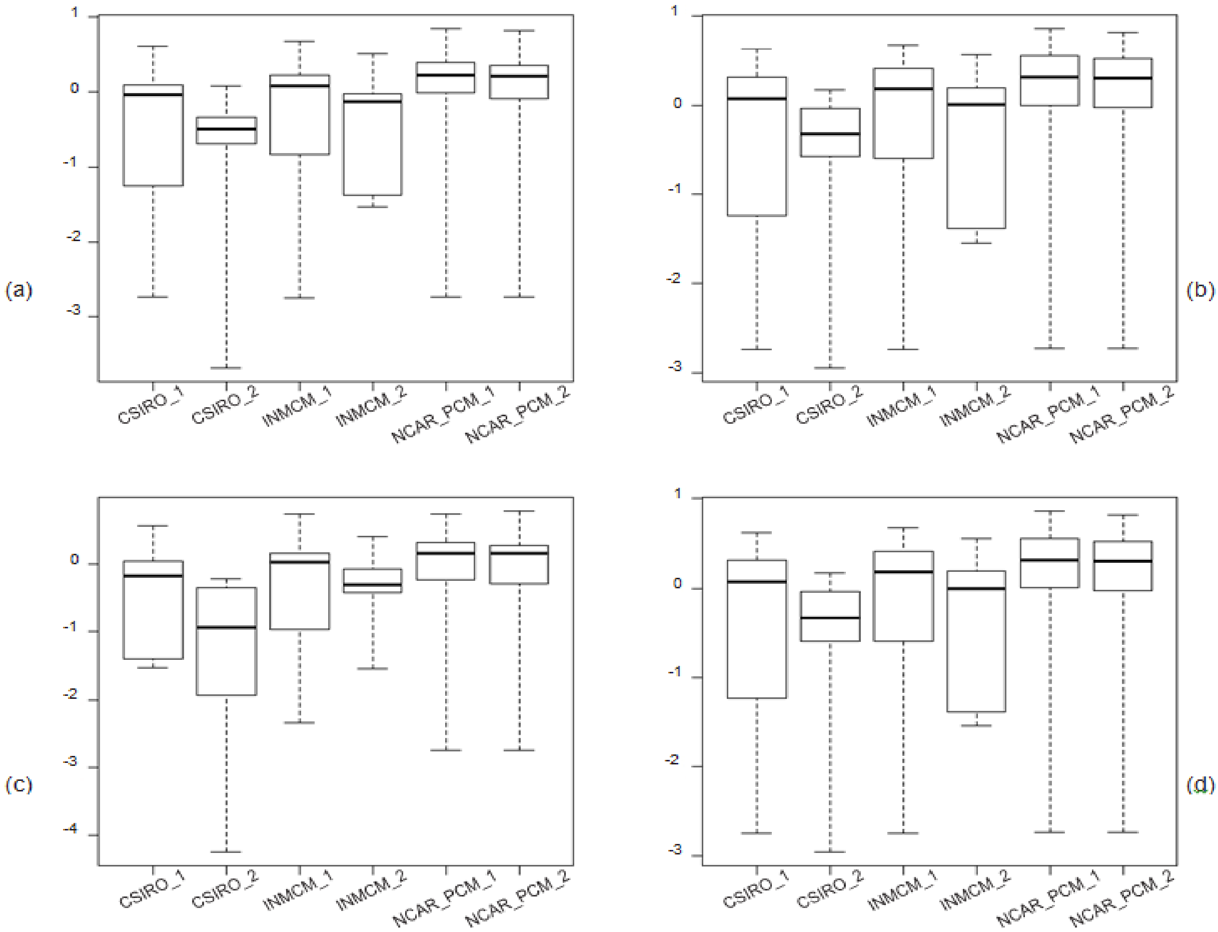


Figure 15. IHA.ED* results for sites across selected regulated sites, under each of the selected climate scenarios for management adaptation alternatives C1 (a), C2 (b), C3 (c) and C4 (d). Adaptation alternatives are described in Section 2.4.

It is clear that population growth and the consequent increase in water use will amplify the impact of climate scenarios (i.e. the minor and moderate alteration scenarios now show significant changes to hydrology with IHA.ED* values close to, or below, 0; see Figures 13, 15). It is noticeable also that the application of the management alternatives along with the climate conditions produces a much wider range of results (Figure 15), with wider boxes, and very extended whiskers towards higher levels of alteration.

Results from the FSI analysis (Figure 16) show management alternative C1 exhibits the greatest variation in the impact on the hydrological outcome, with HI values ranging from around 0.1 to 0.7 for most climate scenarios. There is a significant difference between the impacts of each of the management alternatives. However, Figure 16(b, d) corresponding to management alternatives C2 and C4 appear to have very similar impacts on the hydrological conditions, with narrower ranges and higher median values, in particular for the NCAR_PCM climate scenarios, which is around 0.6. It is noticeable also that the application of the management alternatives along with the climate conditions, in general, produces a much wider range of results with wider boxes, and very extended whiskers towards higher levels of alteration. The exception to this finding would be CSIRO_2 for management alternatives C2 and C4.

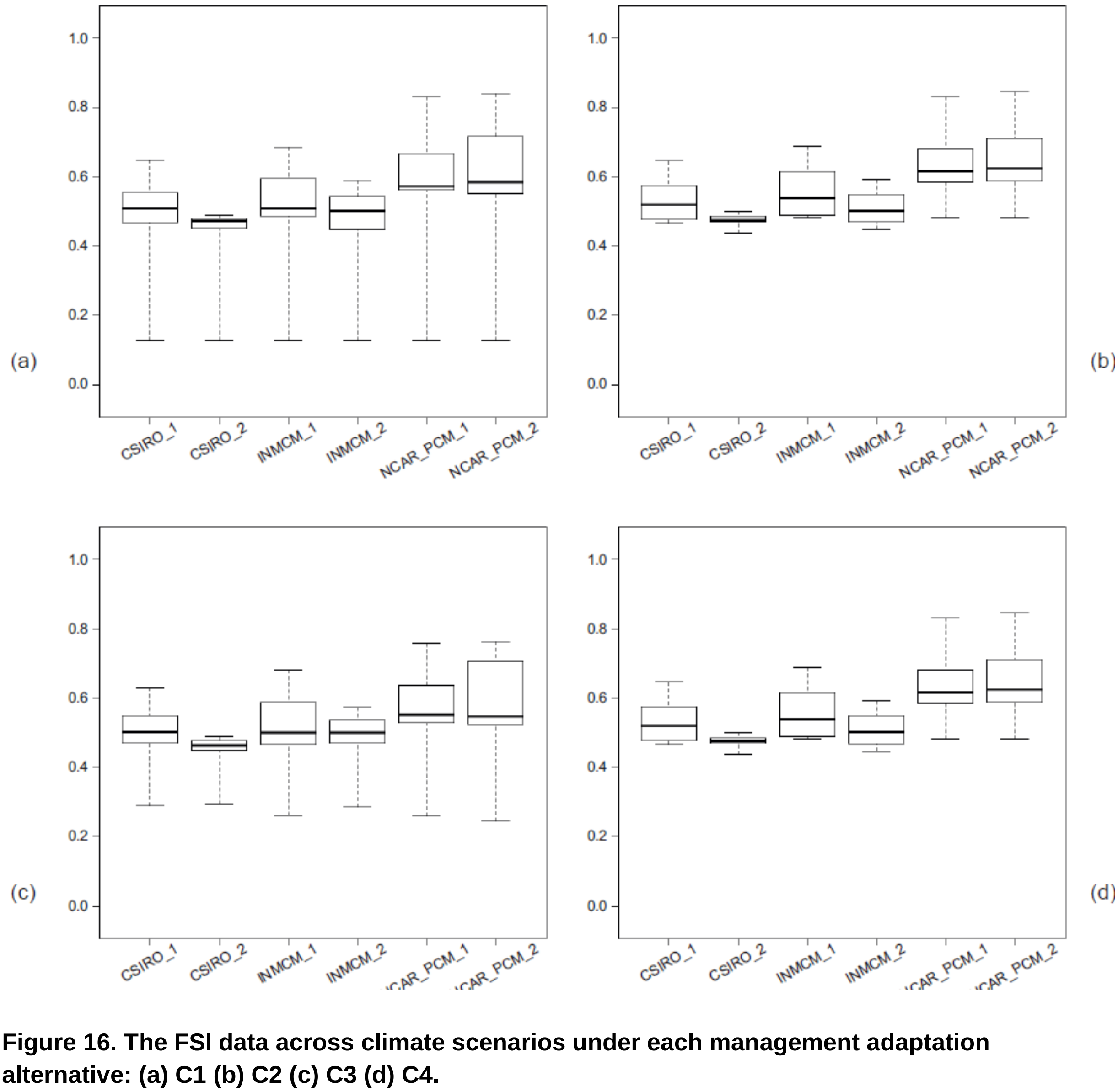


Figure 16. The FSI data across climate scenarios under each management adaptation alternative: (a) C1 (b) C2 (c) C3 (d) C4.

Cross-climate analysis was conducted across all management and regulated conditions to examine plausible impacts under a range of climate scenarios (Figure 17). Results show that regulation has a wide range of effects across these stations, compared to the impact of each climate scenario, while the management adaptation alternatives consistently show much larger levels of alteration than either the regulated conditions or each of the climate scenarios. In the CSIRO_2 climate scenario, the impacts of the management alternatives appear the greatest, in particular at C3. Under the remaining climate scenarios, the median hydrological alteration caused by the management alternatives appears to be relatively similar, around an IHA.ED* value of 0, representing major alteration. The differences between these climate scenarios are in the range of the IHA.ED* values covered; for the NCAR_PCM scenarios, the interquartile range of the IHA.ED* values is very narrow, whereas for the CSIRO_1 scenario, the interquartile range shown by the management alternatives is much wider. Finally, the difference between the impacts of the management alternatives appears to be minimal across all climate scenarios. The management alternatives C1 and C3 show slightly higher levels of alteration, with slightly lower values of IHA.ED*, but this difference becomes increasingly marginal as we move from the CSIRO scenarios through to the NCAR_PCM scenarios.

Results from the FSI analysis indicate that the CSIRO_2 climate scenario shows the greatest impacts when combined with management alternatives, with the lowest median FSI.HI values as well as the smallest range of HI values covered (Figure 18). Under the other climate scenarios, the median hydrological alteration caused by the management alternatives appears to be relatively similar, around a FSI.HI value of 0.5, representing major alteration. The differences between these climate scenarios come in the range of HI values covered; for the NCAR_PCM scenarios, the interquartile range of the HI values is generally wider for these scenarios, relative to the other scenarios. Finally, the difference between the impacts of the management alternatives appears to be minimal across all climate scenarios. Management alternatives C1 and C3 may show slightly higher levels of alteration, with slightly lower values of HI for the NCAR_PCM scenarios, but these differences are marginal.

(ii) Impacts on flow components under climate, regulation and management

Full analyses of IHA and FSI flow parameters were undertaken under each of the climate and regulation scenarios and management alternatives. To cover all possible combinations, a large number of plots were produced. We focus here on presenting plots for selected key findings. The complete set of plots is in Appendix E.

Significant changes in IHA parameters occur under the regulation conditions and it is evident that all hydrological parameters experience major alteration, with maximum percentage changes either around or above 100%. This indicates the considerable flow regime changes caused by regulated conditions. More specifically, the frequency of low pulses is most significantly altered under regulation conditions, with median percentage changes above 100%. The duration of low pulses exhibits a wide range of alteration, which is perhaps significantly skewed by the results from the Queanbeyan region. Finally, the two-monthly mean flows and annual 30-day minima all show similar levels of alteration under regulation. Classed between moderate to major, this alteration appears to be around 70%, but skewed upwards towards a maximum up to 100%. Across all of the climate scenarios, regulation produces much higher levels of hydrological alteration across all the parameters; the relative impacts on different parameters vary between the regulation conditions and the climate scenarios.

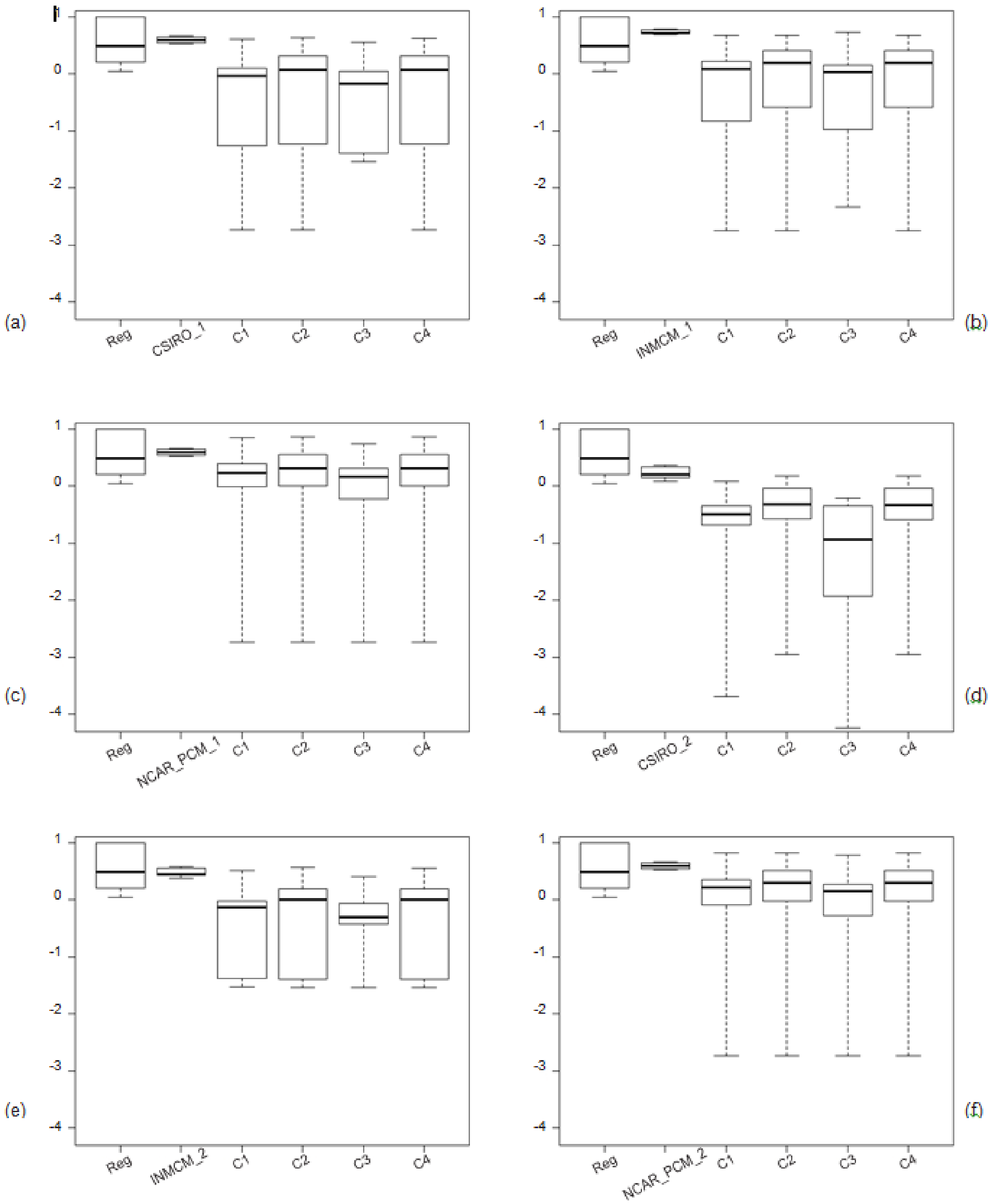


Figure 17. Box-and-whisker plots comparing the IHA.ED* results under regulated conditions (Reg) and the four management alternatives (C1–C4), for each climate scenario: (a) CSIRO_1, (b) INMCM_1, (c) NCAR_PCM_1, (d) CSIRO_2, (e) INMCM_2 and (f) NCAR_PCM_2.

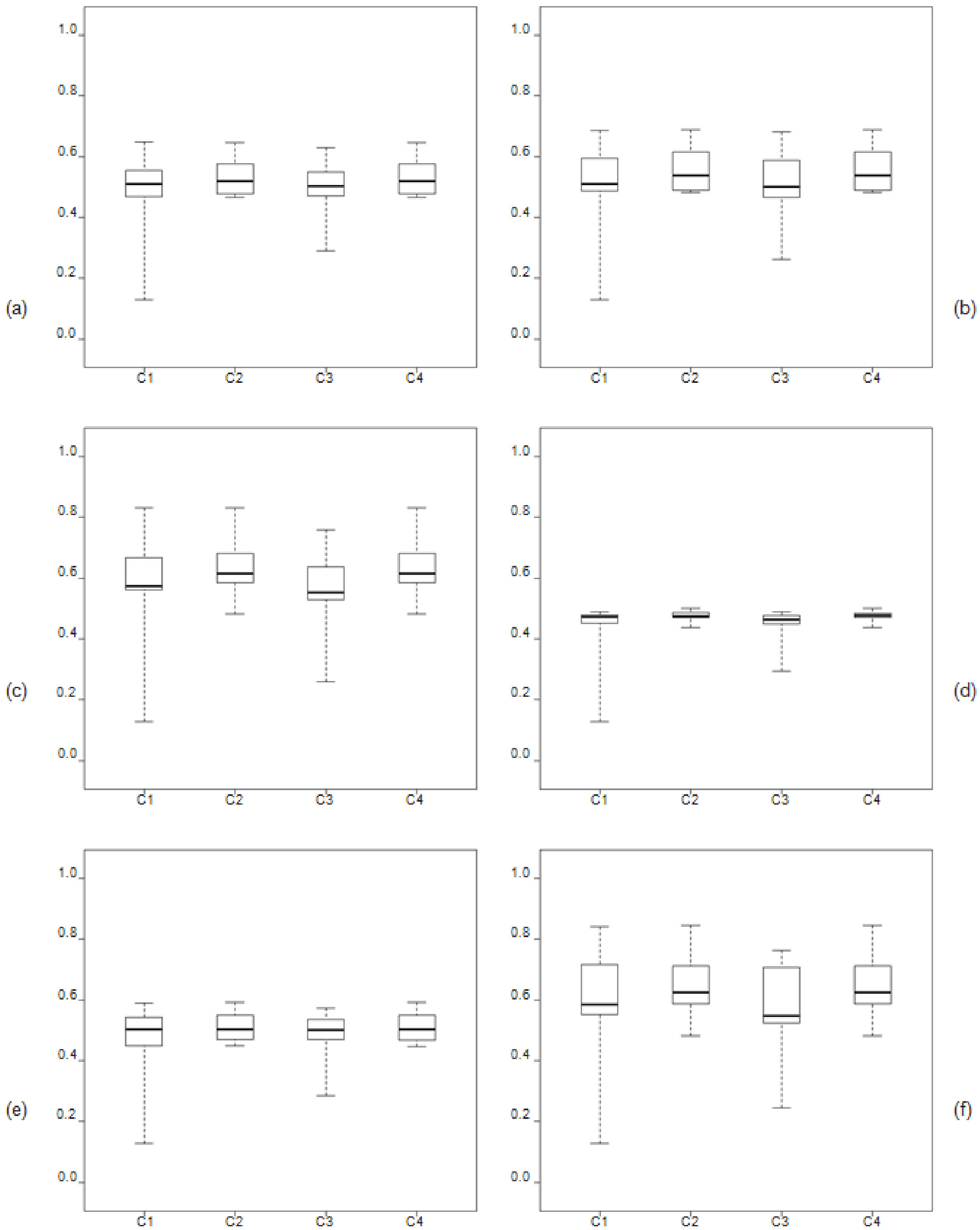


Figure 18. Box-and-whisker plots comparing the FSI HI results under regulated conditions and the four management alternatives (C1–C4), for each climate scenario: (a) CSIRO_1, (b) INMCM_1, (c) NCAR_PCM_1, (d) CSIRO_2, (e) INMCM_2 and (f) NCAR_PCM_2.

In the climate scenarios, the mean monthly flows are often the most affected parameter, but under regulation the frequencies of high and low flows are the more highly altered flow characteristics. Under all climate conditions, the 30-day minima are relatively low compared with the monthly mean flows; however, under regulation, the level of alteration of the 30-day minima is on a par with the monthly mean flows.

Looking at the climate scenarios individually, it is clear that CSIRO_2 produces the most impacted hydrological conditions with major alterations across most of the IHA parameters. The level of alteration then decreases under the CSIRO_1 conditions. For the INMCM climate conditions, the level of alteration shown is moderate across the parameters. Finally, the impacts of the NCAR_PCM scenarios are very minor, and this is the reason this scenario was chosen. Compared to the other scenarios, the characteristics of the hydrology still behave similarly. In particular, the monthly mean flow parameters are the most altered by the climate scenarios, and the 30-day minima and duration of low pulses are the least altered.

Finally the impact of the combined climate scenarios and management alternatives was assessed across all climate scenarios. Across all climate conditions, the combined management adaptation alternatives produce much greater levels of hydrological alteration compared to the regulation conditions or the individual climate scenarios. One of the most noticeable changes from the individual scenarios to the combined options is that the duration of low flows becomes one of the most impacted parameters.

For the CSIRO_2 combined scenarios, the frequency and duration of low flows were much more significantly affected than any other parameters. In particular management alternative C3 produced median percentage change values around 300%. For both INMCM scenarios, the 30-day minima results have an extreme maximum range, which has been distorted by the Queanbeyan region results.

For the NCAR_PCM combined scenarios, the combination of the climate scenario and the management alternatives increased the alteration impact on the hydrological indicators from moderate to major. However, due to the limited impact of the climate scenario, the median results for this combination are still less than 100%. The 30-day minima results are again skewed, partly by the Queanbeyan results (maximum), but the interquartile range is also very wide, and positively skewed.

The difference between the impacts of the two temperature increases for each pair of climate scenarios appears to decrease as the scenarios move from 'major alteration', i.e. CSIRO, to 'minor alteration', i.e. NCAR_PCM. Under the CSIRO conditions, the difference in impact between the two scenarios appears significant. Under the increased temperature of INMCM_2 (2^{o}C option), compared to INMCM_1 (1^{o}C option), the level of hydrological alteration is only increased slightly. Finally, for the NCAR_PCM set of climate scenarios, there appears to be very little difference between the temperature increases.

Similar to the IHA results, FSI flow parameters analysis shows the largest impact under CSIRO_2, with major alterations across Mean annual flow (MAF), High flow (HF), High flow spells (HFS) and Low flow spells (LFS), and moderate alterations across the other parameters, with the exception of Proportion of zero flow (PoZ). The level of alteration then decreases under the INMCM_2 conditions. For the CSIRO_1 and INMCM_1 climate conditions, the level of alteration shown is moderate across most parameters. Lastly, the impacts of the NCAR_PCM scenarios are again very minor, because of the nature of that scenario. All parameters in the NCAR_PCM models are very close to 1 with very little variation between sites.

Across all climate conditions, the combined management adaptation alternatives produce greater change to FSI parameters compared to the 'post-dam' conditions. One of the most noticeable changes from the individual scenarios to the combined options is

the very low outlier values for the proportion of zero flows, which is from the Upper Murrumbidgee site.

For the CSIRO_1 and CSIRO_2 scenarios, the seasonal amplitude, low flow spells, proportion of zero flows and monthly variation parameters have the greatest change. The main difference between these two scenarios is that CSIRO_2 has mean annual flows equal to 0 in almost all cases, the only exception occurring with management alternative C1. For both INMCM scenarios, the mean annual flow, high flow and high flow spells have low values across all management options.

For the NCAR_PCM scenarios, a much greater level of hydrological impact can be seen in the parameters, due to the addition of management alternatives on top of the more conservative climate scenario. Mean annual flow, high flow and high flow spells have low values, similar to the INMCM case, but we also see small low flow spell values for management alternative C1 across both NCAR_PCM scenarios.

As noted in the previous analysis, the difference between the impacts of the two temperature increases for each pair of climate scenarios appears to decrease as the scenarios move from 'major alteration', i.e. CSIRO, to 'minor alteration', i.e. NCAR_PCM. Moreover, there is no evidence of change between NCAR_PCM_1 and NCAR_PCM_2 when looking across the same management option. Under the CSIRO conditions however, we see significant differences even across the same management options.

(iii) Key findings

These are key findings from the IHA and FSI analyses.

- Regulated conditions produce much higher levels of hydrological alteration than the individual climate scenarios.
- Combining climate scenarios and management alternatives produces even higher levels of hydrological alteration, with median percentage change values consistently falling within the 'major alteration' classification (>70%), even for the 'minor alteration' scenarios; NCAR_PCM. Not surprisingly the most conservative climate scenario, CSIRO_2, produces the most impacted results out of the climate scenarios, and combined scenario conditions C3 (assuming high water use) produces more altered hydrological conditions than the other scenarios.
- Under the individual climate scenarios, it is the case that these mean monthly flows are the most impacted parameter.
- Under regulation, the most impacted parameter is shown to be the frequency of low pulses. Results suggest that the number of low pulses that occur throughout a year will alter significantly under a regulated river system.
- Across the combined scenarios the duration of low pulses is often the most altered parameter, when it had been the least so under other conditions. This suggests that the combination of climate scenarios with management alternatives will significantly alter the length of time each low pulse lasts for.
- For the combined scenarios, the frequency of low pulses will also be significantly altered which suggests that throughout the annual cycle there will be more low pulses, each existing for longer lengths of time.
- The full results of the FSI Hydrological Index analysis do not show as clear differences between the climate scenarios and management conditions, as found in the IHA analysis. Although in general the management alternatives have

greater impacts on hydrological outcomes, differences are not as significant as in the IHA analysis.

- The hydrological impact of the management alternatives is significant, but the change between different management alternatives is relatively minor. The CSIRO_2 climate scenario shows the greatest hydrological impact for management alternatives, with HI values around 0.45. Under the remaining climate scenarios, the median hydrological alteration caused by the management alternatives appears to be relatively similar, around an HI value of 0.55, representing major alteration.
- Some of the key messages taken from the FSI parameter analysis echo those given by the Hydrological Index analysis.
- The main drivers of low HI in the management options are the mean annual flow, high flow and high flow spells parameters.
- The effect of temperature on the hydrological parameters is amplified for the less conservative climate scenarios, such as CSIRO, compared to the more conservative climate scenarios, such as NCAR_PCM.
- CSIRO_2 has mean annual flow values equal to zero, indicating extreme hydrological impact in this climate scenario.

3.2.4 Synthesis

We found that regulated conditions in the rivers of the Upper Murrumbidgee catchment produced much higher levels of hydrological alteration than was projected here with any individual climate scenario alone. The effect of climate change on the regulated rivers of the catchment was to amplify the degree of hydrological alteration already experienced. Testing of the management adaptation alternatives proposed in Section 2.4 suggests that the high water demand scenario further amplified the effects of climate change and existing regulation. This means that adaptation policies will need to consider the effects of regulation and consumptive use as a central strategy for protecting freshwater ecosystems into the future.

For unregulated rivers, it is only the most conservative climate scenario (CSIRO), with a 2^{o}C temperature rise, which resulted in significant alteration to the hydrological conditions of the rivers.

3.3 Probabilistic water quality

The second step in linking the climate attributes to water quality and ecological response models is to link the future scenarios to water quality responses. We selected a probabilistic approach to water quality predictions using Bayesian Networks (BNs) to model water quality. We describe below how we developed the water quality models.

3.3.1 Regionalisation

To assist with modelling, we classified the rivers of the Upper Murrumbidgee catchment into regions. This regionalisation was carried out based on expert opinion, Bayesian analysis and multivariate analyses, as explained below.

(i) Regionalisation: Expert opinion

Based on expert opinion, 15 regions were defined based on their dominant land use, landscape position, geology and hydrology (including flow management) (Table 9).

Table 9. Fifteen regions defined in the Upper Murrumbidgee catchment, based on expert opinion

Region	Dominant Land use	Landscape position	Geology	Hydrology
Burrinjuck	Conservation / Dryland agricultural	Mid-slopes	Felsic Intrusive/Sedimentary	Regulated
Bredbo	Relatively natural	Upland	Sedimentary/Intrusive	Unregulated
Cooma	Dryland agricultural	Upland	Mafic Volcanic/Felsic Volcanic	Unregulated
Cotter	Conservation	Upland	Felsic Volcanic/Felsic Intrusive/Sedimentary	Regulated
Ginninderra	Intense (urban) and dryland agricultural	Mid-slopes	Felsic Volcanic	Regulated
Goodradigbee	Conservation	Upland	Felsic Volcanic	Unregulated
Gudgenby	Conservation	Upland	Felsic Intrusive	Unregulated
Mid Murrumbidgee	Dry land agricultural	Mid-slopes	Felsic Volcanic	Regulated
Molonglo	Agricultural /Urban	Mid-slopes	Sedimentary/Felsic Volcanic	Regulated
Numeralla	Relatively natural	Upland	Sedimentary/Felsic Intrusive	Unregulated
Paddys	Relatively natural / Dryland Agricultural	Upland	Felsic Intrusive	Unregulated
Queanbeyan	Conservation /Relatively natural	Upland	Sedimentary/Felsic Intrusive	Regulated
Jerrabomberra	Urban	Mid-slopes	Felsic Volcanic	Unregulated
Upper Murrumbidgee	Conservation / Dryland agricultural	Upland	Sedimentary	Regulated
Yass	Dryland agricultural	Mid-slopes	Sedimentary/Felsic Volcanic	Regulated

(ii) Regionalisation: Bayesian analysis

The water quality characteristics of the sites within each of the 15 regions were assessed using Bayesian Network analysis linking water quality attributes with sites and visually assessing probability distributions for each of the sites within a region (Figure 19). The advantage of this visual approach for assessing the data was that a large proportion of the 40,000 water quality records available could be used in the analysis and missing values at a site did not constrain the analysis. The Bayesian analysis showed that:

- in the Mid Murrumbidgee region, Tuggeranong Creek and Michalago Creek sites had very different P and EC values to other sites within the region;
- sites from the Molonglo region were highly variable, with some of the urban sites displaying quite different pH and EC distributions;
- sites within the Queanbeyan region showed some variability between sites;
- sites from the Cotter region showed moderate to high variability.

On the basis of the Bayesian analysis, possible changes to the regionalisation were proposed.

- Significant differences in salinity levels in the Tuggeranong sites and possibly Michelago from the rest of the Mid Murrumbidgee sites, suggested these sites should be placed into a separate region.
- The Molonglo region defined by expert opinion needed to be separated into three regions, possibly based on location in relation to the Lower Molonglo Water Quality Control Centre (LMWQCC) and the confluence with the Queanbeyan River.

Thus the three proposed regions would be: the Upper Molonglo region including all sites upstream from the Queanbeyan River confluence; the Mid Molonglo river including sites between the Queanbeyan River confluence and upstream of the LMWQCC; and the Lower Molonglo sites downstream of the LMWQCC. It is noted that this would result in only two water quality sites in the Lower Molonglo region.

- The Cotter region could be split into the Upper and Lower Cotter, with sites in the Lower Cotter including Lees Creek, Blue Range and sites below Cotter Dam.
- The regions of Ginninderra and Jerrabomberra, particularly differences observed between sites around Jerrabomberra, could be caused by urban land use. The Mid Molonglo region includes a number of other urban creeks and Jerrabomberra Creek could be included with those.

The outcome of this analysis was a total of 19 regions, with the regions listed in Table 9 augmented by splitting the Molonglo region into three areas (Upper Molonglo, Mid Molonglo, Lower Molonglo), splitting the Cotter region into two areas (Upper and Lower Cotter), including Michelago as a separate region, and incorporating sites from Jerrabomberra Creek into the Mid Molonglo region.

(iii) Regionalisation: Multivariate approach

To verify the regions defined by expert opinion and Bayesian analysis, we examined 9145 records to look for similarity in water quality between defined regions, using cluster analysis in PRIMER Version 6 (Normalised data, group averaging, Euclidean distance measure). The full water quality data set contained several gaps and because of the inability of multivariate statistical techniques to handle data gaps, all records with missing data were removed from the analysis. Total nitrogen, total phosphorus and alkalinity were also removed from the analysis, because data was only available at a low number of sites and sampling occasions throughout the catchment.

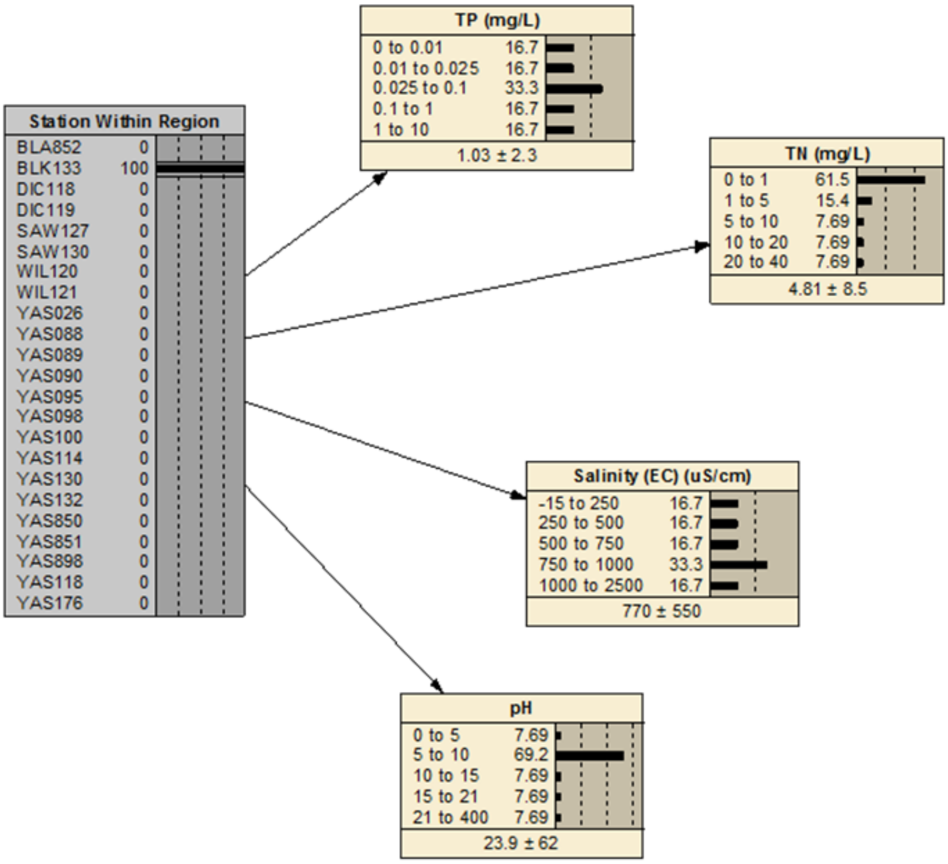


Figure 19. Example Bayesian network showing the probability distributions of the water quality linked to sites of the Yass sub-catchment.

In this example, the water quality probability distribution is shown for list BLK133. A visual assessment of the differences in distribution between sites within the region was undertaken by selecting each station within a region and observing the changes in data distribution. Raw data (prior to cleaning) were used, hence the extremities of the discretisation of data for EC and pH.

To perform the cluster analysis, water quality measurements for all sites within each region were averaged. Significant groupings within the cluster analysis were identified with similarity profile analysis (SIMPROF test). Following the initial cluster analysis run, it was decided to combine sites from the Michelago region into the Mid-Murrumbidgee region, given the close similarity between water quality in the two regions and the small data set for the Michelago region.

Based on water quality data including temperature, electrical conductivity, pH, dissolved oxygen and turbidity, there were three significant groups of regions and Lower Molonglo grouped separately (Figure 20). The 18 regions used for subsequent multivariate analysis are listed in Table 10.

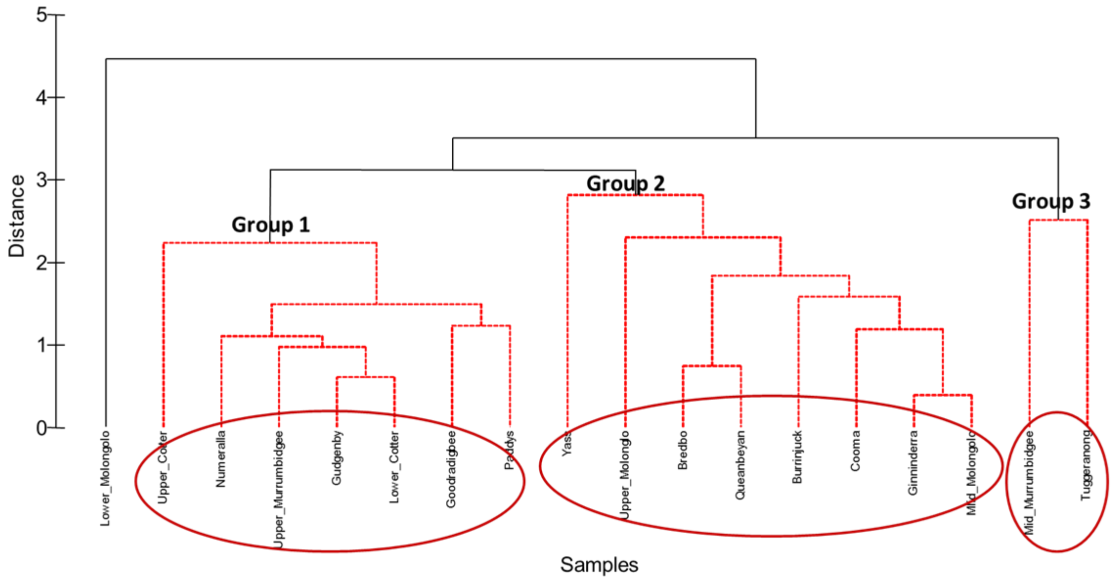


Figure 20. Cluster analysis dendrogram of similarities in water quality for defined regions of the Upper Murrumbidgee catchment. Red lines denote a significant cluster identified with SIMPROF analysis.

Table 10. Regions of the Upper Murrumbidgee catchment tested using multivariate analysis

Group 1	Group 2	Group 3	
Upper Cotter	Yass	Mid Murrumbidgee	Lower Molonglo
Numeralla	Upper Molonglo	Tuggeranong	
Upper Murrumbidgee	Bredbo		
Gudgenby	Queanbeyan		
Lower Cotter	Burrinjuck		
Goodradigbee	Cooma		
Paddys	Ginninderra		
	Mid Molonglo		

Water quality, land use and geology similarities between regions were analysed using the RELATE test in PRIMER Version 6 (Clarke & Warwick 1994; Somerfield & Clarke 1995). RELATE examines relationships between similarity and distances using the Spearmen rank correlation coefficient (Rho). The resemblance matrix based on water quality attributes for each region was compared with resemblance matrices based on land use and geology for each region. Distance matrices were calculated in all cases using the Euclidean Distance. The significance of the correlations was determined by a Monte Carlo permutation procedure (999 permutations). The null hypothesis was that there was no relationship between the sets of samples, and it was accepted if the estimated Rho value was within the permutation distribution of Rho values.

Land use was defined into categories including conservation and natural environments; production from relatively natural environments; production from dryland agriculture and plantations; production from irrigated agriculture and plantations; intensive uses; or water (see Appendix A). Geology was defined as: felsic volcanic; intrusive felsic; mafic

volcanic; sedimentary; or limestone (see Appendix A). Both land use and geology categories were defined as percentage area within each region.

The similarity between defined regions was significantly related to land use (Rho=0.458 sign.=0.001) and not significantly related to geology (Rho=0.169 sign.=0.097) (Figure 21). The lack of a relationship between water quality and geology was unexpected, because it has been well documented that geology has a strong influence on water quality. Furthermore, land use and geology similarities between regions were significantly correlated (Rho = 0.266 sign. = 0.01); therefore, a significant correlation between water quality and geology was expected. The absence of relationship between water quality and geology may be because the categories used in geology were not sufficiently specific to capture geological influences on the water quality, or the geology of the regions is sufficiently chemically homogenous that it does not drive water quality variation.

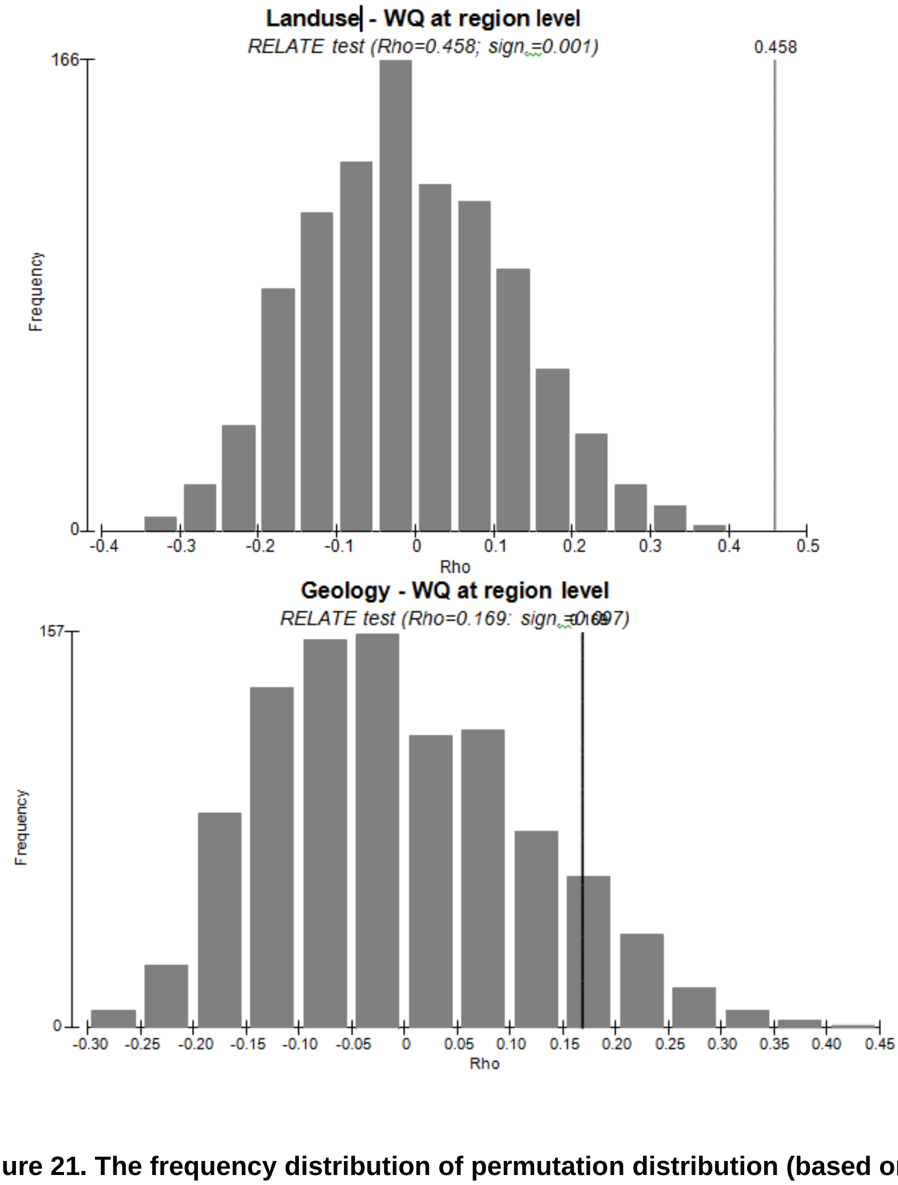


Figure 21. The frequency distribution of permutation distribution (based on 999 perms) of the Spearman rank correlation coefficient (Rho, RELATE test) between: a) land use and water quality (WQ), and b) geology and water quality (WQ). Vertical lines indicate the real Rho values.

3.3.2 *Water quality modelling*

Water quality drivers, processes and outcomes vary spatially, being influenced by a complex set of interacting factors. At a landscape scale, both the drivers of water quality and the processes that contribute to changes in water quality are well understood (Stendera & Johnson 2006). However, the complexity of responses caused by interacting factors means that considerable effort is required to model water quality outcomes (Heathwaite 2010). A variety of water quality models exist (e.g. INCA, Mike11-TRANS and SWAT) and are being applied to investigate the water quality response to climate change. Typically they require more data and assumptions than are available and warranted (Reckhow 1994) and generally have a high uncertainty associated with them (Arhonditsis & Brett. 2004; Reckhow 1999; Wu, Zou & Yu 2006).

Probabilistic models, such as Bayesian Networks (BNs), offer a way of aggregating responses to a set of management actions or environmental conditions; they are being increasingly used in risk assessment where the probability of exceeding a threshold is determined from a set of input conditions. They have been proposed for modelling water quality responses (Reckhow 1999), yet there are few published examples in the literature. More common applications of Bayesian Networks are for ecological response modeling (Borsuk, Stow & Reckhow 2003) where they incorporate elements of water quality modeling. BNs have also been used to assess water quality compliance violations within a water treatment environment (Pike 2004).

Here we used BNs to assess the probability of water quality parameters exceeding thresholds designed to protect aquatic ecosystems given plausible changes in climate in the Upper Murrumbidgee catchment.

(i) Approach

Initial conceptual modeling (Dyer et al. 2011) identified the key water quality attributes for aquatic ecosystems in the Upper Murrumbidgee catchment as being temperature, dissolved oxygen, pH, salts, nutrients and fine sediment (Figure 22). These parameters provided the focus for the water quality modelling.

Key drivers of water quality were defined as flow, climate, geology, land use and landscape characteristics. These provided the focus for the water quality modelling. A significant challenge in using BNs for water quality modelling was noted when the initial conceptual model was converted to a BN. BNs do not appear to be well suited to the integration of spatial information (such as land use and geology) related to a data point; either multiple nodes are required to represent each spatial category (e.g. land use) leading to possible implausible cases, or a large number of categories are required to allow meaningful prediction. This was overcome by defining regions of similar land use, geology and landscape position and using the region as a surrogate for spatial information. This reduces the capacity of the model to be used to predict the consequences of land use changes that may result from climate change, and our focus shifted to isolating the flow-driven water quality changes.

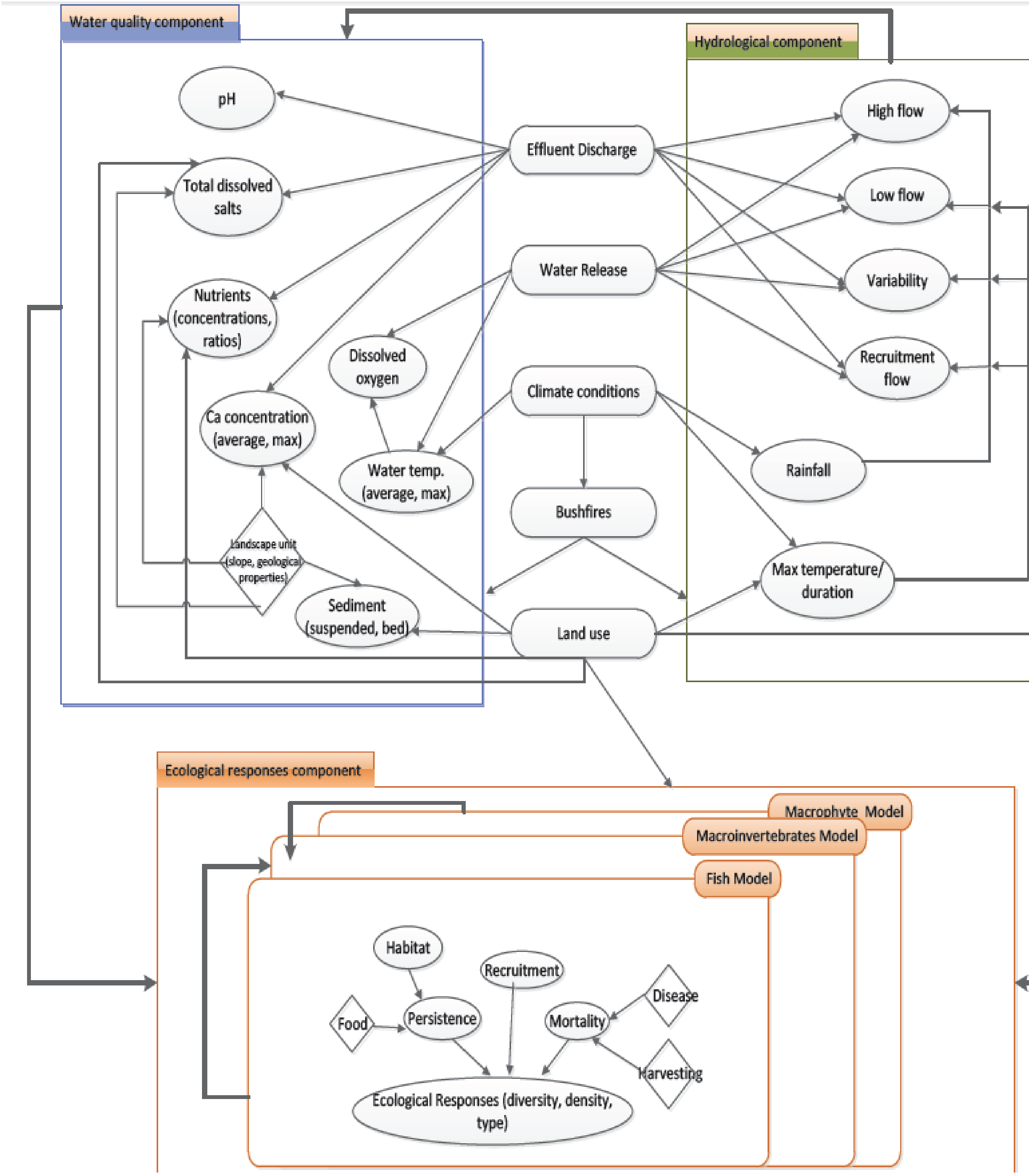


Figure 22. Conceptual integrated modelling framework (presented in Dyer et al. 2011). Key water quality attributes are identified in the upper left hand box of the diagram. *Meaning of shapes*: oval = management policy / uncontrollable driver; rectangle = model component; arrow = data flow link; diamond = input parameter.

Given that we are interested in understanding water quality responses to changes in climate and in particular to changes in flow regimes the approach adopted was to start with a simple model reflecting the key drivers of ecologically relevant water quality

parameters in the catchment. Historical data sets were used to generate frequency distributions of the measured quantities as well as the duration of periods where concentrations are above/below thresholds. These frequency distributions were linked to statistics of flow, climate and landscape attributes (including geology, land use and land management activities).

(a) Bayes net learning

In this application of BNs we use statistical correlations to define the relationships between nodes; those statistical correlations are defined through a process known as Bayes net learning. One of the advantages of the BNs is that conditional probability tables of the network can be filled by “learning” (via automated learning) from data. Netica software (Norsys 2008) can learn from a file of cases, from cases one-by-one, and it can also connect directly with a database, or learn from case files produced in Excel. There are three main types of algorithms that Netica can use to learn conditional probability tables: counting, expectation-maximisation (EM) and gradient descent.

In our study, we used the EM-learning method and connected with text file cases. The text file contained the historical data for the water quality attributes related to the defined regions. Once, the BN learned the distribution of the probabilities of the water quality attributes, the nodes were linked to the thresholds nodes and the probabilities of above/below water quality guidelines were calculated.

For model validation (next section), we also applied the EM-learning method to produce the new conditional probability tables related to event-based data to test the predictions of above/below water quality guidelines based on historical data.

(b) Model structure

Initially, the simplest form of Bayesian model was used to define the relationship between climate scenario, flow and the landscape context and the water quality attributes (Figure 23a). Technical challenges involved in learning the probability distributions associated with land use and geology (caused by sites having varying percentages of states) meant that alternative structures were considered. Given the use of geology, landscape position and land use attributes to define the regions there was redundancy in the model and these attributes were removed and region used as a surrogate (Figure 23b).

(c) Historical Water Quality and Flow Data

Water quality data were sourced from the NSW Government[6], the ACT Government water quality database, ACTEW water quality database, and data collected by research staff at the University of Canberra associated with a variety of research projects. Corresponding historical flow data were also sourced from the same agencies. Bayes net learning was used to incorporate the frequency distributions into the Bayesian model, linking region, flow character, climate scenario and the water quality attributes.

(d) Flow categories

Five flow categories were chosen representing key parts of the flow regime considered likely to have the greatest influence on water quality and subsequent ecological

[6] NSW Government water quality data
<http://waterinfo.nsw.gov.au/water.shtml?ppbm=SURFACE_WATER&rs&3&rskm_url

responses. These were defined on the basis of the historical flow regime and relate to flow percentiles (Table 11).

The change in the frequency of flows in each category was used to test the consequences of each climate scenario.

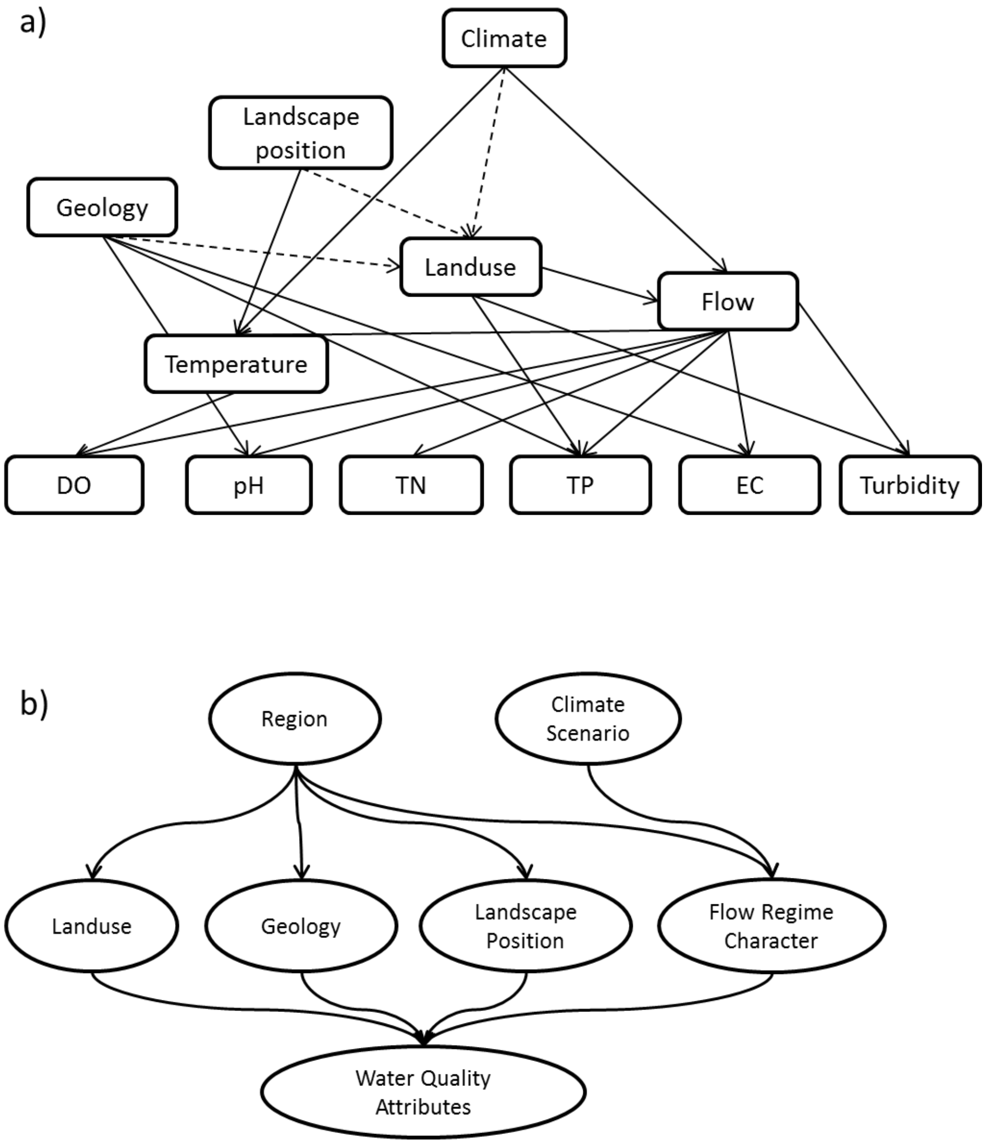


Figure 23. (a) Simplified conceptual model of climate, flow and landscape attribute relationships with water quality in the Upper Murrumbidgee River Catchment. Dotted lines represent indirect relationships. (b) Conceptual model of the initial Bayesian Network model structure.

(e) Thresholds

Thresholds chosen were the trigger values set by agencies to maintain or improve the ecological condition of water bodies. For NSW sites, the key water quality indicators and (default) trigger values selected from the (ANZECC/ARMCANZ 2000) guidelines and specified at <http://www.environment.nsw.gov.au/ieo/Murrumbidgee/maptext-03.htm#wq01> for aquatic ecosystem protection in upland and lowland rivers in south eastern Australia

were used (Table 12). The exception to this was dissolved oxygen, where the (ANZECC 1992) guideline value corresponding to mg/L was used rather than % saturation, because most of the data we had was as mg/L (Table 12). These documents specify a range of trigger values for EC and turbidity for upland and lowland rivers and in each case we used the upper value from the range.

For ACT sites the Environment Protection Regulations SL2005-38 (ACT Government 2013), which cover a variety of water uses and environmental values for each river reach in the ACT, were used for mountain streams, lowland streams and urban streams were used. In the ACT guidelines no total nitrogen guideline is specified; therefore the ANZECC guideline was used.

Table 11. Flow categories used in the Bayesian Network

Flow category	Equivalent percentile range (calculated from historical data)
Very low	0–1
Low	1–10
Moderate	11–89
High	90–99
Very high	99–100

Table 12. Threshold values used for the water quality modelling

Water quality attribute	Indicator numerical criteria (trigger values)
NSW sites	
Total phosphorus	Upland rivers: 20 µg/L Lowland rivers: 25 µg/L for rivers flowing to the coast; 50 µg/L for rivers in the Murray-Darling Basin
Total nitrogen	Upland rivers: 250 µg/L Lowland rivers: 350 µg/L for rivers flowing to the coast; 500 µg/L for rivers in the Murray-Darling Basin
Turbidity	Upland rivers: 2–25 NTU (see supporting information) Lowland rivers: 6–50 NTU (see supporting information)
Salinity (electrical conductivity)	Upland rivers: 30–350 µS/cm Lowland rivers: 125–2200 µS/cm
Dissolved oxygen	Upland rivers: 6 mg/L Lowland rivers: 4mg/L
pH	Upland rivers: 6.5–8.0 Lowland rivers: 6.5–8.5
ACT sites	
Total phosphorus	Mountain streams: ≤40 µg/L Lowland streams: ≤100 µg/L Urban streams: ≤100 µg/L
Turbidity	Mountain streams: <10 NTU Lowland streams: <10 NTU Urban streams: <10NTU
Dissolved oxygen	Mountain streams: ≥6 mg/L Lowland streams: ≥4 mg/L Urban streams: ≥6mg/L
pH	Mountain streams: 6.5–9.0 Lowland streams: 6.5–9.0 Urban streams: 6.0–9.0

(ii) Model validation

Model validation is the process of reviewing and evaluating model performance, i.e. checking that predictions made by the model match those that actually occur. This is not possible when predicting the future because, by definition, the future is yet to occur and one must wait to obtain the data required to validate the models. An alternative is to use a specific event to validate model outputs. In this case, we used water quality data from a special environmental flow release from Tantangara Dam in spring 2011 in the Upper Murrumbidgee Region as the validation data set (a release of approx. 1500 ML/day). The validation data set also included measurements before and after a flood (approx. 5000 ML/day) that occurred a month following the environmental flow release. Both the environmental flow release and flood were within the moderate historical flow percentile range.

The validation data set included data from six sites located downstream of Tantangara Dam in the Upper Murrumbidgee Region (Table 13). Electrical conductivity (EC), pH and turbidity loggers were deployed at these sites to capture the behaviour the flow releases.

We used the historical data set to learn the probability that the water quality would be above/below guideline levels (Table 14). The environmental flow data set was then used to produce new conditional probability tables within the BN that were used to test the predictions of being above/below guideline levels.

When the water quality model was populated with the findings from event based sampling downstream of Tantangara Dam, the percentage of readings within NSW upland river water quality guidelines levels was similar to those predicted by the model built using historical data for moderate flows and dissolved oxygen, total phosphorus, pH and electrical conductivity (Table 14). However, the environmental flow/flood event in the Upper Murrumbidgee River had a greater proportion of turbidity levels above the guideline level than was predicted by the model (Table 14). In addition, the actual percentage of time that total nitrogen was below (within) the guideline level was higher than that predicted by the model (Table 14).

Based on this analysis we concluded that the water quality model based on historical data was a reasonable–good representation of water quality responses within the moderate flow range in an upland catchment for dissolved oxygen, total phosphorus, pH and electrical conductivity. It was not unexpected following the flow release and flood that there would be an increase in turbidity levels. However, the model was not able to encompass the total nitrogen response that occurred during the flow events.

Table 13. Upper Murrumbidgee River sites downstream of Tantangara Dam used for model validation

Model site code	River	Latitude	Longitude	Data logger deployed
MUR938	Murrumbidgee	-35.834	148.804	Yes
MUR943	Murrumbidgee	-36.1713	149.0245	Yes
MUR941	Murrumbidgee	-36.169	149.0215	No
MUR940	Murrumbidgee	-35.9835	148.8515	No
MUR939	Murrumbidgee	-35.983	148.843	No
MUR220	Murrumbidgee	-35.7988	148.6745	No

Table 14. Predicted water quality attribute guideline violations (expressed for moderate flow) based on historical data used to build the Bayesian model (columns 2 and 3) and the violations from the 2011 environmental flow/flood data in the Upper Murrumbidgee Region (columns 4 and 5)

	Predicted violations		Validation data from the environmental flow/ flood	
Water quality attribute	**% Outside guideline**	**% Within guideline**	**% Outside guideline**	**% Within guideline**
Dissolved oxygen	0.51	99.5	1.28	98.7
Total nitrogen	69.6	30.4	20.0	80.0
Total phosphorus	64.5	35.5	62.3	37.7
Turbidity	2.73	97.3	9.72	90.3
pH	3.07	96.9	0	100
Electrical conductivity	2.97	97.0	0	100

3.4 Water quality responses

The compiled model for the water quality attributes is shown in Figure 24 and the beliefs are shown for each node in the form of belief bars. These represent the initial frequency distributions for the water quality attributes for the region of Ginninderra (a mid-catchment area, dominated by urban land use, used to illustrate the model), defined by the historical data set. The threshold nodes indicate the probability that the appropriate jurisdictional guidelines were exceeded. In this region, historically, the probability of exceeding thresholds is very low (<5%) for pH and total phosphorus concentrations; low (5–30%) for dissolved oxygen, total nitrogen concentration and electrical conductivity, and moderate (between 30 and 70%) for turbidity.

For the four climate scenarios tested, most changes in water quality violations observed were negligible, particularly for the 1^{o}C scenarios (Table 16) and most changes suggest a slight reduction in the probability of violating thresholds. The most notable changes occur for total nitrogen concentrations, with a predicted reduction in the probability of exceeding the thresholds for all climate scenarios and most regions, with up to a 24% reduction in the probability of exceeding the total nitrogen thresholds using the 2^{o}C CSIRO projections for the Gudgenby region. Electrical conductivity, pH and dissolved oxygen concentration showed very little response to any of the projected climate changes.

Taking into account the differences between regions, some spatial variations in the predicted changes were observed (Table 15). The greatest projected changes in water quality occurred in the Upper Cotter, Ginninderra, Mid Molonglo and particularly, Gundgenby regions (Table 15).

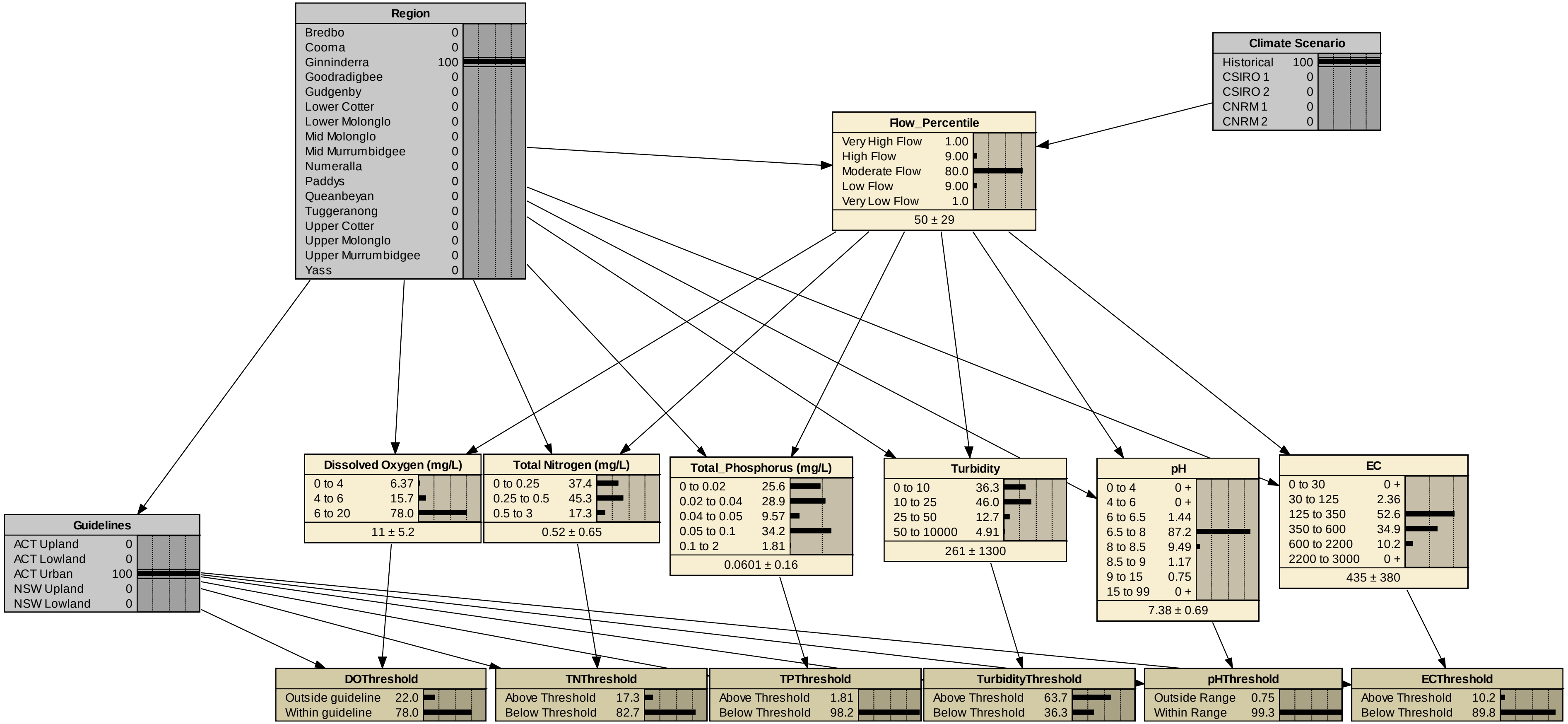


Figure 24. Compiled Bayesian Network water quality model. Model results are shown from the Ginninderra region with historical climate conditions.

Table 15. Change in percentage violations for water quality attributes, in seven regions of the Upper Murrumbidgee catchment under the four selected climate scenarios. Bars in each cell represent the magnitude and direction of the change.

CSIRO_1	DO	TN	TP	Turb	pH	EC	CNRM_1	DO	TN	TP	Turb	pH	EC
Ginninderra	2%	-2%	0%	1%	0%	1%	Ginninderra	0%	-1%	0%	0%	0%	0%
Goodradigbee	0%	-1%	0%	0%	1%	0%	Goodradigbee	0%	-1%	0%	0%	0%	0%
Gudgenby	2%	-10%	0%	-5%	1%	0%	Gudgenby	1%	-5%	0%	-4%	0%	0%
Numeralla	0%	-2%	0%	0%	1%	0%	Numeralla	0%	-1%	0%	0%	1%	0%
Yass	0%	0%	0%	-2%	0%	0%	Yass	0%	0%	0%	-2%	0%	0%
Mid-Molonglo	2%	0%	0%	-2%	0%	0%	Mid-Molonglo	0%	0%	0%	-1%	0%	0%
Upper Cotter	1%	-5%	-3%	-4%	0%	0%	Upper Cotter	0%	-3%	-2%	-2%	0%	0%
CSIRO_2	DO	TN	TP	Turb	pH	EC	CNRM_2	DO	TN	TP	Turb	pH	EC
Ginninderra	5%	-5%	-1%	5%	0%	2%	Ginninderra	1%	-2%	0%	1%	0%	1%
Goodradigbee	-1%	-1%	0%	0%	3%	0%	Goodradigbee	0%	-1%	0%	0%	1%	0%
Gudgenby	5%	-24%	-3%	-9%	1%	1%	Gudgenby	2%	-12%	-1%	-6%	1%	0%
Numeralla	0%	-3%	0%	-1%	2%	0%	Numeralla	0%	-2%	0%	0%	1%	0%
Yass	0%	0%	0%	-4%	0%	0%	Yass	0%	0%	0%	-3%	0%	0%
Mid-Molonglo	6%	-1%	0%	-4%	0%	0%	Mid-Molonglo	2%	0%	0%	-3%	0%	0%
Upper Cotter	3%	-12%	-9%	-6%	0%	0%	Upper Cotter	1%	-7%	-4%	-4%	0%	0%

Our Bayesian Network modelling indicates that the projected water quality changes associated with climate change would be small in the Upper Murrumbidgee catchment. In most cases there is negligible change in the probability that the thresholds designed for the protection of aquatic ecosystems would be violated, and where changes are most notable a decrease in threshold violations is predicted. While many studies predict large changes in water quality attributes with changes in climate (e.g. Wilby et al. 2006; Tu 2009), there are also predictions of much smaller changes. For example, Tong et al. (2012) report changes in mean daily nitrogen concentrations of typically <5% for a range of climate scenarios, which is not inconsistent with our predictions. Rehana & Mujumdar (2012) also predict small changes in the probability of low dissolved oxygen conditions. In addition, note that most published studies represent northern hemisphere examples where concentrations of nutrients are an order of magnitude greater than in the system reported here.

Moreover, our results may be biased by the scale at which the models were developed. The BN used to model changes in water quality does not account for changes that occur at a sub-daily timestep, e.g. changes in storm intensities which occur at small scale are predicted to shift with climate change, resulting in changes in the frequency of peak concentrations of both sediments and nutrients. However, neither the hydrological modelling available nor the historical water quality data available have sufficient resolution to allow such changes to be adequately predicted.
Before effort is directed at understanding the sub-daily water quality and hydrological behaviour, the ecological effects of very-short-duration high concentrations, or high flows, need to be understood to determine if the modelling effort is justified.

4. DRIVER SELECTION AND THRESHOLD DEFINITION

This section describes the methods we used for selecting the predictor variables related to the ecological responses (macro invertebrates and fish) and the threshold values for those predictors (macro invertebrates).This information was subsequently used to structure the integrated Bayesian Network (BN) model and discretise key model nodes.

4.1 Integrated models for predicting management and climate change impacts — Bayesian Network (BN) models

The first step towards forming the BNs was to develop conceptual models (also known as influence diagrams) that identified the primary inputs, drivers and process variables. The influence diagrams were constructed using published literature and expert opinion and were used to map interactions between variables known to have significant influence on water quality and ecological response (for similar constructions see Marcot et al. 2001; Smith et al. 2007). We developed separate influence diagrams to investigate the ecological responses of macroinvertebrates (Figure 25) and native fish (Figure 26) to changes in water flow and water quality, as predicted to result from climate conditions and adaptation policies.

While the influence diagrams informed the structure of the BNs, our aim was to produce BN models that were representative of critical ecosystem characteristics, and able to be populated with existing data sets, and computationally simple.

There are limitations to the size of the networks (approximately 10–12 parent nodes per child) that can be developed, which means that BNs are not suited to being used to investigate relationships within large and complex model structures. Therefore, we used alternative approaches (expert opinion and bottom-up approach using different statistical techniques) to investigate relationships and define the final BN models. Accordingly we (1) identified critical predictors which could be used to structure a BN for selected ecological responses, and (2) quantified thresholds for each of these critical predictors.

In this section we provide a detailed description of the methods we used to select predictor variables for macroinvertebrate and fish communities, and, in the case of macroinvertebrate community, identify thresholds. To select predictor variables two approaches are possible: ‘top-down’ and ‘bottom-up’. The top-down method uses the concept of constraint to identify the constraints that are important at each scale; the bottom-up approach begins with individual or entity-based measurements and adds appropriate constraints to explain the resultant phenomena at broader scales. In the bottom-up approach, the objective is to use information that is available at fine scales to predict phenomena at broader scales for which usually empirical data are lacking.

The merits of each approach can be debated, but the choice depends on the question being investigated, the data collection at broad scales, etc. We chose a “bottom-up” approach. We wanted to provide as comprehensive a picture as possible of what was happening in the ecosystem, and that involved considering a large number of variables. With Bayesian Networks being limited in the number of parent nodes going to one child node, our choice of the bottom-up approach allowed us to reduce the number of predictor variables. This approach is also termed ‘informed-empirical’ where data are used within the context of a conceptual model to select a subset of driver variables.

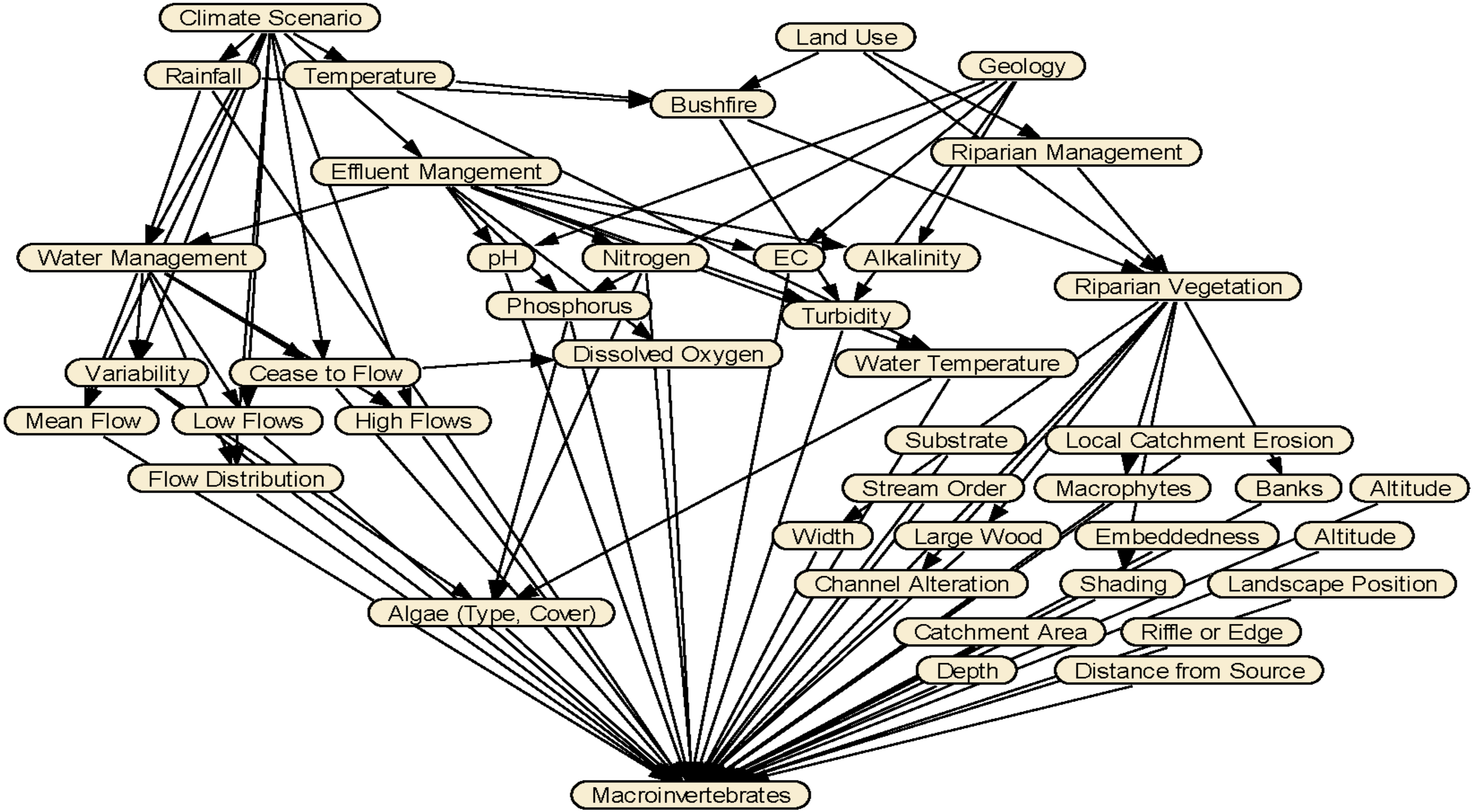


Figure 25. Conceptual diagram used to map variables potentially affecting macroinvertebrate communities

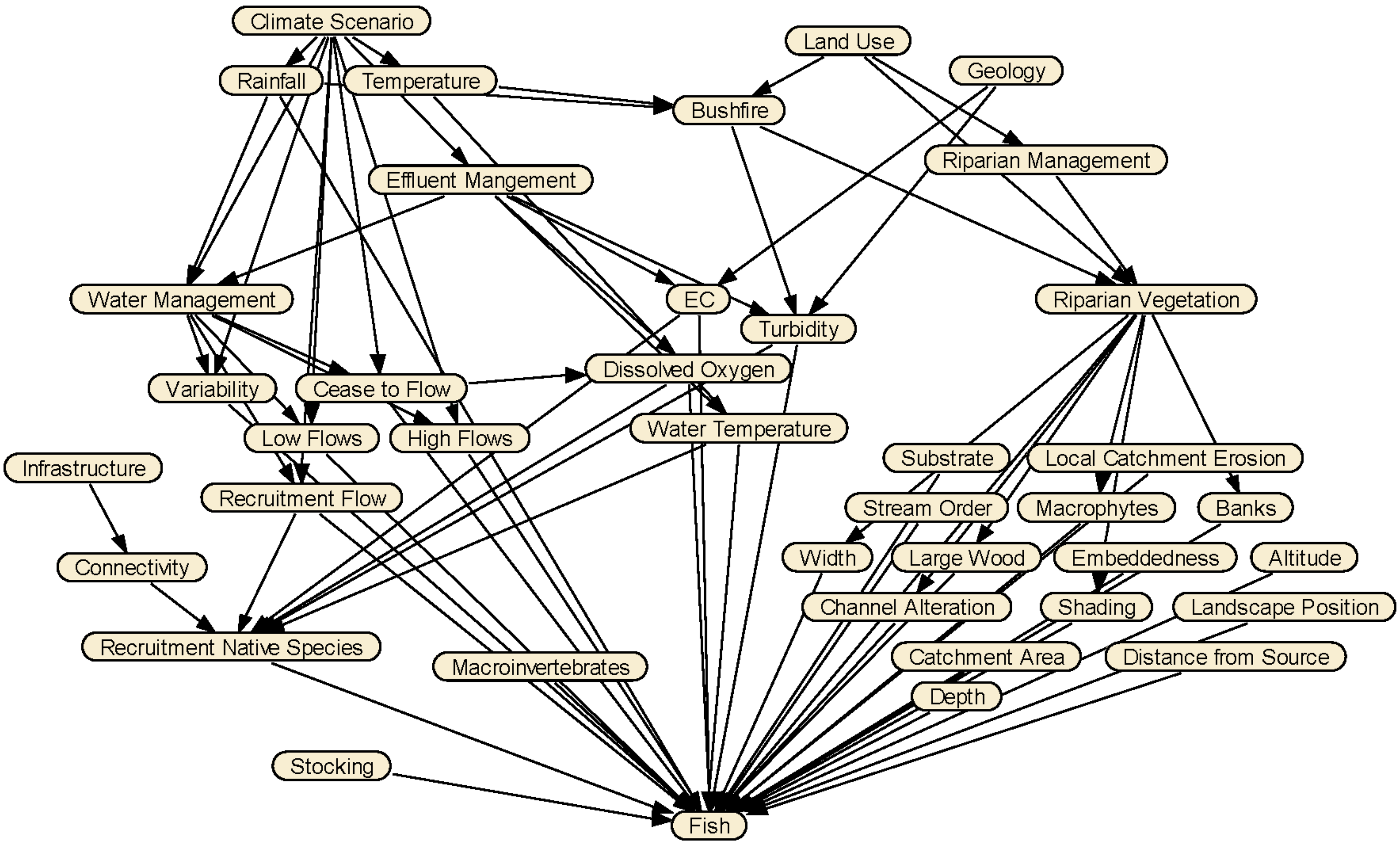


Figure 26. Conceptual diagram used to map variables potentially affecting fish communities

The bottom-up approach for macroinvertebrates was performed using statistical tools (univariate and multivariate), while for fish a combination of expert opinion and univariate statistical methods were applied. The ecology and habitat requirements of native fish in the Upper Murrumbidgee catchment are much better known than those of macroinvertebrates. Therefore, the input of the expert opinion in the fish model was very valuable. Given the lack of knowledge of the ecological requirements of macroinvertebrates, we empirically estimated the threshold values for each of the critical predictor variables. For the native fish community, we applied theoretical values for thresholds as selected from published literature, and expert opinion and guidelines.

4.1.1 Background on thresholds

Environmental changes (e.g. climate, nutrient, hydrological changes) can cause sudden and drastic non-linear shifts in ecosystems, which can have significant consequences on biodiversity (Groffman et al. 2006; Scheffer et al. 2001). Several terms are used to describe these changes: change point, threshold, shift point regime shift, abrupt change, break-point, structural change, tipping point, observational inhomogeneity. Here we use only the single term 'threshold', defined as "a critical value of an environmental driver for which small changes can produce an ecological regime shift" (after Anderson et al. 2008).

Understanding how aquatic communities respond to increasing levels of disturbance, and thresholds in particular, is critical for many aspects of river management, including assessing stream health, predicting future risks, rehabilitating degraded waterbodies, and establishing regulatory criteria (Brenden, Wang & Su 2008). Managers need ecological thresholds that help them to design specific actions to avoid crossing or passing this critical value for the ecosystem.

Several attempts to solve this gap have been launched recently (Bryce, Lomnicky & Kaufmann 2010; Kail, Arle & Jähnig 2012; Utz, Hilderbrand & Boward 2009). Methods for identifying thresholds are relatively new and complex, and little is known about the consistency of results between different approaches. With this perspective, the most common practice among managers is to use theoretical threshold values provided by water quality guidelines such as the ANZECC/ARMCANZ guidelines. This is an acceptable solution to the problem if empirical thresholds do not exist. However, these theoretical values are often not specific to a particular ecological response (e.g. macrophytes, macroinvertebrates, fish), and fail to be useful in a predictive capacity.

To help address this gap between theoretical and empirical thresholds we investigated relationships among an array of predictor variables (land use, geology, habitat and landscape characteristics, hydrology and water quality) and the macroinvertebrate community in the Upper Murrumbidgee catchment. Particularly,

1) we compared the threshold values produced using different statistical methods (univariate and multivariate);
2) we compared the threshold values for different types of community responses; and
3) we used empirical thresholds to develop an integrated Bayesian Network model designed to evaluate the effect of different climate and management scenarios in different regions of the catchment.

4.2 Macroinvertebrates

4.2.1 Methods

(i) Ecological responses

Macroinvertebrate data were acquired from recent and historical monitoring projects conducted by the Institute for Applied Ecology (IAE), the ACT Government, ACTEW Water and the NSW State Government (Table 16). All samples were collected with hand nets (250 µm mesh) from two habitats — riffles and edges — using the standardised rapid biological assessment sampling techniques developed for the Australian National River Health Program (Davies 1994; Nichols et al. 2000; Parsons & Norris 1996; Simpson & Norris 2000). The taxonomic resolution of macroinvertebrate identification in the dataset ranged from subfamily to phylum, but our analysis was carried out at family level because this was the level of most records.

The dataset comprised 1871 samples distributed widely across the Upper Murrumbidgee catchment (Figure 27). Sampling varied between sites: some had been only sampled once or twice while others were sampled at regular intervals and a few had been sampled on several occasions but at irregular intervals (see Appendix J for frequency distribution of the sampling sites).

To provide an integrated view of the macroinvertebrate community, we chose various attributes (community descriptors) that differ in the type of information they provide and their sensitivity to different types of environmental predictors. Three different community measures were studied: two aggregate community indicators (i.e. relative abundance and O/E scores of thermophobic taxa (those which favour cold water) and the macroinvertebrate assemblage (i.e. the array of all families) (see below).

Table 16. Meta-data for macroinvertebrate dataset used in the identification of thresholds

Attribute	Description
Time period of records	1994–2011
Number of records	1871
Number of sites	320
Number of Families	144
Mesohabitat types	
— site samples from edge	153
— site samples from riffle	47
— site samples from edge and riffle	120
Main data sources	NSW Gauging Stations
	ACTEWAGL Stations
	Evan Harrison PhD
	ACT WQ Database
	UC-ECR Tantangara Project

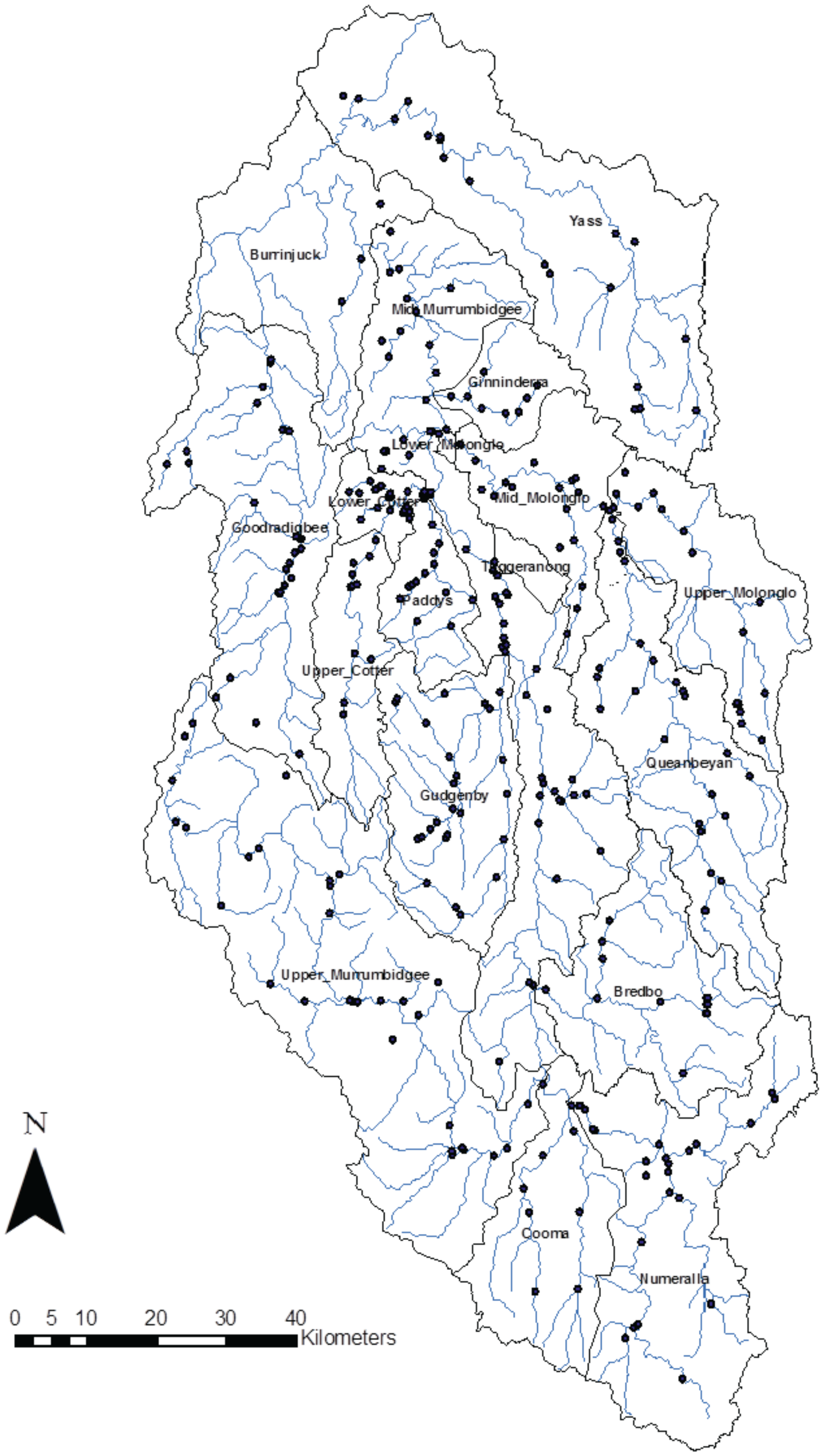


Figure 27. Map of the Upper Murrumbidgee catchment showing the streams (light blue), each region, and the 320 sites where macroinvertebrates were sampled.

Initially we had considered other attributes in addition to these three selected endpoints. The other attributes included: relative abundance of all taxa, EPT richness, %EPT, thermophilic taxa richness (i.e. of taxa which favour warm water), rheophilic and rheophobic taxa richness (i.e. of taxa favouring stream currents or not), relative abundance of rheophilic and rheophobic taxa, and Pielou's evenness. Some of these

responses were discarded because of their weak relationship with the response variables (e.g. Pielou's evenness — data not shown) or because they were highly correlated (>0.7) with other responses: e.g. rheophilic taxa richness and taxa relative abundance were correlated to thermophobic taxa relative abundance at 0.75 and 0.76 level, respectively. This first screening allowed us to come up with the three final endpoints above mentioned: thermophobic taxa relative abundance and O/E scores, and the macroinvertebrate assemblage.

For this project we selected both aggregate indicators and the whole community to test the approaches to modelling and development of thresholds. This was done to allow for the limitations of aggregate indicators, which may reduce the whole community to a single parameter and mask certain ecological responses (King & Baker 2011). We further divided each of the macroinvertebrate ecological responses into either edge or riffle communities, because preliminary multivariate analysis showed strong differences between the two (Appendix J).

In total six different ecological responses were assessed:

(1) O/E scores in riffle,

(2) O/E score in edge,

(3) thermophobic taxa relative abundance in edge,

(4) thermophobic taxa relative abundance in riffle,

(5) macroinvertebrate assemblage in edge, and

(6) macroinvertebrate assemblage in riffle.

(a) O/E scores

In Australia O/E scores are widely used in the bioassessment of river condition (e.g. AUSRIVAS assessments). O/E is an index that compares the observed (O) macroinvertebrate richness (family level) at a site, to that expected (E) under reference or un-impacted conditions. Observed/Expected scores (O/E scores) are derived from the AUSRIVAS predictive model (Coysh et al. 2000; Simpson & Norris 2000). We selected O/E scores as an index of macroinvertebrate community health. The Freshwater Group at the Institute for Applied Ecology (IAE) provided the O/E scores for the sites studied.

(b) Thermophobic taxa relative abundance

Knowledge of taxa traits, particularly those affected directly by climate change can be used to predict the response of the macroinvertebrate community in an increasing temperature scenario. Thermophily or thermal tolerance has been proposed as an indicator of susceptibility of freshwater macroinvertebrates to climate change, because rising air temperatures are expected to increase stream temperatures to the detriment of cold-adapted taxa with narrow thermal tolerances (Chessman 2009, 2012; Tierno de Figueroa et al. 2010).

The thermophily of each family was estimated on a continuous scale as described by Chessman (2009). In short, the thermophily estimate was the mean instantaneous water temperature associated with samples in which that family was detected, divided by the mean water temperature of all samples, ignoring samples for which temperature was not recorded Chessman (2009).

The thermophily estimate produced results in a continuous score. However, we were interested in focusing on the most vulnerable thermophobic species. To establish an objective cut-off for this family selection, we gave two local experts the score list and distribution plots of the given families along the temperature gradient. We asked the experts to select the most thermophobic species. Only those families in which the two experts agreed were selected as “thermophobic taxa”. (See Appendix K for thermophily score list and distribution plots).

(c) Macroinvertebrate assemblage

To compare macroinvertebrate assemblage structure between sites we used the Bray–Curtis similarity measure based on relative abundance data. In total we used the relative abundance of the 144 macroinvertebrate taxa present in the Upper Murrumbidgee catchment — which we sometimes term the ‘whole community’.

(ii) Predictor variables

An initial array of 140 potential predictor variables which describe different types of information were collated. These variables were classified as:

- water quality,
- hydrology,
- habitat,
- land use,
- geology,
- climate, and
- landscape.

Data were from various sources including the IAE, ACTEW, ACT Government, NSW Government, and BoM among others.

To reduce the number of predictor variables and minimise redundancy, we removed correlated variables from the data set by calculating non-parametric Spearman correlations between all predictor variables. If two or more variables were significantly correlated at level >0.70 the variable with the most missing values was discarded.

Following the removal of correlated variables and low frequency variables, a total of 85 predictor variables for the edge community and 92 for the riffle community remained. Appendix L describes these final environmental variables used to model ecological responses, including mean and range.

To avoid statistical constraints because of the high number of predictors in relation to the number of records of the ecological responses, we analysed and modelled the ecological responses separately for each category of dataset (i.e. ecological responses versus water quality, ecological responses versus land use, etc.). The number of variables within each category was different (e.g. water quality: 5 variables; hydrology: 13 variables, land use and geology: 10 variables etc.), which precluded making direct comparisons between the model performances. However, we can compare results between habitats (edge vs. riffle) and between ecological responses (O/E score vs. thermophobic taxa relative abundance vs. whole community).

While this had the advantage of ensuring that our approach was less subject to statistical constraints, it meant that we lost the interactions between variables of different categories. However, relationships between the predictor variables are partly recovered in the Bayesian Network.

4.3 Statistical methods

4.3.1 Selection of predictor variables for threshold analysis and Bayesian Network models

In this study, we only identified threshold values for the predictors highly related to the ecological responses. To select the predictors that best fitted with the ecological response we used two different approaches, depending on the type of the ecological response:

1. BEST test: a multivariate analysis technique used to explore the relationship between the environmental predictors and an entire ecological community (each taxon works as a response variable). Specifically, it compares a fixed matrix (biota) with an array of matrices derived from the environmental data (see (i) below and Figure 28).
2. Boosted regression trees (BRT): univariate analysis used to explore the relationship between environmental predictors and the aggregated community level indicator (i.e. O/E scores and Thermophobic taxa relative abundance).

(i) BEST test

BEST test selects the combination of environmental variables which maximises the rank correlation (Rho, ρ), i.e. it 'best explains' the biotic assemblage structure (PRIMER v6) (Figure 28).

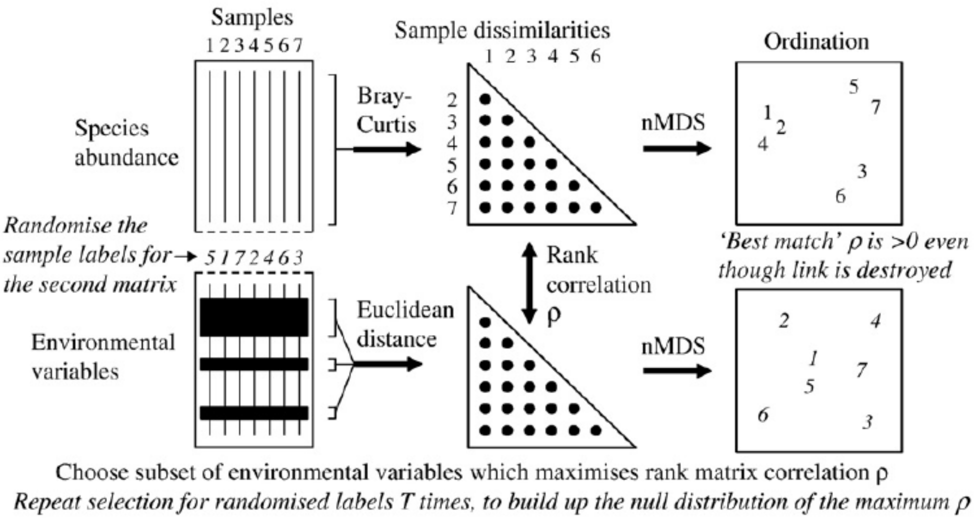


Figure 28. Schematic diagram of the BEST matching procedure and the global BEST test (figure from Clarke et al. 2008, used with permission)

The BEST test was carried out on normalised data to account for the different units of environmental variables. The resemblance matrix for the predictor variables was produced using Euclidean distance; we used Bray–Curtis similarity (previous log-transformed relative abundance) to produce the resemblance biotic matrix.

Two different algorithms are available in PRIMER to carry out the “BEST test”: BioEnV and BVSTEP. We chose the latter because of the high number of predictor variables. BVSTEP adds and removes a variable until the optimum level of correlation (ρ) is reached (it uses a stepwise algorithm employing forward-stepping and backward-elimination as in stepwise regression). A significance test is also calculated based on random permutations of sample names (we applied 999 permutations). The best variable combination is selected after each permutation of samples.

(ii) Boosted regression trees (BRT)

Boosted regression trees were developed from machine learning techniques (Friedman, Hastie & Tibshirani 2000) and can automatically model complex functions and the interactions between variables without making assumptions about the shape of the fitted functions or the interactions between variables (De’ath 2007; Elith et al. 2006). This is a relatively new approach being applied in ecological studies (Elith, Leathwick & Hastie 2008).

BRT combines regression tree and boosting algorithms to produce an ensemble of regression trees. The boosting algorithm improves standard regression tree modelling by adding a stochastic component to the model, which continuously emphasises the poorly explained part of the data space (Elith, Leathwick & Hastie 2008; Friedman, Hastie & Tibshirani 2000). BRT can be considered an advanced form of regression, which also uses a link function to examine a range of response types, including binomial, Poisson and Gaussian (Hastie, Tibshirani & Friedman 2001).

BRT was used to model two of the invertebrate community indicators: O/E scores and Thermophobic taxa relative abundance. Both responses were modelled as a Gaussian response type.

The predictive performance of the BRT models is optimised by means of the learning rate and tree complexity. The learning rate is used to shrink the contribution of each tree as it is added to the model, and to determine the number of nodes in a tree; it should reflect the true interaction order on the indicator being modelled (Friedman 2001). All BRT models had a tree complexity of 5 and were optimised for their learning rate so that a minimum of 1000 trees was fitted for each model (Elith, Leathwick & Hastie 2008). All BRT analyses were carried out in R (version 2.15.0) (R Development Core Team 2011)) using the ‘gbm’ library (Ridgeway 2009) supplemented with functions from (Elith, Leathwick & Hastie 2008).

To assess the contribution of each predictor variable to the BRT models, we assessed the relative contribution of each variable to the model. This measure assesses the number of times a variable is selected for splitting, weighted by the squared improvement to the model as a result of each split, and averaged over all trees (Elith, Leathwick & Hastie 2008).

We assessed model performance using the cross-validated explained deviance — which provides a measure of the goodness-of-fit between the predicted and raw values — and the cross-validated correlation (cvCor), which provides a measure of correlation between the recorded observations and the model fitted values. The cvCor is calculated as a Pearson correlation coefficient and thus takes into account how far the prediction varies from the observed data (Parviainen, Luoto & Heikkinen 2009).

To visualise the fitted functions from the BRT model, partial dependence plots were used (Elith, Leathwick & Hastie 2008). These functions show the effect of a variable on the response, while controlling for the average effect of all other variables in the model (marginal effect).

We used the two approaches (BEST test and BRT) to select those variables which would be used to construct the Bayesian Network model for each ecological response. The criterion to select variables was based on variable performance measures from the Best Test and the BRT. Best Test provides a Rho coefficient (ρ) and BRT provides the contribution of each variable to explain of total variability. However, the cutoffs of these two parameters need to be defined. We fixed the following criteria for our study:

- for BRT models (O/E scores and Thermophobic taxa relative abundance): the predictors are considered as "the best" when their contribution to explaining the total variability is ≥10%. In cases where where no drivers contributed 10% or more, then those that contributed at least 7% were used.
- for the BEST test (Whole community): this test selects a combination of variables which optimise the Rho coefficient (ρ). The variables are selected in descending order of their contribution to maximise the value of Rho. We selected as "the best predictors" the first variable which always had the highest ρ, and the remaining variables which increased ρ by 0.1 (Table 17).

Table 17. Example of variable selection using the BEST test

The first variable, "Flow_Perc90_365" (=Low Flow), was selected because it is the variable with the highest ρ. The second variable, "Flow_Xile" (=Flow Percentile) was also selected because it contributed 0.1 to increase the ρ. The third variable, "Flow_cv90" (=coefficient variation in preceding 3 months) was not selected because it does not increase the ρ by more than 0.1.

WHOLE COMMUNITY - EDGE

No.Vars	Corr. Selections	Rho coeff. (ρ)	Variables	Selection criterion
	1	**0.335**	**flow_perc90_365**	Selected as "the best drivers"
	2	0.437	flow_perc90_365 + **flow_Xile**	
	3	0.463	flow_perc90_365 + **flow_Xile** + **flow_cv90**	Disregarded (contribution to increase ρ <0.1)

(iii) Threshold estimation

Thresholds have received considerable attention recently, causing a proliferation of different methods for identifying them in the last few years (Brenden, Wang & Su 2008). The methods differ in their assumptions regarding the nature of the disturbance-response variable relationship, which can make selecting between the approaches difficult. Moreover, the majority of methods for identifying ecological community thresholds are designed for univariate indicators or multivariate dimension-reduction of community structure (e.g. nMDS scores). Most are insensitive to responses of individual taxa with low occurrence frequencies or highly variable abundances, properties of the vast majority of taxa in ecological community data sets.

Taking into account these considerations, we applied three methods, which use different algorithms to identify thresholds:

- Quantile Piecewise Linear (QPL): a regression tree method that uses quantiles to partition groups (Koenker & Bassett 1978). We used it to estimate the thresholds

for univariate ecological responses or aggregated community indicators, i.e. O/E scores and Thermophobic taxa relative abundance.

- Linkage Tree (LINKTREE): a multivariate adaptation of regression trees (Clarke, Somerfield & Gorley 2008). We used it to estimate the thresholds for multivariate ecological response, i.e. Whole community.
- Threshold Indicator Taxa Analysis (TITAN): similar to regression trees (De'ath & Fabricius 2000) and change-point analysis (King & Richardson 2003; Qian, King & Richardson 2003) but it uses indicator species scores (Dufrêne & Legendre 1997) instead of deviance reduction to locate taxon-specific change points. We used it to estimate the thresholds for Whole community but focused on taxon-specific responses unlike LINKTREE.

(a) Quantile Piecewise Linear (QPL)

Quantile Piecewise Linear (QPL) is an appropriate analytical tool for defining limiting relationships from data that typically appear as wedge-shaped distributions in plots of biotic response to some stressor. (That is, they show small changes in the mean value of the response variable along the gradient of the independent variable, but large changes at the upper end of the distribution (Bryce, Lomnicky & Kaufmann 2010; Cade, Terrell & Schroeder 1999; Dunham, Cade & Terrell 2002; Kail, Arle & Jähnig 2012; Koenker & Bassett 1978).

QPL quantifies the rate of change in the quantiles of the dependent variable, including the lower and upper ends of the distribution (Cade, Terrell & Schroeder1999; Kail, Arle & Jähnig 2012; Koenker & Bassett 1978), whereas other statistical methods, such as least-squares regression, focus on the centre of the distribution (mean or median). For each specific quantile (tau (τ), e.g. 0.1, 0.25, 0.50, 0.75, 0.90), a linear function is fitted such that approximately τ proportion of the observations are below and $1-\tau$ are above the line.

For wedge-shaped relationships, the slopes increase for higher quantiles. Similar slopes among the upper regression lines indicate that the response variable is not limited by other unmeasured factors (Bryce et al. 2008; Cade, Terrell & Schroeder 1999; Kail, Arle & Jähnig 2012). The most appropriate regression line is the largest quantile with the narrowest confidence intervals for a regression line slope that does not contain zero (Bryce, Lomnicky & Kaufmann 2010; Cade, Terrell & Schroeder 1999; Kail, Arle & Jähnig 2012).

To identify threshold values based on QPL, we followed three steps.

1. Identification of the "best quantile".
   a. Question: Is it a wedge-shaped relation? We screened visually for wedge-shaped bivariate relationships between ecological responses (i.e. O/E scores and Thermophobic taxa relative abundance) and predictors (previously selected by BRT). Scatterplots with quantile regression lines (10th, 25th, 50th, 75th, 90th) were inspected.
   b. Question: Do the slopes between quantile regression lines differ? If yes, do the slopes in the upper quantile differ? (i.e. is the predictor variable the main limiting factor?). We tested slope variances for continuous quantiles between 0.05 and 0.95.

c. Question: Which is the best quantile to describe the relationship (i.e. the largest quantile with the lowest uncertainty for the regression slope)?

For steps 1 a, b and c we used the quantreg package in R.2.15.1 (R Development Core Team 2011).

2. Visualisation of the threshold.

   a. Once the best quantile was identified (step 1c), we visually identified possible sharp changes in the response variable (i.e. threshold) at this quantile. Locally weighted quantile regression (loess-QR) was selected for the visually identified threshold (Kail, Arle & Jähnig 2012; King & Baker 2010; Koenker 2011). Loess-QR was carried out for using the function lprq in the quantreg package (R.2.12.0 R Development Core Team 2011).

   b. The function lprq locally fits a linear regression model at several points equally spaced along the independent variable with a specific bandwidth. An intermediate bandwidth resulting in the sharpest change was selected visually. Furthermore, two additional curves were included in the figures to show that lower bandwidths result in an angular curve, which is strongly influenced by single data points, and the curve approaches the linear quantile regression line for higher bandwidths.

3. Estimation of the threshold.

   Visual procedure (step 2) is sufficient to provide a rough idea where the threshold is expected. To support this, we used the QPL approach available in GUIDE v12.6 (Loh 2002) to identify the threshold values at the best quantile.

(b) Linkage Tree (LINKTREE)

Linkage Tree (LINKTREE) is a routine implanted in PRIMER (Clarke, Somerfield & Gorley 2008). It is a multivariate adaptation of "Multivariate Regression Trees" (MRT) developed by De'ath (2002). LINKTREE is a binary divisive cluster analysis based at each step on maximising the ANOSIM R statistic for the two groups that are produced at each split, and represented in the hierarchical diagram.

LINKTREE combines two techniques of PRIMER: the "BEST test" (see above) to select the environmental variables that best explain the biotic pattern and "similarity permutation tests" (SIMPROF test) to provide objective stopping rules for further subdivisions (Clarke, Somerfield & Gorley 2008). The SIMPROF test looks for statistically significant evidence of genuine clusters in samples which are *a priori* unstructured. (For further information, Clarke, Somerfield & Gorley 2008.) LINKTREE produces a divisive, constrained, hierarchical cluster analysis of samples, based on the assemblage data. The constraint is that each binary division of the tree corresponds to a threshold on one of the environmental variables, and maximises the separation of the two groups (ANOSIM R statistic).

LINKTREE needs to fix three parameters: the minimum split size, minimum split R and minimum group size. We considered, as a rule of thumb, minimum split size = 4, minimum split R = 0 (to allow all possible splits) and minimum group size = the number of samples corresponding to 20% of the given dataset (e.g. if the edge–water quality dataset contained 134 records, minimum group size was 27, i.e. 20% of 134).

(c) Threshold Indicator Taxa Analysis (TITAN)

Threshold Indicator Taxa Analysis (TITAN) splits sample units into two groups at the value of a predictor variable (xi, ξ, candidate change points) that maximises the

association of each taxon with one side of the partition (i.e. above and below ξ). Association is measured by taxon abundances weighted by their occurrence in each partition (Dufrêne & Legendre 1997) and standardised as z-scores to facilitate cross-taxon comparison via permutation of samples along the predictor.

TITAN distinguishes declining (negative response) and increasing (positive response) taxa and tracks the cumulative responses of increasing and decreasing taxa in the community. Bootstrapping is used to identify reliable threshold indicator taxa ("purity") and the uncertainty around the location of taxon and community change points ("reliability"). Evidence for a community threshold is obtained through synchronous changes in the abundance of many taxa within a narrow range of predictor values. TITAN was run with the TITAN package in R.2.9.2. (Further details of the TITAN method: Baker & King, 2010.)

Note that for our purposes, we were interested in the threshold derived from the taxa showing a negative response, because this value indicates the point at which the community shifts to more tolerant taxa.

4.3.2 Results and discussion

Overall, the BRT models performed well, explaining between approximately 30 and 80% of the variation in the ecological responses modelled (O/E scores and Thermophobic taxa relative abundance) (Table 18). In general the cross-validation correlations were around 0.5, except for the water quality model for O/E scores in riffle sites, which had a lower value. Relationships between predictors and ecological responses were stronger for edge than in riffle, in contrast to the study conducted by Marchant & Dean (2012) in Victorian streams (Table 19).

The influence of land use and geology variables differed between edge and riffle sites (Table 19). BRT models based on these variables for O/E scores or Thermophobic taxa relative abundance, explained 2-fold more using the edge data compared with the riffle data. Furthermore, it is noteworthy that only land use variables were selected in edge, while only geology variables were selected in riffle, for either of the ecological responses (Table 19; Figures 29–32).

These differences might be explained by the physical position of the edge and riffle habitats in the stream. The riffle habitat may be more dependent on the geology of the stream environment, potentially because of groundwater interactions or substrate structure. By contrast, the edge habitat, located on the margins of the stream bank is likely to be more affected by local land use practices.

Models for edge sites also showed that habitat variables were more influential than in riffle sites. The three response variables in edge (O/E score, Thermophobic taxa relative abundance, Whole community) were related to habitat variables associated with the bank habitat and the riparian zone (Table 19; Figures 29–32). In riffle sites, only the whole community response showed a strong relationship with any of the habitat variables tested, responding to periphyton (Table 19).

Table 18. Performance of the BRT models for the ecological responses for the univariate aggregate indicators: O/E score and Thermophobic taxa relative abundance. Mean explained (%) refers to the variability explained by the model.

O/E SCORE	Habitat		Climate		Water Quality		HYDROLOGY		LANDUSE - GEOLOGY	
	Edge	Riffle	Edge	Riffle	Edge	Riffle	Edge	Riffle	Edge	Riffle
Mean explained (%)	79.37	57.89	71.01	70.27	51.56	28.95	71.67	52.63	60.94	39.47
Mean total deviance	0.063	0.038	0.069	0.037	0.064	0.038	0.060	0.038	0.064	0.038
mean residual deviance	0.013	0.016	0.020	0.011	0.031	0.027	0.017	0.018	0.025	0.023
estimated cv deviance (se)	0.034 (0.002)	0.029 (0.002)	0.035 (0.002)	0.027 (0.002)	0.038 (0.002)	0.033 (0.001)	0.033 (0.001)	0.028 (0.004)	0.031 (0.002)	0.028 (0.002)
training data correlation	0.90	0.79	0.85	0.84	0.72	0.58	0.86	0.76	0.78	0.64
cv correlation (se)	0.672 (0.018)	0.485 (0.028)	0.695 (0.03)	0.519 (0.033)	0.638 (0.011)	0.364 (0.045)	0.651 (0.031)	0.543 (0.052)	0.717 (0.015)	0.525 (0.024)

Thermophobic Abundance	Habitat		Climate		Water Quality		Flow		Land use and Geology	
	Edge	Riffle	Edge	Riffle	Edge	Riffle	Edge	Riffle	Edge	Riffle
Mean explained (%)	56.95	67.78	60.64	63.93	44.50	39.98	62.46	41.77	48.49	28.87
Mean total deviance	70.309	194.823	75.014	187.515	69.058	192.160	49.761	157.548	67.129	191.547
mean residual deviance	30.277	62.781	29.526	67.628	38.325	115.340	18.682	91.734	34.576	136.254
estimated cv deviance (se)	53.331 (6.731)	154.794 (13.64)	50.654 (9.381)	121.995 (14.427)	50.665 (8.107)	149.726 (15.075)	36.225 (6.167)	132.538 (27.16)	47.872 (9.558)	159.486 (15.359)
training data correlation	0.79	0.856	0.80	0.81	0.69	0.66	0.82	0.70	0.70	0.54
cv correlation (se)	0.506 (0.022)	0.469 (0.028)	0.61 (0.031)	0.595 (0.033)	0.545 (0.037)	0.482 (0.039)	0.517 (0.029)	0.436 (0.053)	0.552 (0.047)	0.425 (0.027)

Table 19. The three ecological responses considered in this study (O/E scores, Thermophobic taxa relative abundance, Whole community) and the best predictors associated with each ecological response. Predictors are classified according to the following categories: Habitat, Climate, Water Quality, Hydrology, Land use. For each driver, the deviance explained within the BRT model for the univariate aggregated indicators (i.e. O/E scores and Thermophobic taxa relative abundance) or the Rho coefficient (ρ) derived from the BEST text for the whole community, is shown in brackets. For explanation of predictors and their units see Appendix L.

CATEGORY OF DRIVER	PROCEDURE	ECOLOGICAL RESPONSE	BEST DRIVERS (per mesohabitat)	
			Edge	Riffle
HABITAT	BRT	O/E Score	% Cover of Riparian Zone by Shrubs (10.66%)	
		Thermophobic Abundance	Habscore (12.54%)	
	BEST test	Whole macroinvertebrate	Shading of River (ρ=0.150)	Periphyton (ρ=0.127)
CLIMATE	BRT	O/E Score	Temp max (annual mean) (33.05%)	Rainfall (annual mean) (8.4%) Temp max (annual mean) (7.2%) Temp min (annual mean) (7.2%)
		Thermophobic Abundance	Temp max (annual mean) (22.68%)	Rainfall (annual mean) (20.18%)
	BEST Test	Whole macroinvertebrate	Temp max (annual mean) (ρ=0.246)	Rainfall (annual CV) (ρ=0.15)
WATER QUALITY	BRT	O/E Score	EC (31.79%) DO (9.98%)	Water Temperature (7.7%) EC (7%)
		Thermophobic Abundance	EC (21.91%)	Water Temperature (18.8%) EC (7.9%)*
	BEST Test	Whole macroinvertebrate	EC (ρ=0.308)	Water Temperature (ρ=0.114)
HYDROLOGY	BRT	O/E Score	Flow (annual mean) (13.9%) Num. Days CTF (Year) (11.1%)	Flow (annual mean) (12.5%) High Flow (10th Year) (9.8%)
		Thermophobic Abundance	Flow Percentile (11.07%) Flow (annual CV) (10.65%)	Low Flow (90th Year) (11.35%)
	Best Test	Whole macroinvertebrate	Low Flow (90th Year) Flow Percentile (ρ=0.415)	High Flow (10th Year) (ρ=0.123)
LANDUSE GEOLOGY	BRT	O/E Score	Intense (37%)	Granite (8.10%) Other (8.06 %)
		Thermophobic Abundance	Agriculture (11.57%)	Sandstone (7.91%)
	BEST Test	Whole macroinvertebrate	Intense (ρ=0.208)	Volcsed (ρ=0.181)

In the case of water quality variables, electrical conductivity (EC) was selected as an important predictor of both edge and riffle macroinvertebrates for most of the ecological responses, but was more important for edge (Table 19). In addition, the water temperature was significant for all three ecological responses in riffle (Table 19; Figures 29–32). Dissolved oxygen was also important for O/E scores in edge (Table 19). In riffle communities this may be a reflection of the relationship between water temperature and water levels. When discharge is low, the water temperature is expected to be higher, changing the community living in the riffle. The importance of flow is also shown in the relationship with the hydrological variables, as the ecological responses in riffle were associated with low and high flow. Meanwhile in the edge, other hydrological variables such as the coefficient of variation, cease to flow (CTF) or flow percentile were selected (Table 19; Figures 29–32).

Finally, climate variables (air temperature and rainfall) explained a large part of the variability in both edge and riffle ecological responses (Table 19). In edge sites, mean annual maximum temperature was the only predictor variable selected across the three ecological responses (Table 19; Figures 29–32). In riffle, rainfall appeared to be the most important climatic variable instead of temperature (Table 19).

The relationship between temperature and the ecological responses in edge sites and the rainfall in riffle sites may again be related to local differences between these habitats. Macroinvertebrates in the riffle sites are likely more dependent on the water discharge which in turn is influenced by the rainfall; meanwhile, the edge is more associated with the land–water–air interface and so may be more dependent on the air temperature.

In addition, partial dependence plots produced using the BRT approach allowed an initial view of the relationship between the response variables and the ‘best’ variables, giving an estimate of where we could expect the threshold value (or regime shift) to occur. For instance, O/E scores in edge (Figure 29) responded positively to % Cover of riparian zone by shrubs, and to dissolved oxygen. Based on the graphs, one might reasonably expect threshold values around 20% and 8 mg/L, respectively for these two predictors. On the other hand, O/E scores in edge responded negatively to maximum temperature, conductivity, flow, cease-to-flow and intense land use. Threshold values around 19^{o}C (temperature), 100 µS/cm (EC) and 10% (intense land use) could be predicted. In the case of the cease-to-flow (number of days) there appear to be two thresholds, first around 25 days and a second one before 50 days. Mean daily flow for the year (ML/day) appeared to have a unimodal response, with an increase in the O/E scores around 100 ML/day, and then a decrease in the variables response after this threshold.

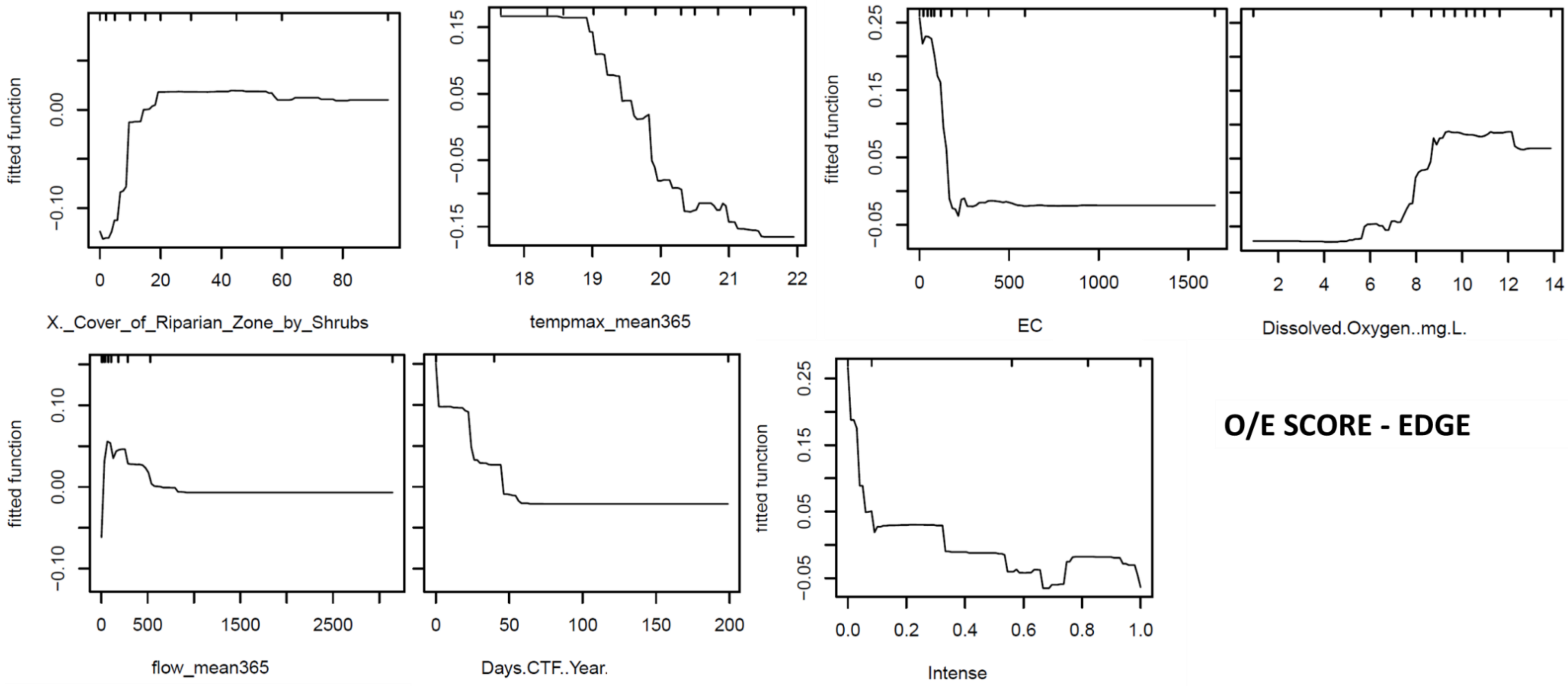


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Figure 29. Partial dependence plots for the seven most influential variables in the model for O/E score in edge. For explanation of variables and their units see Appendix C. Y axes are on the logit scale and are centred to have zero mean over the data distribution. Rug plots at inside top of plots show distribution of sites across that variable, in deciles.

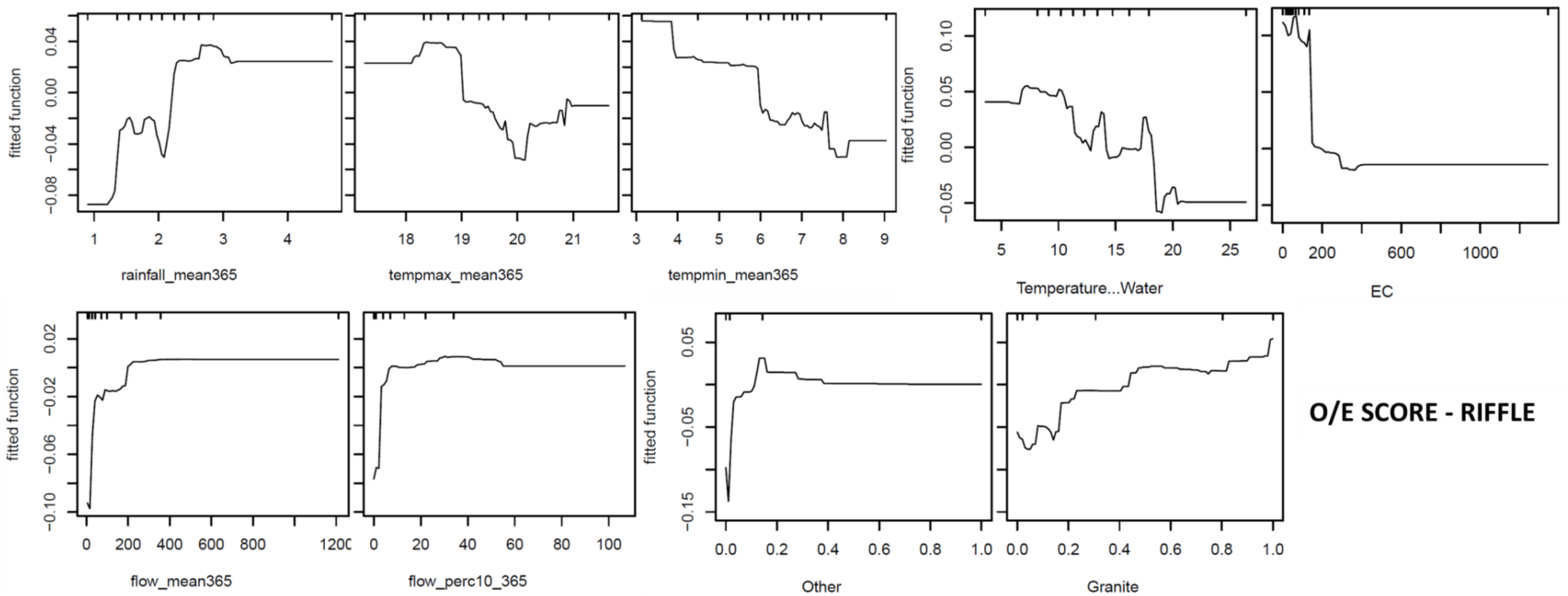

Figure 30. Partial dependence plots for the nine most influential variables in the model for O/E score in riffle. Note that the influence of these variables was very weak (see text in Results and Methods). For explanation of variables and their units see Appendix L. Y axes are on the logit scale and are centred to have zero mean over the data distribution. Rug plots at inside top of plots show distribution of sites across that variable, in deciles.

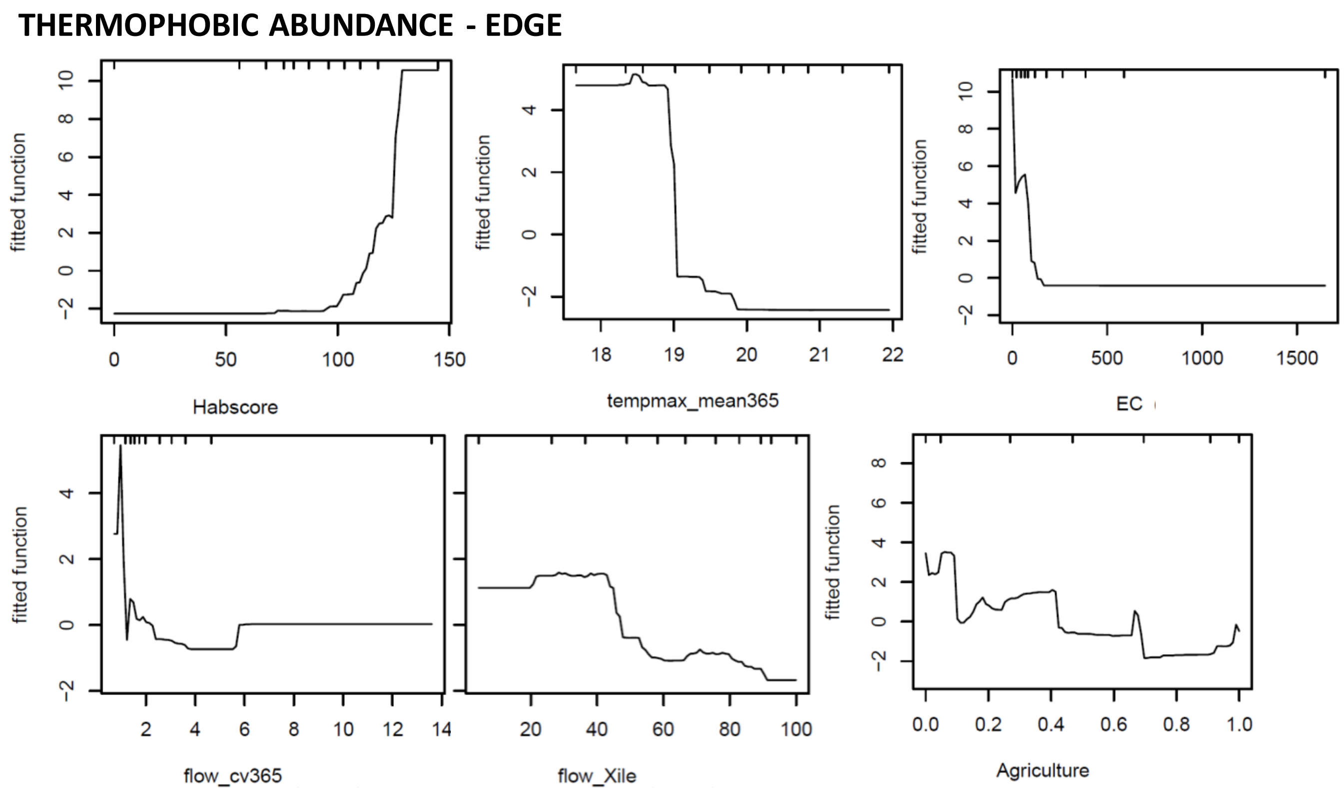

Figure 31. Partial dependence plots for the six most influential variables in the model for thermophobic taxa relative abundance in edge. For explanation of variables and their units see Appendix C. Y axes are on the logit scale and are centred to have zero mean over the data distribution. Rug plots at inside top of plots show distribution of sites across that variable, in deciles.

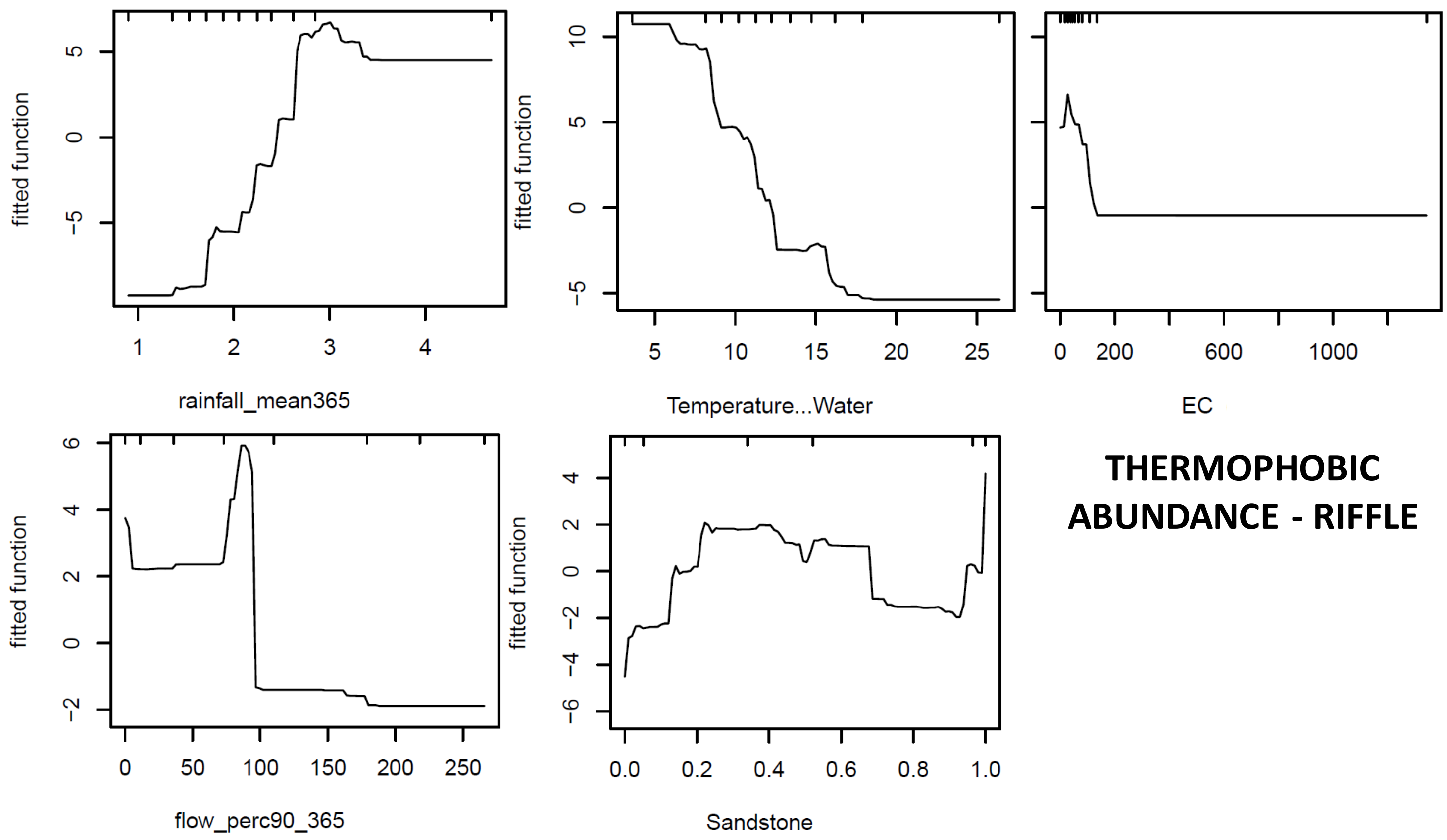

Figure 32. Partial dependence plots for the five most influential variables in the model for thermophobic taxa relative abundance in riffle.

For explanation of variables and their units see Appendix L. Y axes are on the logit scale and are centred to have zero mean over the data distribution. Rug plots at inside top of plots show distribution of sites across that variable, in deciles.

(i) Threshold estimation

Overall, threshold values estimated by the different methods and associated with the different ecological responses were relatively similar (Tables 20–22; Figure 33). TITAN tended to give lower values than other methods (e.g. for % Shading in the river in edge, conductivity in edge, % Volcanic sediment in riffle; Figure 33 (a), (f) and (j), respectively). However, when the 95% confidence intervals associated with TITAN values were taken into account, the differences with other methods were not as pronounced.

This suggests that the signals between environmental predictors and the different community responses (either multivariate or univariate) are robust. These results are very interesting from an ecological and management view point, because it guarantees that choosing one of the community metrics can help preserve other aspects of the community; they are not exclusive.

The hydrological variables produced the most variable threshold values. For instance, for the low flow in the edge sites and the whole community, the LINKTREE method resulted in two distinct thresholds (first split around 5–7 days and second split around 48–49 days, Figure 33(g); Figure 34). In contrast, only one threshold was identified for the low flow using TITAN which corresponded to the second one given using LINKTREE (Figure 33(g)). Notwithstanding this, when we looked at the output produced by TITAN, two other potential changes points (light and dark grey arrows in Figure 35(a)) in addition to the peak (black arrow) are highlighted. The first potential change point appeared to be consistent with the first split detected by LINKTREE.

Table 20. Threshold values of the “best predictors” associated with O/E scores.

The variables are described in Appendix L.
For % Shading of river, 1 = <5%, 2 = 6–25%, 3 = 26–50% , 4 = 51–75% , 5 = >76%.

Category of driver	O/E-score	QPL	
		Edge	Riffle
HABITAT	% Cover of Riparian Zone by Shrubs	23%	
CLIMATE	Temp max (annual mean)	20.87	18.95
	Rainfall (annual mean)		2.65
	Temp min (annual mean)		7.35
WATER QUALITY	EC (µS/cm)	118.85	146.55
	DO (mg/L)	7.44	
	Water Temperature (◦C)		13
HYDROLOGY	Flow (annual mean)	82.04	249
	Num. Days CTF (Year)	43	
	High Flow (10th Year) (num days)		10.5
LANDUSE GEOLOGY	Intense	1.5	
	Granite		5.5
	Other		2.5
LANDSCAPE	Elevation	from 600 m every 100 m	

Table 21. Threshold values of the "best predictors" associated with Thermophobic taxa relative abundance.

The variables are described in Appendix L.

Category of driver	Thermophobic Abundance	QPL	
		Edge	Riffle
HABITAT	Habscore	84	
CLIMATE	Temp max (annual mean)	18.95	
	Rainfall (annual mean)		2.65
WATER QUALITY	EC (µS/cm)	153.55	97
	Water Temperature (◦C)		12.5
HYDROLOGY	Flow Percentile	86.71	
	Flow (annual CV)	2.3	
	Low Flow (90th Year) (num days)		<5.5 / >95
LANDUSE GEOLOGY	Agriculture	<2 / 10 / >70	
	Sandstone		NA
LANDSCAPE	Elevation	from 600 m every 100 m	

Table 22. Threshold values of the "best predictors" associated with Whole community.

The variables are described in Appendix L.
Note that LINKTREE and TITAN deal with the macroinvertebrate assemblage, but LINKTREE uses all taxa, meanwhile TITAN is based on indicator taxa.
For periphyton, 1= <10%, 2 = 10–35%, 3 = 35–65%, 4 = 65–90%, 5 = >90% are categorical.

Category of driver	Whole community	LINKTREE (all taxa)		TITAN (indicator taxa)	
		Edge	Riffle	Edge	Riffle
HABITAT	Shading of river	>3(<2)		0.5 CI (0–2)	
	Periphyton		>3(<2)		2(1–2)
CLIMATE	Temp max (annual mean)	<19(>19)		19.0 CI (18.9–19.4)	
	Rainfall (annual CV)		<3.07 (>3.07)		3.0 CI (2.9–3.1)
WATER QUALITY	EC (µS/cm)	<158 (>155)		95.0 CI (73.7–119.5)	
	Water temperature (^{o}C)		<12.5 (>12.5)		11.89 CI (9.81–12.28)
HYDROLOGY	Low flow (90th year) (num days)	<5(>7) 48(>49)		42 CI (12–49)	
	Flow percentile	<69.3(>69.3) <46.2(>46.2) <87(>87.2)		69.5 CI (19.3–86)	
	High flow (10th year) (num days)		<0(>1)		2 CI (0–3)
LAND USE GEOLOGY	Intense	<2.33(>4.01)		7.35 CI (0–9.20)	
	Volcsed		<2.87 (>2.87)		0.025 CI (0–2.87)
LANDSCAPE	Elevation	<601(>612)	<601(>612)	640 CI (574–692)	612 CI (566–671)

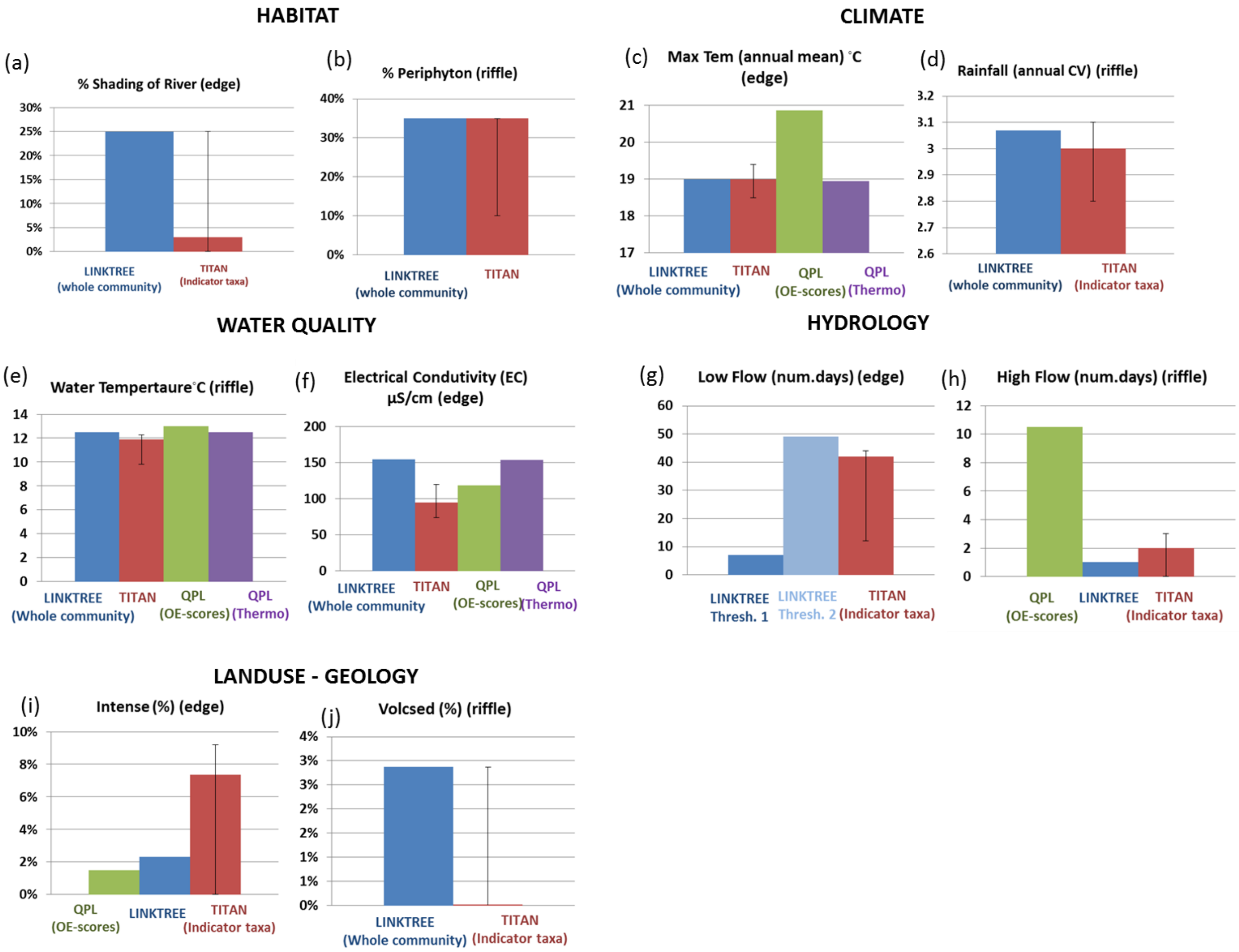

Figure 33. Bar plots showing thresholds produced by different methods and associated with different ecological responses.

Note that not all thresholds have been plotted. See Tables 20–22 for those missing in this figure.

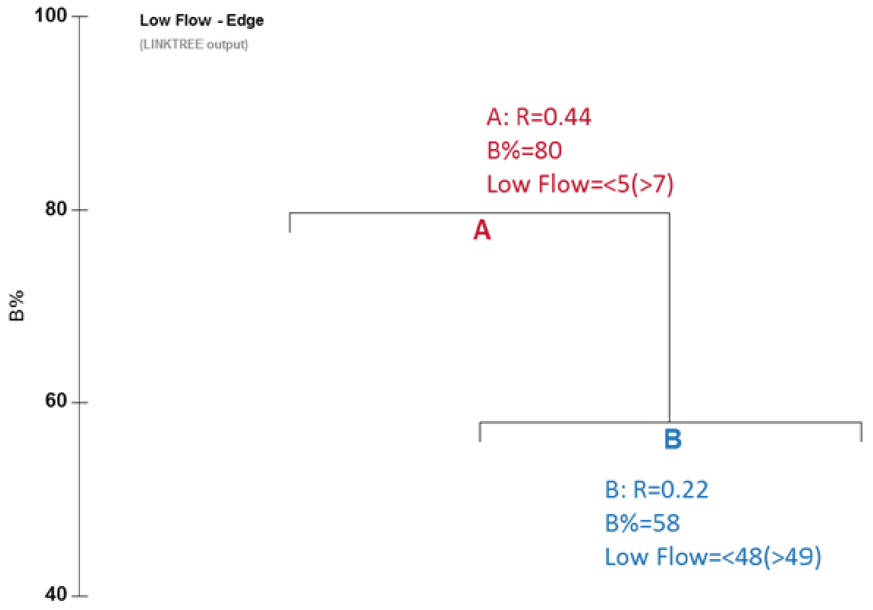

Figure 34. Linkage tree analysis (LINKTREE), showing divisive clustering of records from species compositions from the edge mesohabitat, constrained by inequalities on low flow (cluster plot).

Given for each split is the optimal ANOSIM R value (relative subgroup separation; from 0 to 1 being 1 maxima separation between groups) and B% (absolute subgroup separation, scaled to maximum for first division; it indicates how well separated the two groups of samples are, from 0 to 100 being 100 maxima separation). For each binary partition (A, B), first inequality defines group to left, second inequality (in brackets) group to right. Minimum group size= 66 and minimum split size= 4. Stopping rule of $p<0.05$ for the SIMPROF test.

In relation to the high flow in riffle, TITAN and LINKTREE produced similar thresholds for the whole community. This threshold differed from that produced for the O/E scores using the QPL method. In this case, it appears that the ecological response determines the threshold value. However, thresholds estimated for O/E scores should be interpreted with caution, because the relationship between high flow and O/E scores was weak (Figure 36(a)) and the confidence intervals around the slopes were broad for the best quantile (Figure 36(b)).

4.3.3 Water quality thresholds: guidelines versus estimated thresholds

As mentioned previously, three water quality attributes (EC, water temperature and dissolved oxygen) of the five considered in this study (Table 19) were related to the ecological responses. Only EC (conductivity) and dissolved oxygen could be compared with theoretical thresholds provided by the ANZECC guideline (Table 23) because theoretical values for the water temperature have not been established.

Overall, threshold values for conductivity that were estimated using different methods and different ecological responses were similar (Table 20–22; Figure 33(f)). These estimated thresholds are within the ranges of those proposed by the ANZECC guidelines (ANZECC/ARMCANZ 2000). The accepted thresholds provided by the ANZECC guideline for conductivity are very broad (Table 23). However, the conductivity thresholds

estimated in this study would be closer to the lower threshold proposed for lowland rivers in the ANZECC guidelines and the middle of the thresholds proposed for upland rivers. It is worth noting that most guidelines incorporate a safety factor and are likely to represent an overestimate of what is considered a suitable limit.

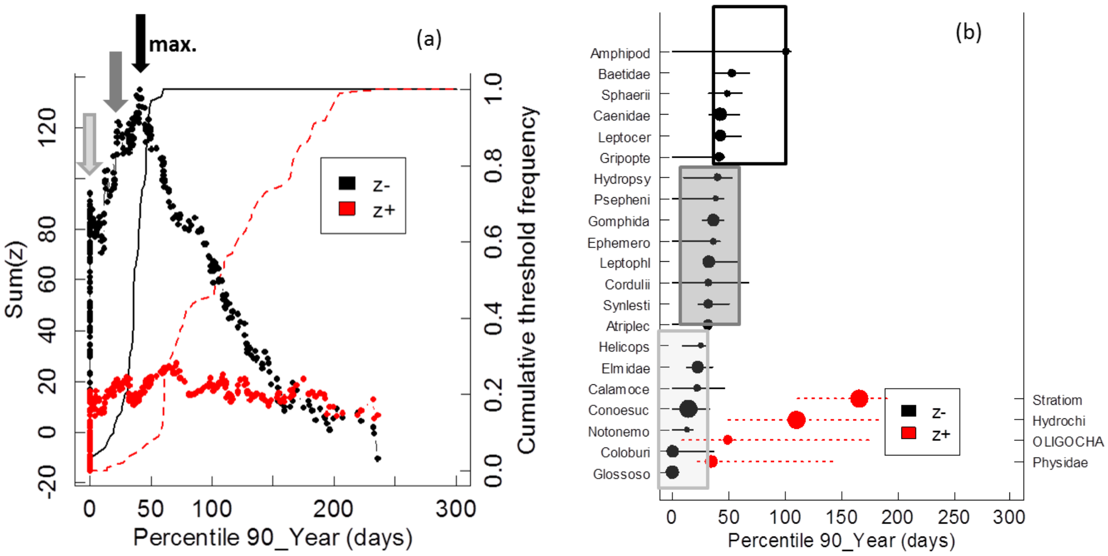

Figure 35. (a) TITAN of macroinvertebrate community response to low flow (num. days) in Edge showing (a) sum(z) across the Low Flow, and (b) significant indicator taxa.

In (a) TITAN sum(z-) and sum(z+) values correspond to candidate change points (□) along the gradient. Peaks in sum(z-) correspond to locations along the gradient where synchronous declines of taxa occur, with the most substantial peak occurring at 42 days (maximum remarked with a black arrow). Solid and dashed lines represent the cumulative frequency distribution of change points (xcp [thresholds]) among 100 bootstrap replicates for sum(z-) and sum(z+), respectively. Arrows indicating two potential change points (light and dark grey arrows) and the peak(blackarrow) are shown.
In (b) significant (purity ≥ 0.95, reliability ≥ 0.90, $p < 0.05$) indicator taxa are plotted in increasing order with respect to their observed change point. Black symbols correspond to negative (z-) indicator taxa, whereas red symbols correspond to positive (z+) indicator taxa. Symbol sizes are in proportion to magnitude of the response (z scores). Horizontal lines overlapping each symbol represent 5th and 95th percentiles among 100 bootstrap replicates. Individual taxa are included in boxes in approximate correspondence with sum values of the potential change points in (a).

Table 23. Comparative table of the thresholds provided by the ANZECC guidelines (ANZECC/ARMCANZ 2000) and the thresholds estimated in our study (these latter specified for each type mesohabitat — edge and riffle — and the type of the ecological response associated with each method).

<table>
<tr><th>Method</th><th>Source</th><th>Salinity µS/cm</th><th>Salinity µS/cm</th><th>Dissolved oxygen mg/L</th></tr>
<tr><th></th><th></th><th>Upland rivers</th><th>Lowland rivers</th><th>Upland rivers</th></tr>
<tr><td>Theoretical</td><td>Guidelines thresholds (ANZECC/ARMCANZ)</td><td>30–350</td><td>125–2200</td><td>6</td></tr>
<tr><td rowspan="2">QPL
O/E scores</td><td>Edge</td><td colspan="2">118.85</td><td>7.44</td></tr>
<tr><td>Riffle</td><td colspan="2">146.55</td><td></td></tr>
<tr><td rowspan="2">QPL
Thermophobic abundances</td><td>Edge</td><td colspan="2">153.55</td><td></td></tr>
<tr><td>Riffle</td><td colspan="2">97</td><td></td></tr>
<tr><td rowspan="2">LINKTREE
Whole community</td><td>Edge</td><td colspan="2">155</td><td></td></tr>
<tr><td>Riffle</td><td colspan="2"></td><td></td></tr>
<tr><td rowspan="2">TITAN Indicator taxa</td><td>Edge</td><td colspan="2">95 (73.7–119.5)</td><td></td></tr>
<tr><td>Riffle</td><td colspan="2"></td><td></td></tr>
</table>

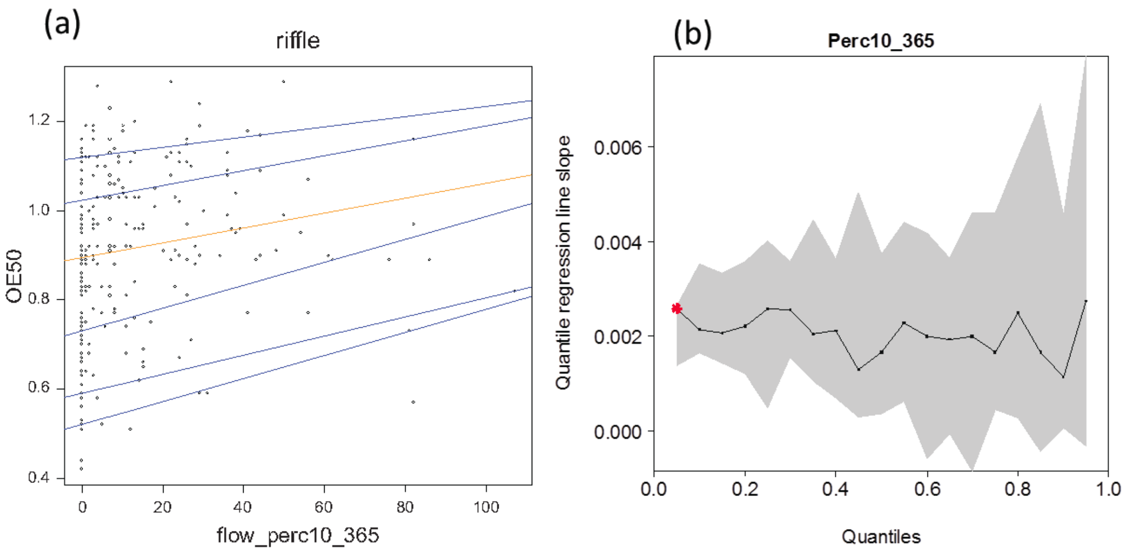

Figure 36. (a) Scatterplots of O/E scores and high flow in riffle. The 0.10–0.90 quantile regression lines are given (tau=0.50 in orange). **(b) Quantile regression line slopes and 90% confidence intervals (grey area) for different quantiles (from 0.05 to 0.95 with a 0.05 step width). * = t**he quantile selected for further analysis.

Having a guideline which provides trigger values is helpful, particularly from a management viewpoint. However, they can be too general and therefore not be useful for specific ecosystems and / or for identifying the ecological responses. For instance, it is known that within the same biological community, species have different conductivity tolerance levels. In this sense, TITAN could be considered a better predictor of thresholds than the other methods, because it is able to discriminate the indicator taxa

into “tolerant” (z+) and “sensitive” (z-). This may be why the conductivity threshold derived using TITAN was slightly lower than others.

Figure 37 shows the responses of the tolerant (z+) and sensitive (z-) taxa to electrical conductivity. “Sensitive taxa” (z-) declined sharply between 0.08 µS/cm (Glossosomatidae) and 320 µS/cm (Acarina) (Figure 37(b)), resulting in a sum(z-) change point of 95.0 µS/cm with a relatively narrow 95% confidence interval (CI) (73.7–119.5 µS/cm). On the other hand, positive (z+) indicators (i.e. “tolerant taxa”) increased between 52 µS/cm (Vellidae) and 1003 µS/cm EC (Parastacidae), resulting in a distinct sum(z+) peak at 78.4 µS/cm; however, this increase was not very distinct and the 95% CI covered a broad range (69.6–1251.5 µS/cm) which prevented a clear threshold from being identified. It should be noted however that for our purposes, we are interested only in the threshold for the negative response (z- or sensitive taxa).

TITAN provides numerous advantages for estimating thresholds, since it is able to consider the individual responses of indicator taxa (based on purity and reliability). Furthermore, it provides a range of uncertainty around the threshold value (i.e. as lower and upper CI). However, it has recently received some criticism, mainly based on the requirement for removal of rare taxa (further information in Cuffney et al. 2010). And as stated before, TITAN only selects a single peak (maximum) to be the threshold, although others may also be important (e.g. low flow, Figure 35).

In contrast to TITAN, LINKTREE considers the whole macroinvertebrate assemblage with no restrictions in relation to rare taxa. However, no distinction between “sensitive” and “tolerant” taxa is possible, potentially leading to less distinct thresholds.

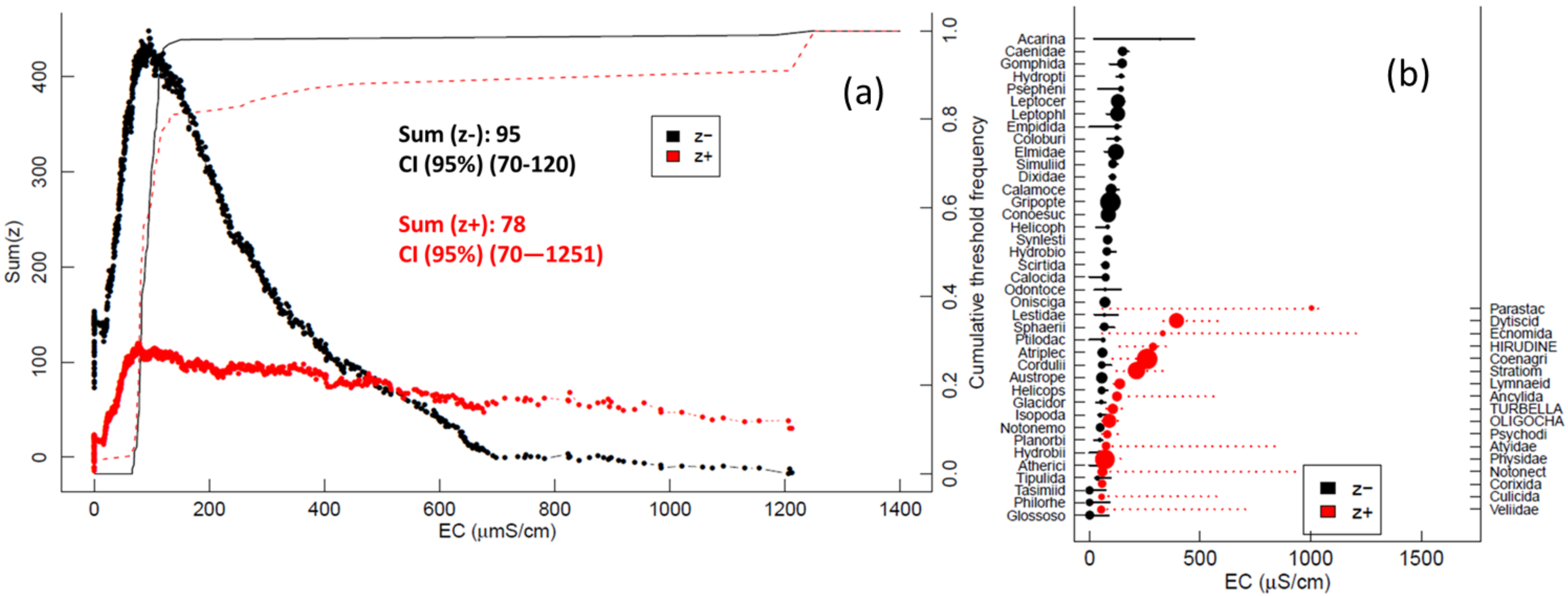

Figure 37. (a) TITAN of macroinvertebrate community response to electrical conductivity (EC, μS/cm) in Edge showing (a) sum(z) across the EC, and (b) significant indicator taxa

In (a) TITAN sum(z-) and sum(z+) values correspond to candidate change points (□) along the gradient. Peaks in sum(z-) correspond to locations along the gradient where synchronous declines of taxa occur, with the most substantial peak occurring at 95 µS/cm. Solid and dashed lines represent the cumulative frequency distribution of change points (xcp [thresholds]) among 100 bootstrap replicates for sum(z-) and sum(z+), respectively.
In (b) significant (purity ≥ 0.95, reliability ≥ 0.90, $p < 0.05$) indicator taxa are plotted in increasing order with respect to their observed change point. Black symbols correspond to negative (z-) indicator taxa, whereas red symbols correspond to positive (z+) indicator taxa. Symbol sizes are in proportion to magnitude of the response (z scores). Horizontal lines overlapping each symbol represent 5th and 95th percentiles among 100 bootstrap replicates.

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Figure 38 shows the output of LINKTREE for the whole macroinvertebrate assemblage in relation to conductivity in edge. It is not possible to give an uncertainty value for a split in a dendrogram. However, LINKTREE produces a test to check whether such a split should be made or not (SIMPROF test; Clarke et al. 2008). If there is no statistical evidence of heterogeneity in the samples below any particular node in the dendrogram then there is no basis for splitting that group further. This provides confidence in the splits that are produced. In the dendrogram (Figure 38) three possible thresholds (splits) at 158, 39 and 52 µS/cm are shown. However, the ANOSIM R statistic values were very low for the splits B and C. This implies that despite significant heterogeneity in the samples between C and D (i.e. differences between the macroinvertebrate assemblages) the very low R values preclude us considering these splits. So, the conductivity threshold value in this case was determined based on the first split.

The main advantage of LINKTREE is that splits can be constrained for more than one environmental driver which is the main limitation of most of the procedures. In this study, we considered only one predictor to be able to compare with the other two approaches, TITAN and QPL.

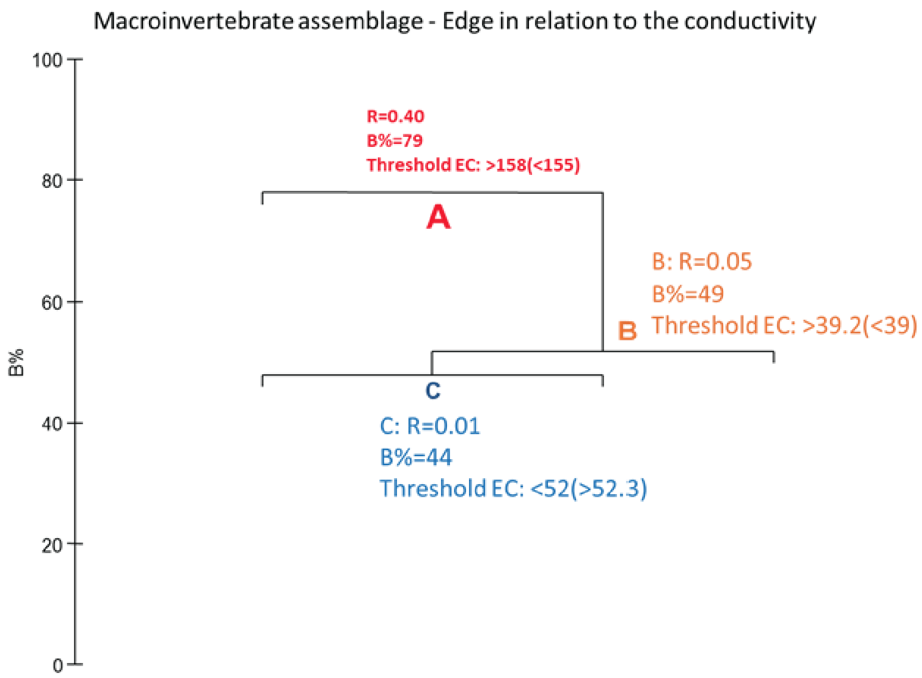


Figure 38. Linkage tree analysis (LINKTREE), showing divisive clustering of records from species compositions from Edge mesohabitat, constrained by inequalities on coefficient of variation of the electric conductivity (EC) (cluster plot). Given for each split is the optimal ANOSIM R value (relative subgroup separation; from 0 to 1 being 1 maxima separation between groups) and B% (absolute subgroup separation, scaled to maximum for first division; it indicates how well separated the two groups of samples are, from 0 to 100 being 100 maxima separation). For each binary partition (A, B, C…), first inequality defines group to left, second inequality (in brackets) group to right. Minimum group size= 27 and minimum split size= 4. Stopping rule of p<0.05 for the SIMPROF test.

LINKTREE and TITAN deal with the macroinvertebrate assemblage. Aggregated community indicators (or metrics) can also provide very useful information. Both types of ecological

responses are important in developing an understanding of how organisms respond to environmental predictors (Barbour et al. 1999). Therefore, we produced conductivity thresholds for the metrics: O/E scores in edge and Thermophobic taxa relative abundances in riffle. We found that the scatterplot for both responses did not show a wedge-shaped distribution for which QPL is specially designed (Figure 39(a) and Figure 40(a)). However, differences in the response from the lower to the higher quantiles were significant ($p<0.05$ for both responses), including significant differences in the slopes of the upper quantiles (0.75 and 0.90, $p<0.05$). The best quantile (largest with the narrowest CI) resulted in 0.85 and 0.7 for O/E scores in edge and thermophobic taxa relative abundance in riffle, respectively. Note that the latter has a high CI variability.

Therefore, the estimated conductivity threshold in this case has to be carefully considered. This uncertainty is similar to the weak relationship detected in the BRT between the thermophobic taxa relative abundance in riffle and conductivity.

No specific conductivity threshold was evident in the scatterplot for O/E scores or Thermophobic taxa relative abundance. However, the locally weighted quantile regression models for different bandwidths showed a change at around 100 µS/cm (Figure 39(b) and Figure 40(b)).The quantile regression tree analysis (GUIDE result) resulted in a threshold of 118.85 µS/cm for the 0.85 quantile in O/E scores for edge and 96.6 µS/cm for the 0.7 quantile in Thermophobic taxa relative abundance for riffle.

It was not possible to compare the dissolved oxygen threshold, because it was only related to the O/E scores in edge (Table 20). The dissolved oxygen threshold value was estimated with QPL and yielded a value slightly higher than the theoretical value proposed by the ANZECC guideline (Table 23). The uncertainty surrounding the slopes was generally wide, which makes us carefully consider this threshold (Figure 41(c)). The locally weighted quantile regression model showed the shape of a sigmoidal curve, at the most optimal bandwidth (i.e. red, Figure 41(b)). This is the typical curve describing a regime shift. We observed that the change occurred at ~7 mg/L. The quantile regression tree analysis (GUIDE result) resulted in a threshold of 7.44 mg/L.

The results obtained in our investigation of thresholds gave us the confidence to use the empirically derived thresholds in Bayesian Networks.

4.3.4 Use of thresholds in the Bayesian Networks

One of the difficulties in constructing Bayesian Network is the discretisation of the continuous variables in categories. To discretise continuous variables in the BN, we have applied the thresholds to establish these categories. We discretised the variables as “above” and “below” the estimated threshold. Tables 24–29 show the categories of the “best predictors” and the thresholds for each of them which we used in the Bayesian Networks for macroinvertebrates.

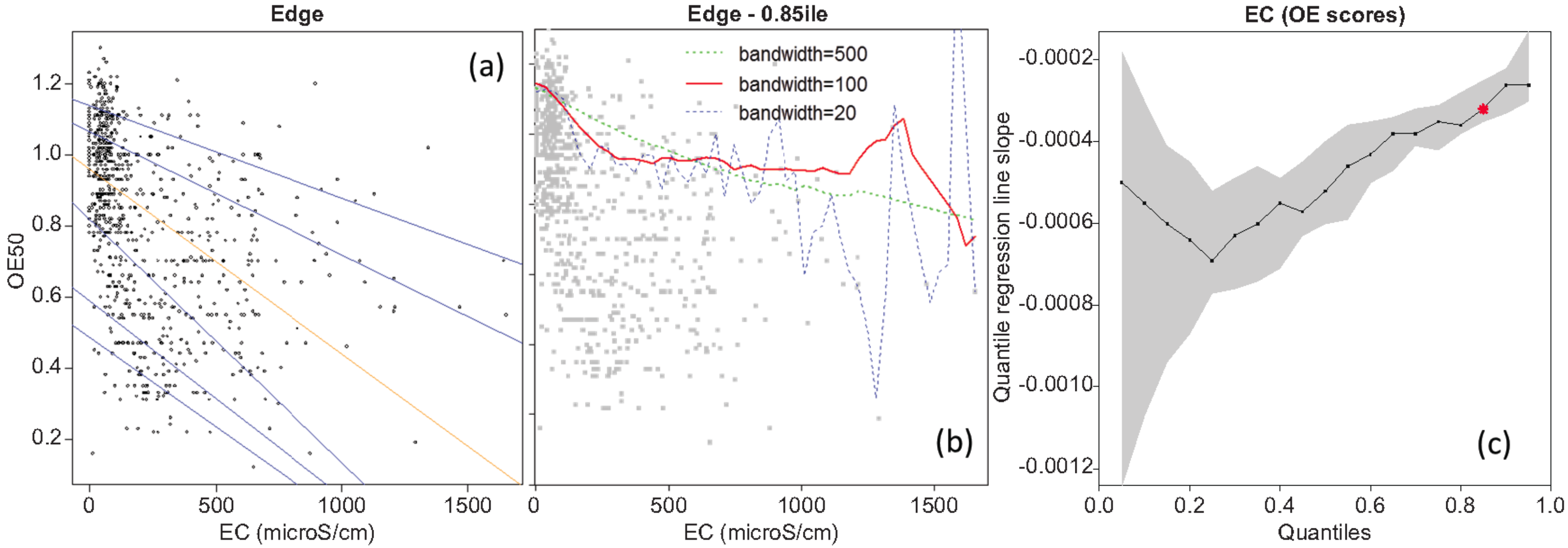


Figure 39. (a) Scatterplots of O/E scores and electrical conductivity (EC) in Edge. The 0.10–0.90 quantile regression lines are given (tau=0.50 in orange). **(b) Locally weighted quantile regression models with different bandwidths. (c) Quantile regression line slopes and 90% confidence intervals (grey area) for different quantiles (from 0.05 to 0.95 with a 0.05 step width).** * = the quantile selected for further analysis.

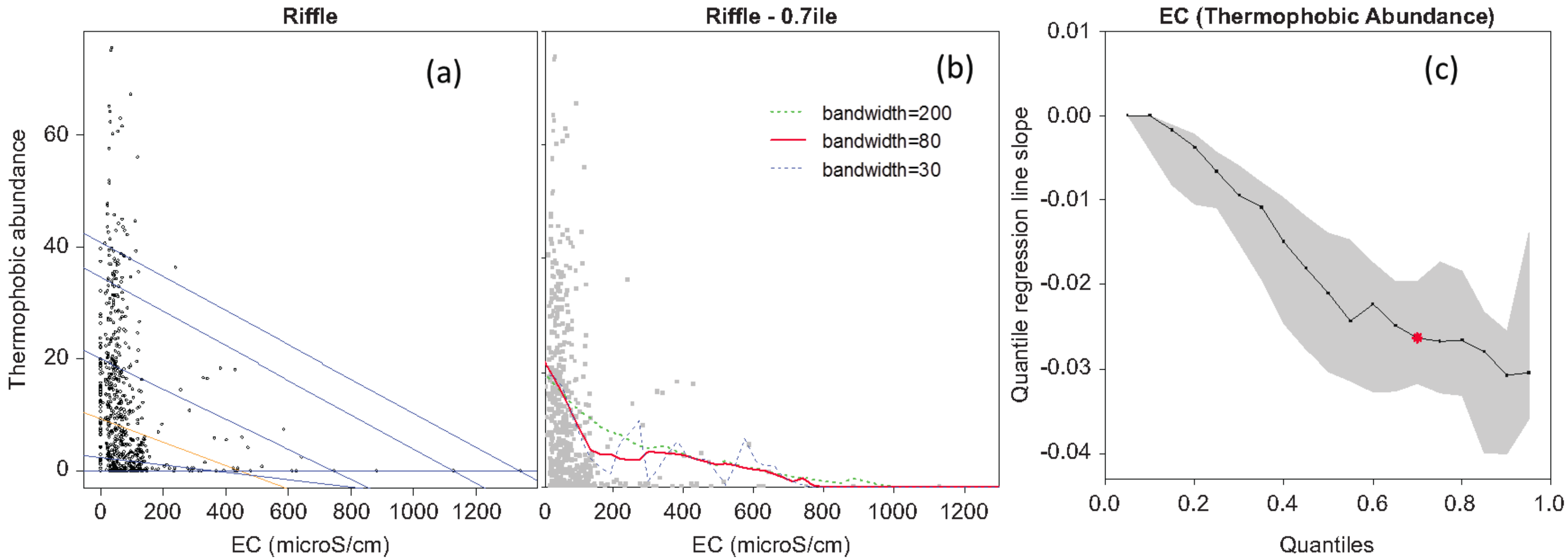


Figure 40. (a) Scatterplots of Thermophobic taxa relative abundance and electrical conductivity (EC) in riffle. The 0.10–0.90 quantile regression lines are given (tau=0.50 in orange). **(b) Locally weighted quantile regression models with different bandwidths. (c) Quantile regression line slopes and 90% confidence intervals (grey area) for different quantiles (from 0.05 to 0.95 with a 0.05 step width).** * = the quantile selected for further analysis.

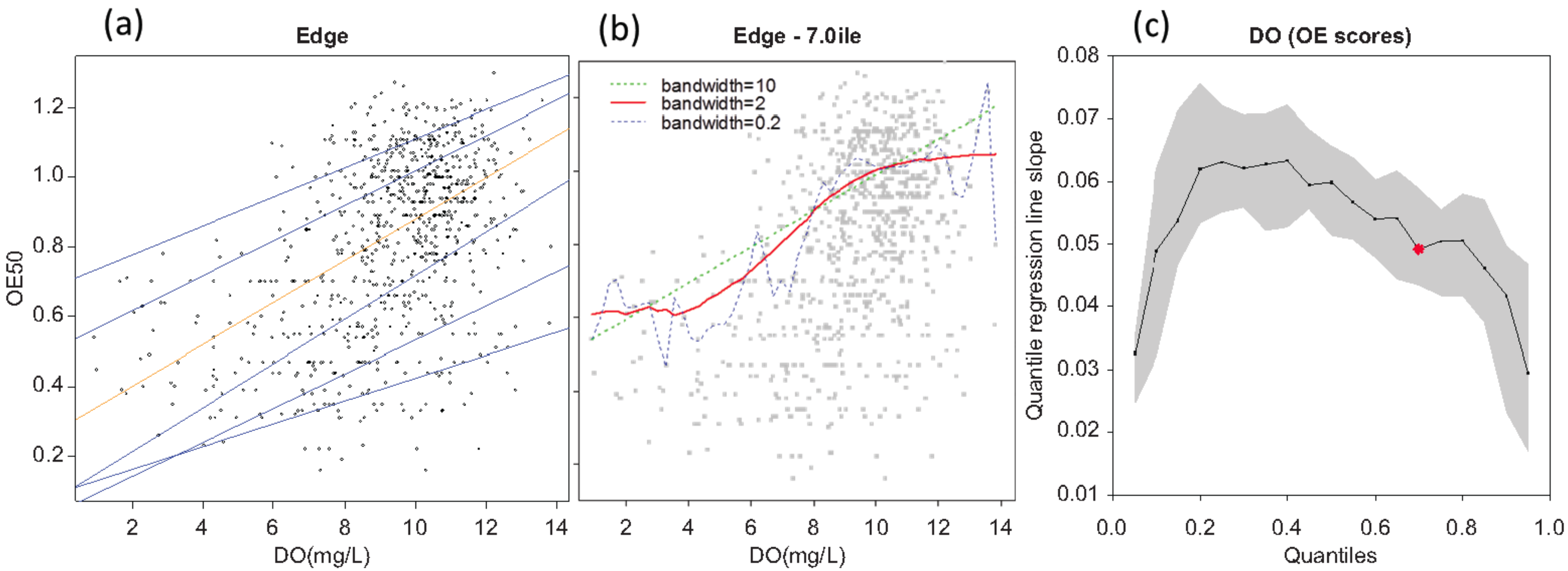


Figure 41. (a) Scatterplots of O/E scores and dissolved oxygen (DO) in edge. The 0.10–0.90 quantile regression lines are given (tau=0.50 in orange). **(b) Locally weighted quantile regression models with different bandwidths. (c) Quantile regression line slopes and 90% confidence intervals (grey area) for different quantiles (from 0.05 to 0.95 with a 0.05 step width).** * = the quantile selected for further analysis.

Table 24. Categorisation of the best predictors associated with O/E score in edge, based on the estimated thresholds

CATEGORY OF DRIVER	O/E-SCORE in EDGE	No change	Change 1	Change 2	Change 3
HABITAT	% Cover of Riparian Zone by Shrubs	0 - 23%	23-60%	60 - 100%	
CLIMATE	Temp max (annual mean)	<19	19-21	>21	
WATER QUALITY	EC	0-118	118-350	350-800	
	DO	0-7.44	7.44-9	>9	
HYDROLOGY	Flow (annual mean)	0-82	82-300	300-500	
	Num. Days CTF (Year)	0-20	20-43	43-80	>80
LANDUSE GEOLOGY	Intense	0-2%	2-10%	10-50%	>50%
LANDSCAPE	Elevation	<600		>600	

Table 25. Categorisation of the best predictors associated with O/E score in riffle based on the estimated thresholds

CATEGORY OF DRIVER	O/E-SCORE in RIFFLE	No change	Change 1	Change 2	Change 3
CLIMATE	Rainfall (annual mean)	<2.65 per day (965 year)		>2.65 (965)	
	Temp max (annual mean)	<19		>19	
	Temp min (annual mean)	<7		>7	
WATER QUALITY	Water Temperature	<13		>13	
	EC	0-147	147-350	350-800	>800
HYDROLOGY	Flow (annual mean)	249	249 - 400	400-800	>800
	High Flow (10th Year)	<10.5	10.5-30	30-60	>60
LANDUSE GEOLOGY	Granite	0-5%	5-10%	10-50%	50-70%
	Other	0-3%	3-10%	10-40%	0.40%
LANDSCAPE	Elevation	<482	482-674	674-872	>872

Table 26. Categorisation of the best predictors associated with Thermophobic taxa relative abundance in edge based on the estimated thresholds

CATEGORY OF DRIVER	Thermophobic Abundance in EDGE	No change	Change 1	Change 2	Change 3	Change 4
HABITAT	Habscore	<84		>84		
CLIMATE	Temp max (annual mean)	<19		>19		
WATER QUALITY	EC	0-154	154-350	350-800	>800	
HYDROLOGY	Flow Percentile	0-40	40-87	> 87		
	Flow (annual CV)	2.30-4	4-6	> 6		
LANDUSE GEOLOGY	Agriculture	<2 %	2-10%	10-50%	50 - 70%	>70%
LANDSCAPE	Elevation	<600	600-682	>682		

Table 27. Categorisation of the best predictors associated with Thermophobic taxa relative abundance in riffle based on the estimated thresholds

CATEGORY OF DRIVER	Thermophobic Abundance in RIFFLE	No change	Change 1	Change 2	Change 3
CLIMATE	Rainfall (annual mean)	<2.65 per day (965 year)		>2.65 (965)	
WATER QUALITY	Water Temperature	<12.5		>12.5	
	EC*	0-97	97-350	350-800	>800
HYDROLOGY	Low Flow (90th Year)	<5.5	5.5-95	> 95	
LANDUSE GEOLOGY	Sandstone	NA			
LANDSCAPE	Elevation	<549	549-744	744-902	>902

Table 28. Categorisation of the best predictors associated with the macroinvertebrate assemblage in edge based on the estimated thresholds

CATEGORY OF DRIVER	Whole Community in EDGE	No change	Change 1	Change 2	Change 3	Change 4
HABITAT	Shading of River	<3%	3-25%	>25%		
CLIMATE	Temp max (annual mean)	<19		>19		
WATER QUALITY	EC	<70	70-120	>120		
HYDROLOGY	Low Flow (90th Year)	<12	12-49	>49		
	Flow Percentile	<19	19-46	46-70	>70	
LANDUSE GEOLOGY	Intense	<2%	2-7%	2-7%	7-9%	>9%
LANDSCAPE	Elevation	<700		>700		

Table 29. Categorisation of the best predictors associated with the macroinvertebrate assemblage in riffle based on the estimated thresholds

CATEGORY OF DRIVER	Whole Community in RIFFLE	No change	Change 1	Change 2	Change 3
HABITAT	Periphyton	<35%	35-65%	65-90%	>90%
CLIMATE	Rainfall (annual CV)	<3		>3	
WATER QUALITY	Water Temperature	<12		>12	
HYDROLOGY	High Flow (10th Year)	0-3		>3	
LANDUSE GEOLOGY	Volcsed	0-3%		>3%	
LANDSCAPE	Elevation	<700		>700	

4.4 Fish

4.4.1 Approach 2: Background

In addition to the macroinvertebrate ecological responses, various native fish species were also modelled. The fish community of the Upper Murrumbidgee catchment is severely degraded with only 12 native fish species present (Lintermans 2002). In combination with macroinvertebrate communities these remaining native fish are important indicators of freshwater ecosystem health.

In the case of fish also, we conducted a bottom-up approach, which combined expert opinion with statistical tools (univariate). As with the macroinvertebrate models, only the best predictors related to the fish responses were used to structure the Bayesian Network. For fish, thresholds were not estimated; instead we used theoretical values selected from published literature, expert opinion and guidelines.

4.4.2 Methods

(i) Ecological response

Data on the presence and absence of native fish and total species richness was collated for the study area. The fish species used in modelling and their respective presences and absences are listed in Table 30.

(ii) Environmental predictors

Environmental predictor data were collected as outlined in the macroinvertebrate section. We collated data for predictor variables identified as important based on expert opinion. However, environmental data associated with smaller scale habitat predictors (e.g. tree cover) were not available. We removed highly correlated (>0.7) predictor variables. In total we used 14 environmental predictor variables in BRT modelling (Table 31).

Unlike macroinvertebrates, the number of records of the response variables was sufficiently large relative to the predictor variables. Thus for fish, all categories of the predictors were modelled together. We produced eight models, one for each species plus and one for species richness.

Table 30. Fish dataset description and environmental variables used in Boosted regression tree modelling

Species	Presences	Absences
Australian Smelt	30	226
Golden Perch	34	222
Macquarie Perch	44	212
Mountain Galaxias	80	176
Trout Cod	15	241
Two-spined Blackfish	91	165
Western Carp Gudgeon	30	226

Table 31. Environmental predictors used in fish models.

Variables are fully described in Appendix L.

Environmental predictor variable
ALTITUDE
Agriculture
Days.Cease-to-flow.Year.
flow_cv365
flow_perc10_365
flow_perc90_365
Intense
Natural
rainfall_cv365
rainfall_mean365
Temp
tempmax_cv365
tempmax_mean365
Turbidity

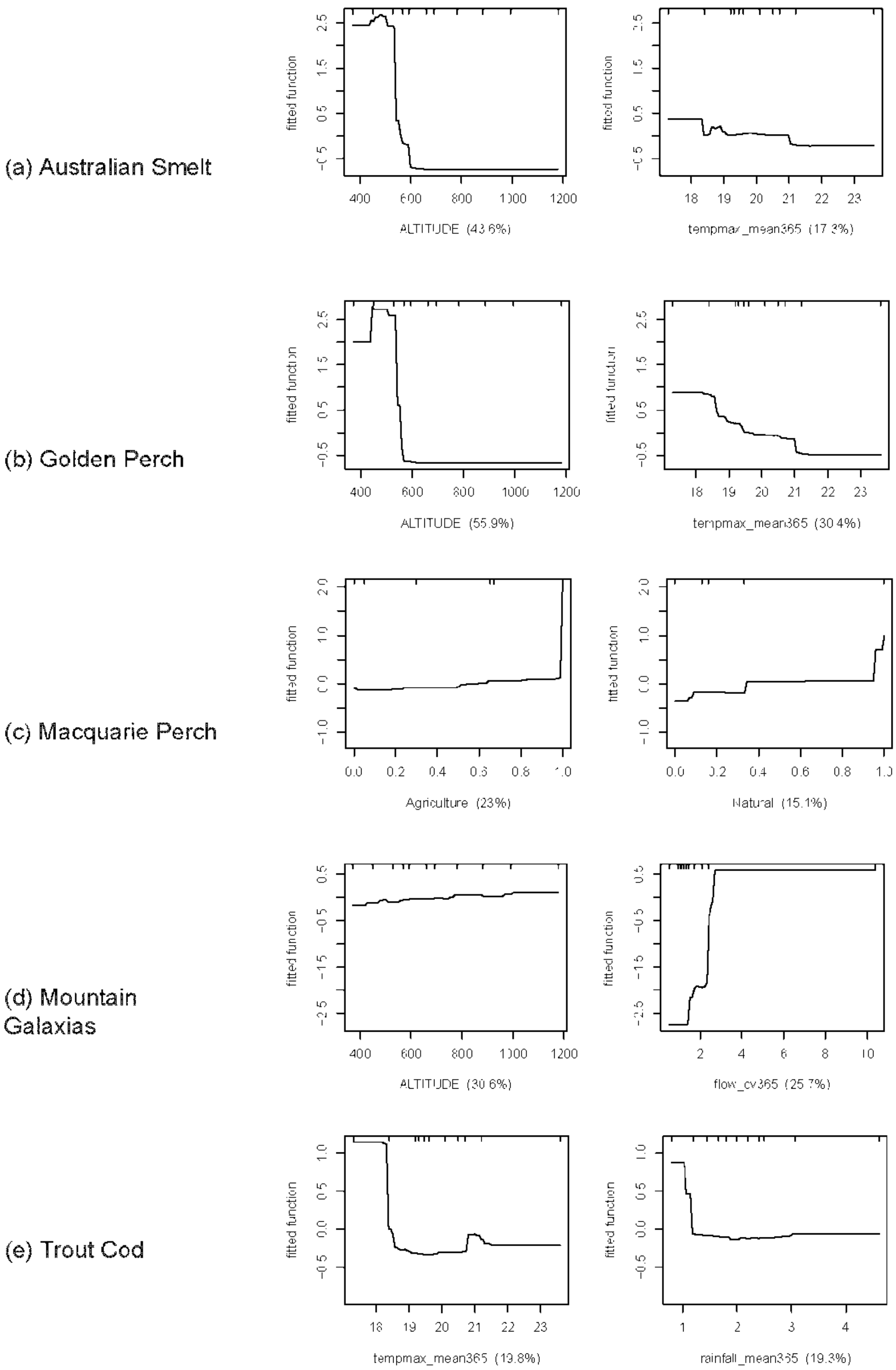
(iii) Statistical methods

We selected the environmental predictors to be used in Bayesian Networks by following the same approach we used for macroinvertebrates, using BRT modelling (see section 4.3.1(ii)). The presence or absence of fish were modelled in the form of logistic regression (after Elith, Leathwick & Hastie 2008) and species richness as a Poisson response type.

Variables which explained greater than 10% (Table 32) were used in the Bayesian Network model for fish. Trout Cod and fish species richness were not included in the Bayesian network model because they had no environmental predictors that explained greater than 10% (Table 32). In addition, dissolved oxygen was also included in the Bayesian Network model because it is known to be an important limiting factor for native fish. The relationship of dissolved oxygen with the fish species modelled was based on expert opinion.

4.4.3 Results

All BRT models for the fish species explained more than 50% of the variation in fish occurrence, except for Trout Cod (Table 32). Model performance was good for all species with cross-validated ROC scores of over 0.7 (Table 32). Altitude (Elevation) was consistently the variable of most relative importance (Table 32). Environmental variables relating to land cover (Agriculture and Natural), flow variability (flow_cv365), rainfall mean and variability (rainfall_mean365 and rainfall_cv365) and Temperature (tempmax_mean365), were also important predictor variables (Table 32).


(a) Australian Smelt
fitted function
ALTITUDE (43.6%)
tempmax_mean365 (17.3%)
(b) Golden Perch
ALTITUDE (55.9%)
tempmax_mean365 (30.4%)
(c) Macquarie Perch
Agriculture (23%)
Natural (15.1%)
(d) Mountain Galaxias
ALTITUDE (30.6%)
flow_cv365 (25.7%)
(e) Trout Cod
tempmax_mean365 (19.8%)
rainfall_mean365 (19.3%)

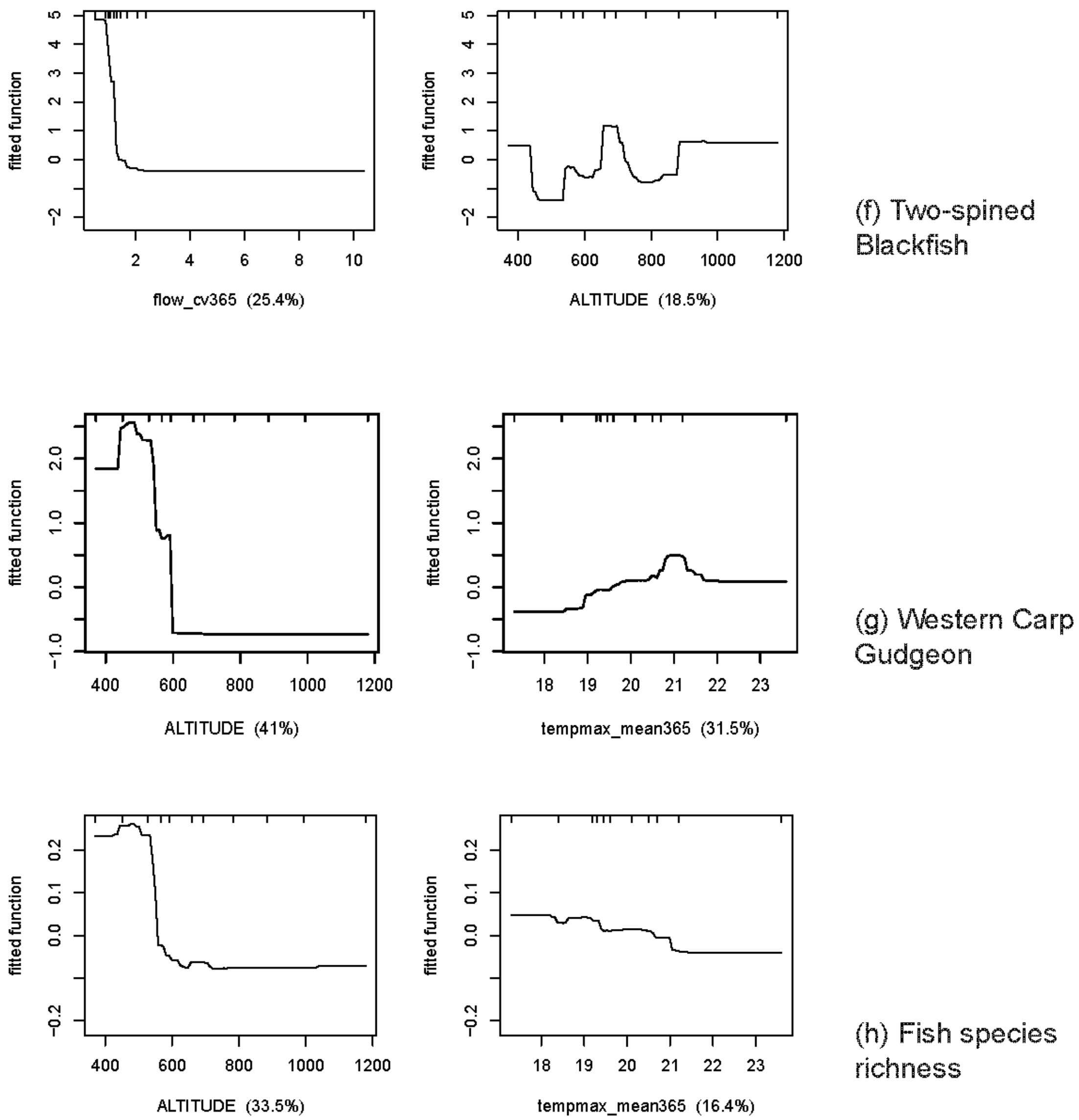


Figure 42. Partial plots from boosted regression tree models for the two most influential environmental predictor variables for each fish species modelled

All species, except Two-spined Blackfish and Mountain Galaxias, had a negative relationship with the variable 'Altitude', declining as altitude increased (Figure 42(a)). Similarly the fish species examined, except for Western Carp Gudgeon, had a negative relationship with Temperature (tempmax_mean365), with lower likelihood of occurrence as tempmax_mean365 increased (Figure 42). Macquarie Perch had a positive relationship with both the variables 'Agricultural' and 'Natural land cover'. The positive relationship with agricultural land cover is likely influenced by one area in the catchment with high agricultural cover (Figure 42(c)).

Table 32. Importance of environmental predictor variables as a percentage explained by each within BRT models. In bold are variables contributing >10%, which were used in the BN for fish.

	Australian Smelt	Golden Perch	Macquarie Perch	Mountain Galaxias	Trout Cod	Two-spined Blackfish	Western Carp Gudgeon	Fish species richness
ALTITUDE	**30.24**	**37.69**	9.48	**22.44**	5.61	**18.20**	**28.84**	**18.63**
Agriculture	2.37	0.72	**15.57**	5.70	3.27	**11.94**	3.56	1.01
Days.Cease to flow..Year.	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
flow_cv365	1.88	0.43	8.04	**18.82**	5.72	**24.91**	1.32	7.68
flow_perc10_365	0.78	0.74	1.55	0.91	1.08	1.38	0.28	1.42
flow_perc90_365	0.63	0.03	1.29	1.14	0.90	1.11	0.16	2.24
Intense	0.19	1.02	0.02	0.02	0.77	0.01	2.72	0.02
Natural	7.42	3.21	**10.26**	**11.65**	2.33	6.79	4.13	3.88
rainfall_cv365	6.02	0.98	5.22	2.62	6.67	**12.18**	2.77	3.30
rainfall_mean365	4.35	1.50	6.87	2.54	8.46	9.21	2.28	4.61
Temp	2.16	0.22	2.66	2.46	0.16	6.63	1.41	2.83
tempmax_cv365	1.32	0.11	3.03	0.36	0.11	0.67	0.44	0.47
tempmax_mean365	**12.02**	**20.52**	6.39	3.28	8.67	4.91	**21.53**	9.14
Turbidity	0.01	0.23	0.01	1.34	0.14	0.18	0.16	0.37

Table 33. Performances of the BRT models for the fish species

	Australian Smelt	Golden Perch	Macquarie Perch	Mountain Galaxias	Trout Cod	Two-spined Blackfish	Western Carp Gudgeon	Fish species richness
Mean explained (%)	69.4	67.4	70.4	73.3	43.9	98.1	69.6	55.6
Mean total deviance	0.723	0.783	0.918	1.242	0.446	1.302	0.723	0.248
Mean residual deviance	0.221	0.255	0.272	0.332	0.249	0.025	0.22	0.11
Estimated cv deviance (se)	0.438 (0.053)	0.402 (0.047)	0.604 (0.034)	0.648 (0.048)	0.396 (0.041)	0.31 (0.055)	0.453 (0.052)	0.186 (0.016)
Training data correlation	0.872	0.849	0.899	0.912	0.698	0.999	0.879	0.783
CV correlation (se)	0.618 (0.074)	0.721 (0.041)	0.622 (0.035)	0.739 (0.03)	0.224 (0.078)	0.903 (0.017)	0.581 (0.074)	0.484 (0.069)
Training data ROC score	0.992	0.987	0.994	0.992	0.983	1	0.993	–
CV ROC score (se)	0.928 (0.025)	0.956 (0.013)	0.885 (0.018)	0.926 (0.013)	0.776 (0.058)	0.991 (0.004)	0.993 (0.015)	–

4.5 Synthesis and conclusions

- We used a bottom-up approach to select predictors for both fish and macroinvertebrates Bayesian Network (BN) models. This resulted in a limited numbers of drivers which were strongly related to the ecological responses. This reduction in predictor variables allowed us to construct parsimonious BNs and make full use of available data.
- The predictor variables which were selected varied between ecological responses (macroinvertebrates and fish) and between edge and riffle (macroinvertebrates). This is important for management, because it highlights the specificity and diversity of relationships in freshwater ecosystems. This also highlights the need to be clear about objectives and endpoints for predictive modelling.
- Using macroinvertebrates, threshold values across methods were generally similar, despite focusing on different community measures.
- The discretisation of the continuous predictor variables for macroinvertebrate BN models was based on statistically derived thresholds. For fish, the discretisation of the BN was based on expert opinion, literature, guidelines and data distribution. However, both approaches were valid and provided useful information (see Section 6). They represent different ways of addressing the problem. For macroinvertebrates, the discretisation of the variables was less practical than for fish, because the habitat requirements of macroinvertebrates are less clearly understood and thresholds of response within the Upper Murrumbidgee catchment have been questioned by local agency staff. Therefore, we tried to fill this knowledge gap by identifying locally relevant and specific thresholds of macroinvertebrate community responses in the catchment under future scenarios.
- Water quality and hydrological characteristics were identified as important predictor variables. However, the magnitude of the relationship and the important predictor variables differed based on the response. Therefore indicators of the community (e.g. O/E scores), may not be relevant to others (e.g. Thermophobic taxa relative abundance). It is good to have alternative methods for threshold identification, such as TITAN and LINKTREE, which take into account the Whole community.
- Comparing empirically derived thresholds against theoretical values is good practice, but not very common (Huggett 2005). Recently, debate about thresholds has grown because of its implications for management (Lindenmayer & Luck 2005; Bestelmeyer 2006). Despite this, most thresholds still come from a theoretical framework or from historical data distributions (as is the case with the ANZECC guidelines). Empirical thresholds cater specifically to the ecological response of interest and therefore may be of more use than general guidelines, particularly when it comes to the need to predict responses. The disadvantage of using empirically derived thresholds from a management perspective is that they are specific to a single ecological response and unless there is co-incidence across ecological responses and across regions the implementation within guidelines becomes complicated. However, empirically derived thresholds can help refine the theoretical thresholds as proposed in the ANZECC guidelines.

5. THE BAYESIAN NETWORK

In this section we outline the construction of the Bayesian network that links the projected water quality and quantity changes with ecosystem changes. The structure of the models is described, as is the discretisation of each node in the network. The section concludes with an assessment of key uncertainties and limitations of the networks.

5.1 Bayesian Network structure and features

We used the commercially available software package Netica™ Version 4.16 (Norsys Software Corporation, Vancouver, Canada, 2007) to construct the Bayesian Networks (BNs) and model causal relationships between multiple environmental factors, including climate change and management scenarios, environmental attributes, water quality attributes and macroinvertebrate populations and native fish species.

5.1.1 Node selection and development

We constructed six macroinvertebrate models (Figures 43–48) which represented both edge and riffle communities separately, and a single native fish models (Figure 49). Edge and riffle macroinvertebrate communities were modelled separately because the influencing environmental variables (as defined externally from the BNs, see Section 4) differed between the two.

All models contained the primary input nodes Region, Climate Scenario and Management Scenario. All models also contained the intermediate node Flow Distribution because of its use in defining water quality changes with climate (see Section 3). Other nodes in each of the BNs were selected based on the strength of their influence on the response variable, as calculated externally from the BNs (see Section 4). Other intermediate nodes were model dependent, and selected based on the analysis of the key drivers (see Section 4).

5.1.2 Formation of node states

To enable parameter relationships to be analysed, each node in the BN was allocated a series of discrete “states” in a summary table (McCann, Marcot & Ellis 2006). For parent (input) nodes, each of these states had a “prior” (expected) probability associated with it (Morawski 1989). For each child (intermediate or output) node, a conditional probability distribution was calculated for each combination of values of the parent nodes. Data were converted to text files and imported into the BN. Using these data, Bayesian learning was used to determine the relationships between parent and child nodes to populate the conditional probability tables (Marcot et al. 2006).

In Netica, Bayesian learning uses an expectation maximization algorithm to iteratively process data until model fit is maximised or the desired number of iterations is reached (Norsys 2007). Following Bayesian learning, the relationships between variables were represented as probabilities in the conditional probability tables of the BN. Conditional probability tables are used in the BN to quantify the relationships between different variables (i.e. between the parent and child nodes; Smith et al. 2007).

Section 4 above describes in detail the development of the thresholds we used in each of the nodes to develop node states, and Tables 34–36, at the end of this section, summarise the model nodes and their relative states.

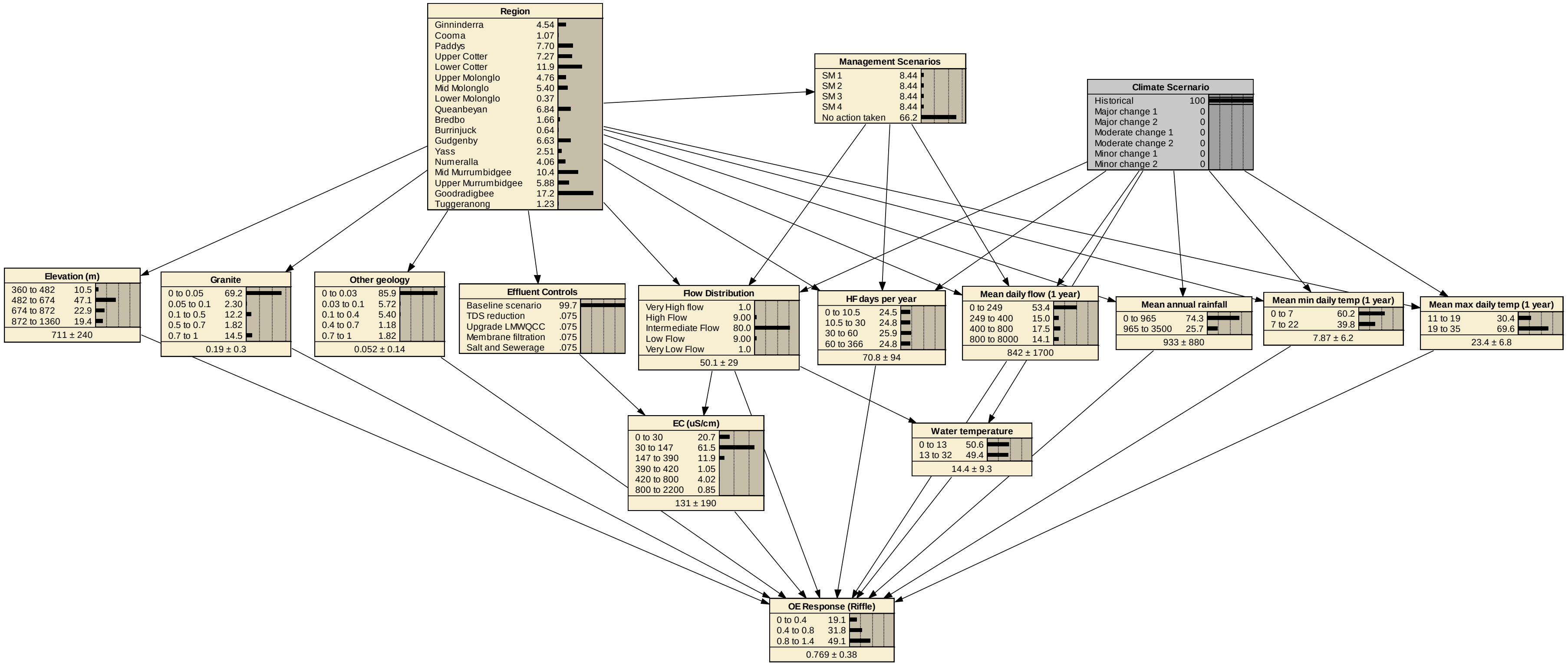


Figure 43. A compiled Bayesian Network macroinvertebrate model for O/E scores in riffle habitat. Model shown is an example only and represents all management scenarios and all regions and historical climate.

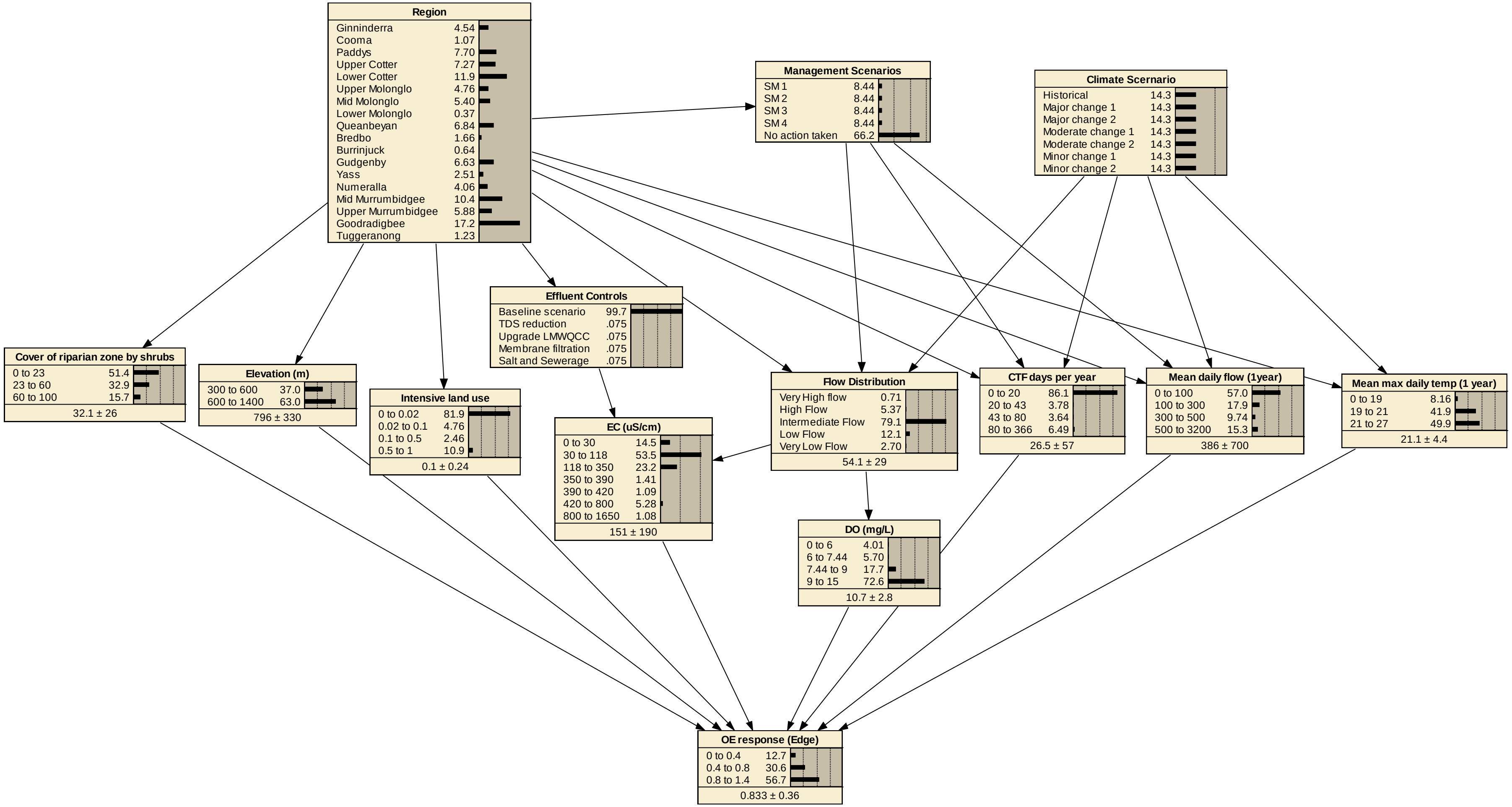


Figure 44. A compiled Bayesian Network macroinvertebrate model for O/E scores in edge habitat. Model shown is an example only. Results represent all regions, management scenarios and climate scenarios.

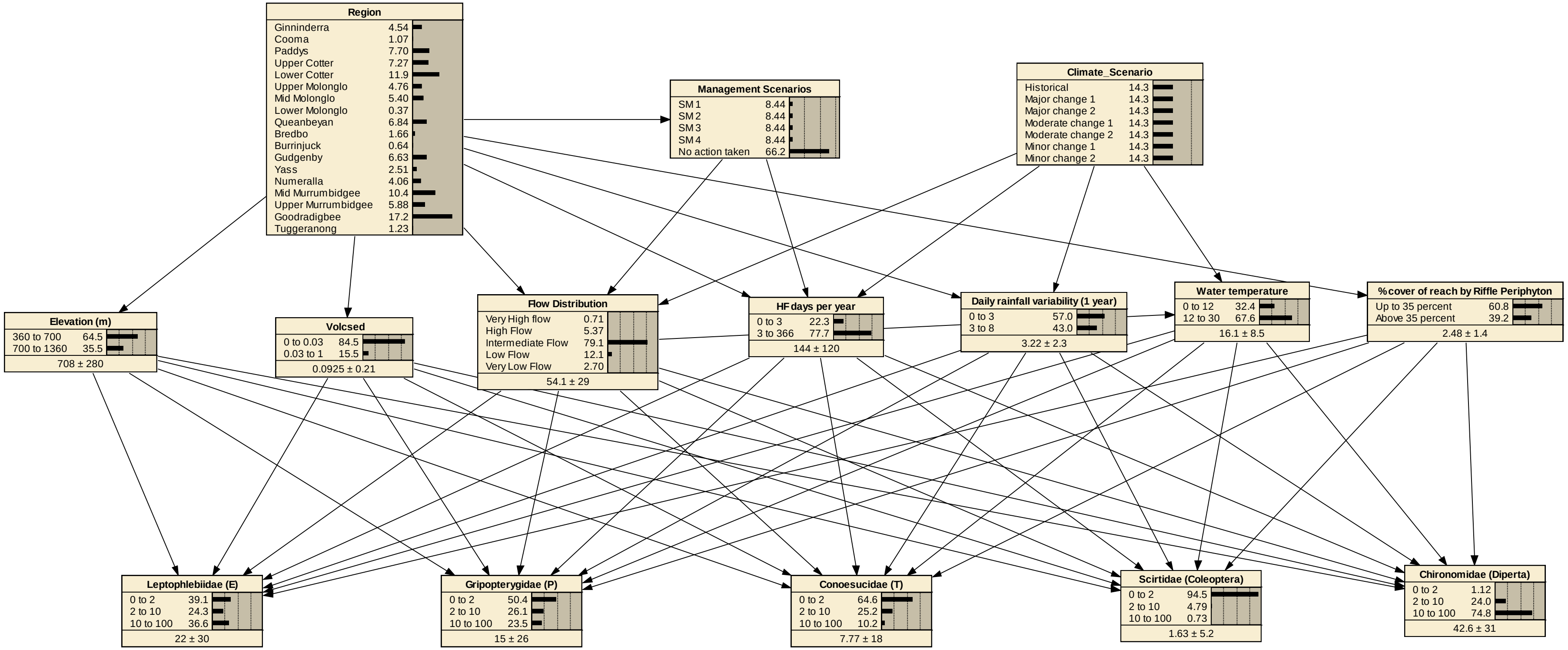


Figure 45. A compiled Bayesian Network of the macroinvertebrate community indicator model for riffle habitat. Model shown is an example only. Results represent all regions, management scenarios and climate scenarios.

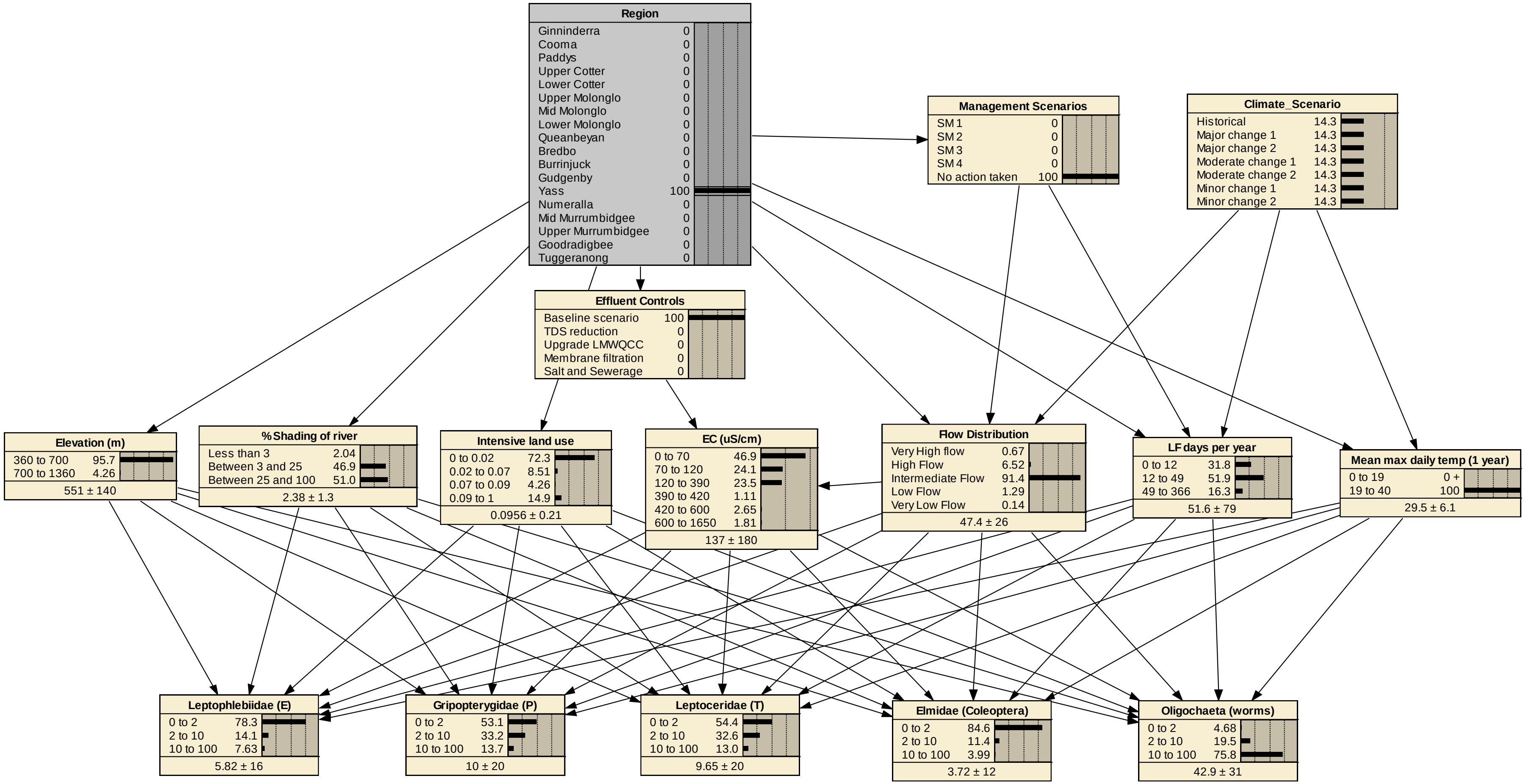


Figure 46. A compiled Bayesian Network of the macroinvertebrate community indicator model for edge habitat. Model shown is an example only and represents the Yass region.

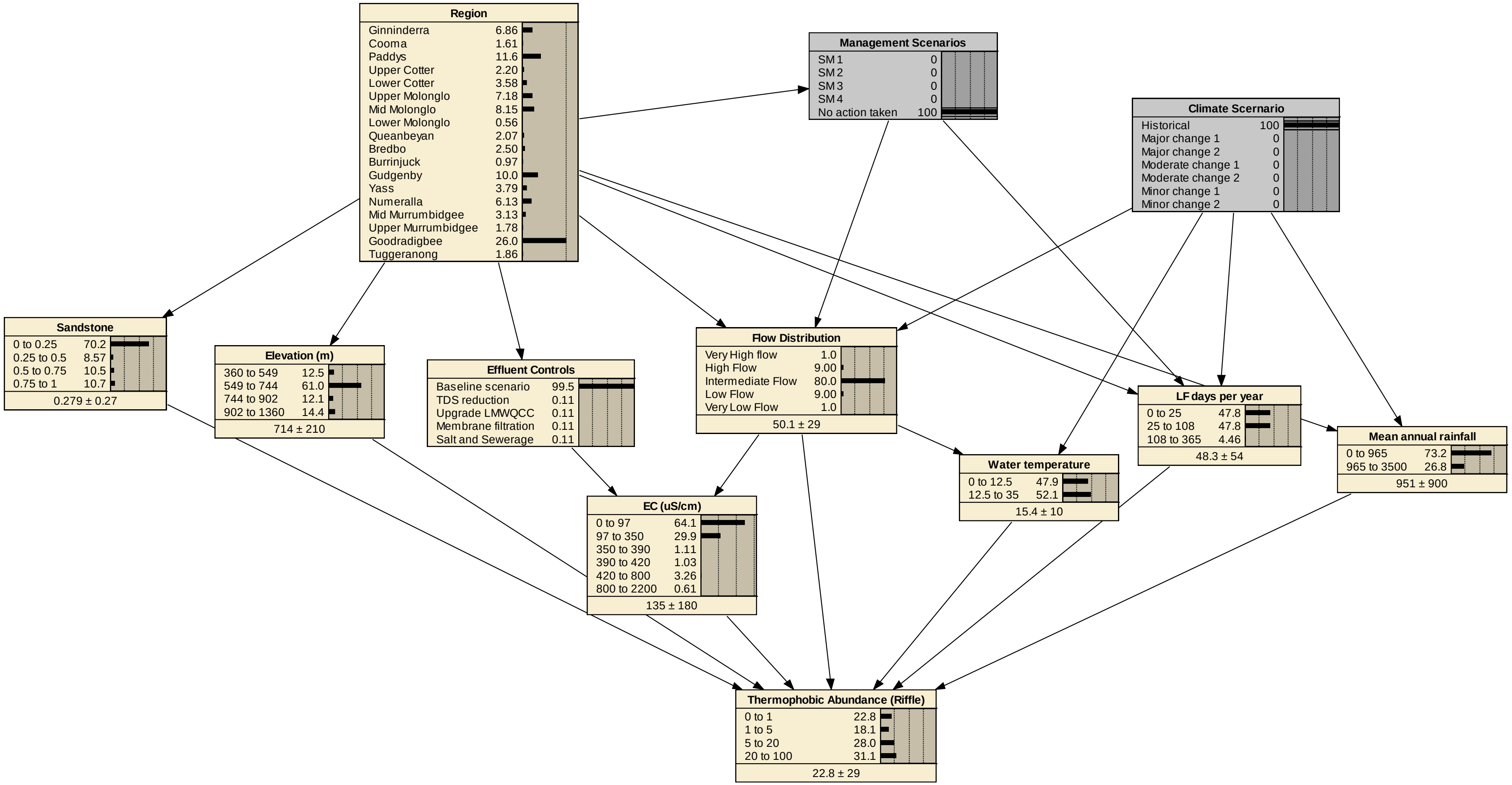


Figure 47. A compiled Bayesian Network macroinvertebrate model of Thermophobic taxa relative abundance in riffle habitat. Model shown is an example only and represents all regions, no management scenarios and historical climate.

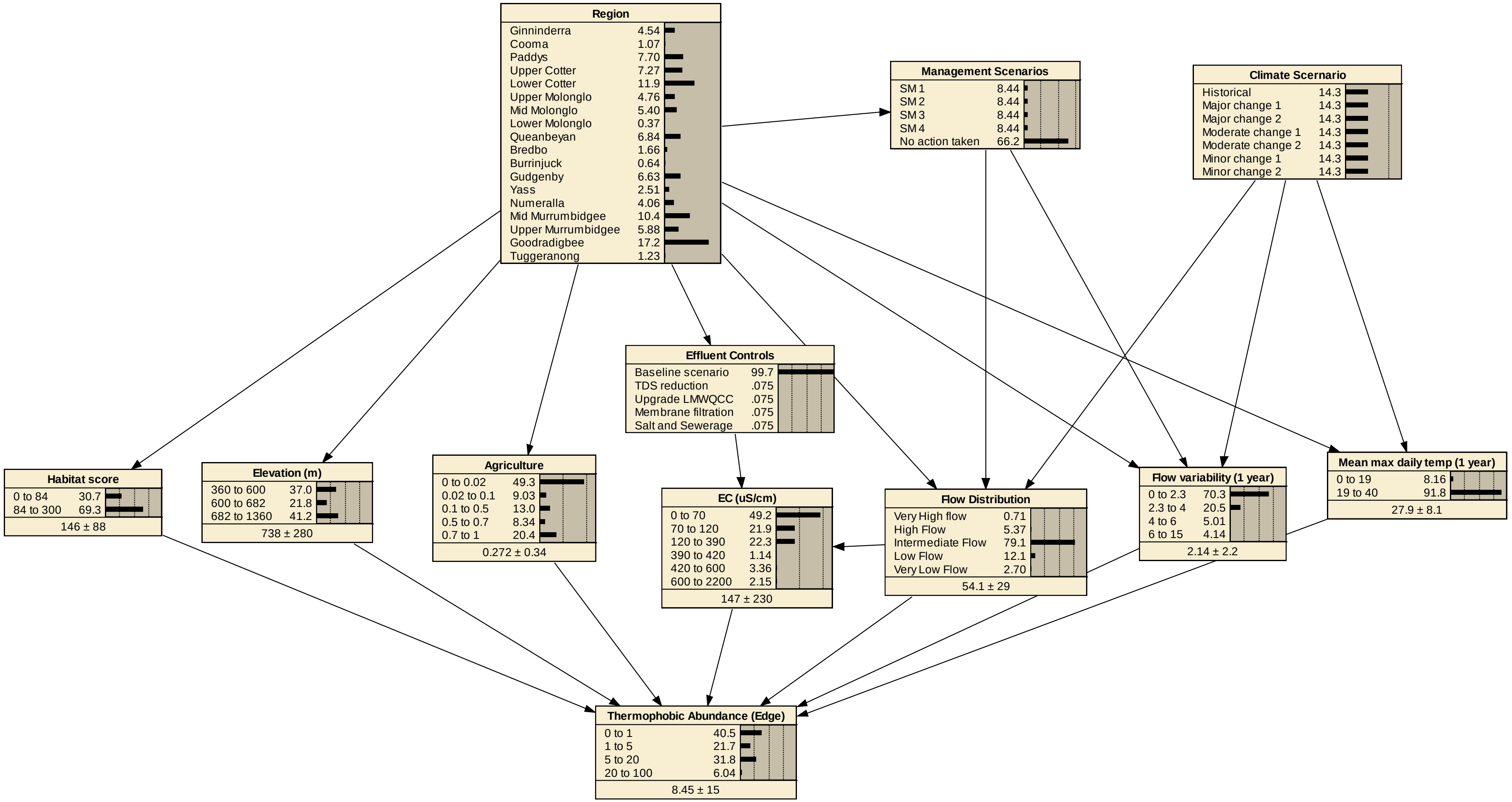


Figure 48. A compiled Bayesian Network macroinvertebrate model of Thermophobic taxa relative abundance in edge habitat. Model shown is an example only. Results represent all regions, management scenarios and climate scenarios.

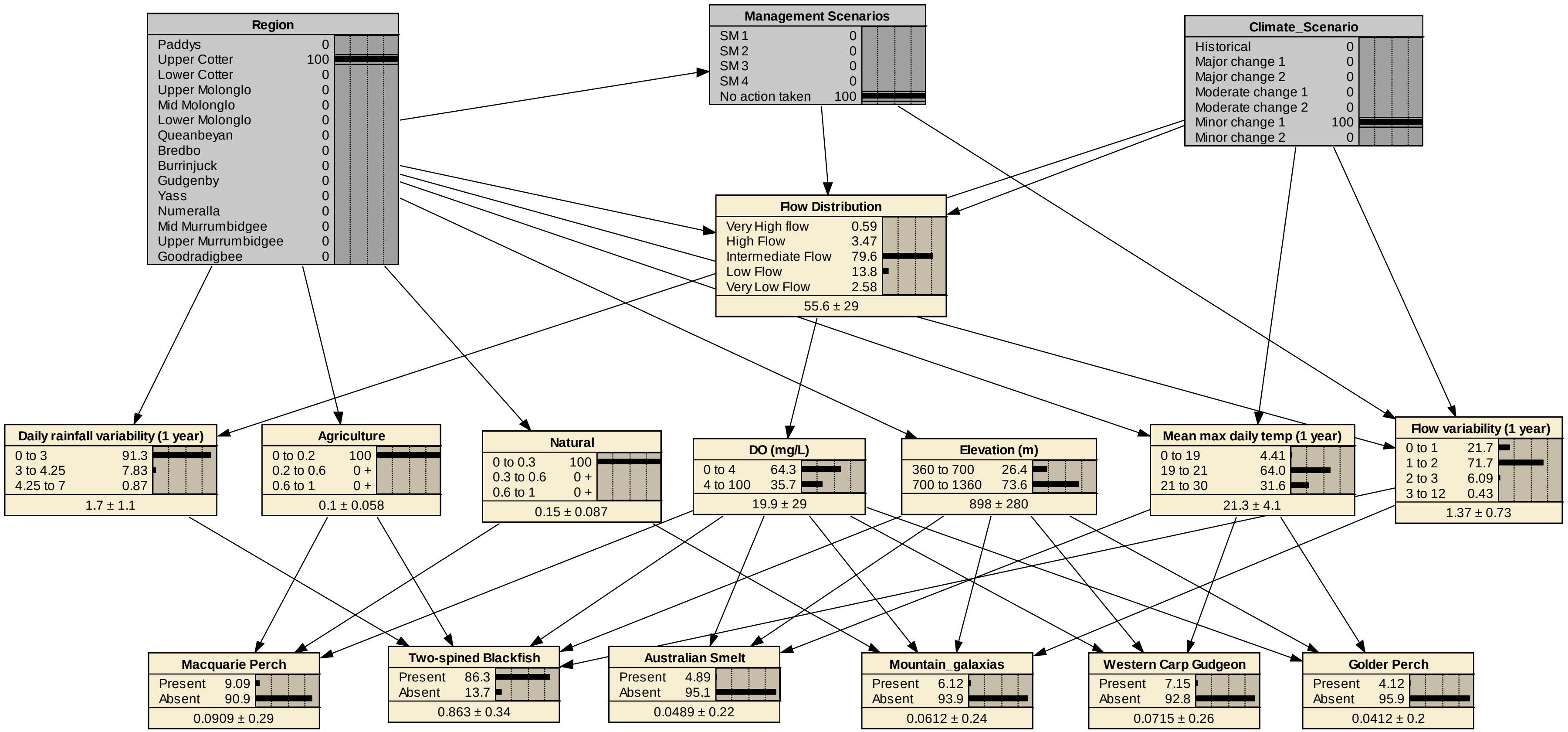


Figure 49. A compiled Bayesian Network model for fish. Model shown is an example only and represents the Upper Cotter region, no management actions and a minor change in climate with a 1ºC temperature increase.

5.1.3 *Primary input nodes (all models)*

(i) Region

We identified a total of 18 regions in the Upper Murrumbidgee catchment: Ginninderra, Cooma, Paddys, Bredbo, Numeralla, Yass, Upper and Lower Cotter, Mid and Upper Murrumbidgee, Lower, Mid and Upper Molonglo, Queanbeyan, Tuggeranong, Goodradigbee, Gudgenby and Burrinjuck. Section 3 above discusses the regionalisation of the study area.

These regions formed the 18 states of the Region node, which was the primary input node for all BN models. This enabled regional analysis of the data using the BNs, which was deemed the most practical option from a management perspective. Through selecting a region in the Region node, only values for this region would be shown in all nodes of the model. Region directly influenced all flow and environmental variables (Figures 43–49).

(ii) Climate Scenario

In the conceptual models, environmental input variables such as mean daily rainfall, pan evaporation, mean daily maximum air temperature and bushfire impact were shown separately, as was the parameter ‘water demand’. In developing the BNs, we used these parameters in the external hydrological modelling (see Section 3), with flow time series produced for each of the states of the parent node Climate Scenario. Modelled data for sample sites across the Upper Murrumbidgee were sourced from ACTEW, and from iCAM (a research group at the ANU) using data sourced from SEACI (South Eastern Australia Climate Initiative), and from BoM (Bureau of Meteorology) databases (BoM 2012).

Of the 31 climate scenarios that were calculated using external hydrological models, we chose six for input into the BN models. These climate change scenarios represented minor, moderate and major changes in flow conditions (Section 3). The scenarios formed six of the seven states of the Climate Scenario input node, and we used them to investigate the impact of different climate scenarios on water quality and ecological responses. The seventh default value, “historical”, was also included to enable a comparison of predicted climate scenarios for historical climate conditions (see Table 34). In the BN, the Climate Scenario node influenced water and air temperature and all flow nodes.

The BNs were not constructed to enable assessment of the impact of extreme events, such as long term droughts, high intensity storms (rainfall values of very high intensity) or floods (suspended high rainfall frequency). We considered the possibility of including extreme events, such as major storms, but did not include them in the final model because of possible incompatibility with climate scenarios. Inclusion of extreme events in the modelling framework is the subject of further research by the project team.

(iii) Management Scenarios

The Management Scenarios node was used to encompass the effect on flow of the four management adaptation alternatives we had devised (see Section 2[z]). Management alternatives affected flow management in the catchment, and they could be investigated under different climate scenarios in the BN.

The four future management alternatives vary human water supply and demand and flow regime (see Section 2.4 and Figure 10), and a fifth option (no action taken) represents

current management practice. The management alternatives are relevant to five regions only (Mid and Upper Murrumbidgee, Upper and Lower Cotter and Queanbeyan) and represent the regulated rivers of the catchment. The Management Scenarios node formed a primary input node into all flow nodes (Table 34; Figures 43–49).

(iv) Effluent Control

Management options which directly related to water quality were identified separately in the BN. Effluent control options and their potential effects on water quality were sourced from the Canberra Sewerage Strategy 2010–2060 (ActewAGL 2011) (see Section 2). The influence of the Effluent Control node was therefore inherently regionalised to the sampling locations close to the effluent output site in the Lower Molonglo region only.

The Effluent Control node impacted directly on the water quality node containing electrical conductivity values (given in the Electrical Conductivity node). The Effluent Control node included five management options, which were used to develop node states:

two different sewage treatment options (the use of a bioreactor or membrane filtration for nutrient removal, and an upgrade to the LMWQCC to improve water quality),

reduction in total dissolved solids output,

a combination of salt and sewage control, and

a baseline of the current system output (see Section 2).

Each option was defined by its impact on electrical conducivity. The implementation of a bioreactor or membrane filtration, an upgrade to the LMWQCC or reduction in total dissolved solids output to improve water quality, reduced electrical conductivity below 420 µS/cm. The combination of sewerage upgrade and salt removal reduced electrical conductivity below 390 µS/cm. The baseline scenario reflected the current sewage treatment and electrical conductivity values in the river and was the default value for all other regions.

5.1.4 Intermediate nodes (Model specific)

We assessed a total of 95 riffle and 82 edge variables for their relevance and impact on the selected ecological responses (outlined above in Section 4 and in Appendix L). Only the most influential variables were selected for the BN models. Variables therefore differed in each model depending on the ecological response they were associated with (see Section 4).

(i) Flow Distribution

The Flow Distribution node was the only intermediate node used in all BNs. The Flow Distribution node was used to link the strong interaction between flow and water quality attributes under different climate scenarios. For discretisation in the BN, we selected five categories of flow distribution:

- very low flow (equivalent to the 99th percentile flow from the historical flow time series);
- low flow (equivalent to the 90<99th percentile flow from the historical flow time series);

- intermediate flow (equivalent to the 10<90th percentile flow from the historical flow time series);
- high flow (equivalent to the 1<10th percentile flow from the historical flow time series); and
- very high flow (equivalent to the 1st percentile flow from the historical time series).

Percentile flows were defined as being the percentage of time a particular stream flow was equalled or exceeded, using values from the historical flow records at the sampling sites. Hence the 99th percentile flow was equalled or exceeded 99% of the time, thus representing very low flows. Defining the flow categories based on percentile flows from the historical time series effectively standardises the data from rivers of differing size.

The Flow Distribution node was a child node of the primary input nodes Region, Management Scenario and Climate Scenario, so that the impact of management and climate scenarios on flow (and thus water quality) could be investigated separately or in combination with each other for each region. The conditional probability tables of the Flow Distribution node were based on externally modelled data that captured the impact of management alternatives and climate scenarios on flow. This modelled data was then entered directly into the BN to populate the conditional probability table of the Flow Distribution node.

(ii) Other Flow nodes

Other model-specific nodes which captured the change in flow included:
the

Cease-to-flow node (the number of days cease-to-flow occurred in a year),

High and Low Flow nodes (the number of days of high and low flow in a year),

Flow Variability (the variability of flow across a defined period of time, including three months, six months or a year, calculated using the coefficient of variation), and

Mean Annual Flow (which was the average daily flow over one year).

Each of these flow nodes was a child node of the primary input nodes Region, Management Scenario and Climate Scenario, so that once again the impact of management alternatives and climate scenarios on changes in flow attributes could be investigated by region, which was deemed a more management orientated approach.

We modelled the data for input into flow nodes separately from the BN, because the impact of management alternatives and climate scenarios on the different flows required external calculation. Then we imported these calculations into a smaller subset BN, which included Region, Management Scenario and Climate Scenario, as well as flow nodes.

We used this small sub-set of nodes from the BN to learn the relationship between these nodes and to calculate conditional probability tables of the flow nodes which we then copied into the final BN models. In the final models, flow nodes were directly linked to the ecological response node of the model.

Thresholds that were used to discretise flow node states were calculated externally (see Section 4) and were dependent on ecological response.

(iii) Water and Air Temperature

Air temperature changes under different climate scenarios were captured in the Mean Maximum and Mean Minimum Daily Temperature nodes. Water temperatures were captured in the Water Temperature node. Data input into temperature nodes included historical temperature records for each region, with an added 1ºC or 2ºC depending on the relevant climate scenario.

We used a smaller BN, created using only the Region, Climate Scenario and Temperature nodes, to learn the relationships between these nodes and calculate the conditional probability tables. The raw data for modelled temperature was imported into this smaller sub-set of nodes from the whole BN model, and the outputs were then used to populate conditional probability tables of the temperature nodes in the final BN models. Temperature nodes were directly linked to the ecological response node of the model. Section 4 outlines the calculation of the thresholds we used to determine temperature node states in the final models.

(iv) Rainfall

Rainfall nodes included Mean Annual Rainfall (calculated from historical records and modelled records for each region) and Rainfall Variability (the coefficient of variation value for rainfall over 365 days). Rainfall nodes were child nodes of both the Region and Climate Scenarios nodes, and were a parent node to the ecological response. We modelled climate scenario impacts on rainfall, separately from the BN, as a time series by region. The raw data for modelled rainfall was then imported into a smaller sub-set of nodes from the whole BN model, including Region, Climate Scenario and Rainfall nodes, to learn the relationship and calculate the conditional probability tables for the rainfall nodes. These outputs were then used to populate conditional probability tables of the relevant nodes in the final BN models. We externally calculated the thresholds that were used to discretise flow node states (see Section 4); they differed for each final BN model in which rainfall nodes occurred.

(v) Water Quality

Each of the seven models contained different water quality nodes, because of the impact of different water quality variables on ecological response. Water quality parameters used in the final models included electrical conductivity (EC), dissolved oxygen (DO), and Water Temperature (see description above). Water quality nodes were child nodes of Flow Distribution; therefore data records for water quality values (1872 in total) were matched to the relevant flow percentiles at the time and location of data collection. Section 4 above outlines the calculation of thresholds used to develop water quality node states.

5.1.5 External Environment nodes

(i) Land Use

Land use and geology was calculated as a percentage of the catchment area that each sampling site was within. Catchment areas were calculated using the watershed function within Arc View 9.3.1 (ESRI 2009) and a 25 m digital elevation model (LPI - NSW Department of Finance and Services, 2006). The proportion of different land uses was calculated using ArcMap GIS Version 9.3. GIS layers and shape files of land use were sourced from Land Use of Australia, Version 4, 2005-06 (ABARES-BRS 2010) (Appendix M).

For the purpose of the BN, we defined four categories of land use:

- Natural (conservation areas, forested and natural environments),
- Agriculture (dominated by grazing and cropping),
- Intensive (Urban and industrial),
- Water.

Only the Intensive and Agriculture land uses were found to be important for invertebrate response; we therefore included them in the final BN models. The land use 'Natural' was included in the fish model.

Each land use node was a child node of Region and a parent node of ecological response, which enabled regional investigation of ecological response to land use. Because land use was calculated as a proportion of total land use, values for the node were between 0 and 1. Thresholds for land use were calculated externally from the BN to form node states (see Section 4).

(ii) Geology

Similar to land use, the type and proportion of different geology within each sampling station catchment area was calculated using ArcMap GIS Version 9.3. Four types of geology were used in final BN models:

- Volcsed,
- Sandstone,
- Granite and
- Other Geology (this was used to describe highly heterogeneous areas with a mixture of geological types) (see Appendix L).

Being a calculated proportion, values for the geological nodes were between 0 and 1. Each Geology node was a child node of Region and directly impacted on the ecological response node. Thresholds for geological node states were model specific and calculated externally (see Section 4).

(iii) Elevation

The Elevation (altitude) node indicated the proportion of sites above or below the threshold of elevation (metres above sea level) in a given region. It was therefore a child node of the Region node. The threshold used in the Elevation node was calculated externally and differed between models. Elevation was a parent node to the ecological response node.

(iv) Habitat

Three habitat nodes, Habscore (habitat score), Shrubs (the proportion of shrubs under 10 m) and Shading of River (%) were used in the final BNs (see Appendix L). The Habscore node included values ranging from 0 to 300, while the Shrubs and Shading of River (%) nodes, as a percentages, included values between 0 and 100. Habitat nodes were child nodes of Region and impacted directly on ecological response. As with other nodes, thresholds used to determine states in habitat nodes were calculated externally.

5.1.6 Ecological response nodes (output)

Each of the six macroinvertebrate BNs had different ecological response nodes. (Section 4 outlines the process of selecting ecological responses.) Models were separated into edge and riffle models, with three models representing each aquatic habitat. General response nodes for four of the models included O/E scores (observed (O) to expected (E) scores in AUSRIVAS) for both edge and riffle models (Figures 43 and 44), and Thermophobic taxa relative abundance for both edge and riffle models (Figures 47 and 48).

For macroinvertebrate community indicator models, relevant variables and corresponding thresholds were assessed for the 42 most common macroinvertebrate species or categories (including invertebrate orders or groups) in edge and riffle communities. The most common macroinvertebrates selected were present in greater than 20% of data records and had a total relative abundance of >200 from 1872 data records.

We used small BN models to determine the five most responsive macroinvertebrate species or categories for both edge and riffle habitats. In the final model (Figures 47 and 48), we selected one species from each of the sensitive Ephemeroptera, Plecoptera and Trichoptera (EPT) orders, as well as one other species susceptible to change (Coleoptera species). The fifth macroinvertebrate selected was deemed a “common” macroinvertebrate that may respond positively to climate change: Oligochaeta (worms) and Chrionomidae (flies).

The fish BN model contained six native fish species: (i) Two-spined Blackfish, (ii) Macquarie Perch, (iii) Mountain Galaxias (iv) Australian Smelt, (v) Golden Perch and (vi) Western Carp Gudgeon. The discretisation of these response nodes was based on the presence and absence of each species.

5.2 Operation of BNs

To explore scenarios and interactions between variables, we needed to select a given state (i.e. condition) for the input nodes Region, Management Scenarios and Climate Scenarios. For example, if a state within the Region node was not selected in the models, results displayed included data from all regions.

Similarly, management scenarios were selected only in regions where management scenarios could be applied. Otherwise, the default “no action taken” was automatically shown for regions where management scenarios could not be applied.

All climate scenarios were possible in all regions, so we needed to select the desired climate scenario; otherwise results reflected a blend of scenarios.

Effluent options needed only to be selected in the region Lower Molonglo, and they were defaulted to “Baseline Scenario” in other regions.

5.2.1 Disadvantages and limitations

While BNs have numerous advantages for modelling ecological systems, there are some limitations that need to be considered. Two of the main disadvantages of BNs are the need to discretise continuous variables, and their inability to support feedback loops (Uusitalo 2007).

Generally the discretisation of continuous variables is undesirable (Pollino et al. 2007) and it may cause a loss in statistical power if the relationship being tested is in fact linear (Myallmaki et al. 2002). While feedback loops were not very relevant our model, we attempted to address the issue of discretisation of continuous variables through the calculation of model-specific thresholds (as outlined in Section 4). In instances where it was difficult to find discrete thresholds the BN was discretised based on the data distribution of the predictor variable of interest.

When modelling ecosystems using a BN, it is impractical to factor in all variables in the model because the BN becomes too unwieldy. Instead, BNs are used to model major system components and links with the most relevant variables. Modellers need to acknowledge the existence of other, albeit more minor, elements that cannot be included, that may impact on results. In an attempt to reduce this limitation, in our models the most influential variables on ecological response were selected through external analysis (see Section 4).

Finally, BN models are also limited by the data used to develop conditional probabilty tables. Where data are patchy or limited, the BN is more capable than other models to cope with such gaps. However, where data are non-existent for combinations within the conditional probabilty tables (for example, there may be no recorded ecological response for a low proportion of shrubs, high proportion of sandstone, high elevation and low flow, etc.), the BN will apply uniform probabilities to such combinations. As such these are distributions are generally uninformative and limit the ability of the model to predict patterns in situations where data is non-existent.
Despite these limitations, BNs are becoming increasingly popular in the environmental and ecological sciences (Howes, Maron & McAlpine 2010). This may be a consequence of the numerous advantages they provide when dealing with high levels of uncertainty and variability which often characterise the data used to build models of ecological systems.

Table 34. Parent node descriptions and applicable models. Nodes have static states (non-threshold dependent)

Parent Node	Description	States	Applicable models
Region	Enabled regionalisation of data and BN findings. See Section 3 for regionalisation methods.	*Ginninderra, *Cooma, Paddy's, Upper Cotter, Lower Cotter, Upper Molonglo, Mid Molonglo, Lower Molonglo, Queanbeyan, Bredbo, Burrinjuck, Gudgenby, Yass, Numeralla, Goodradigbee, Mid Murrumbidgee, Upper Murrumbidgee, *Tuggeranong	All models *Fish model does not include these regions because of data limitations
Climate scenarios	Provided input for climate change scenarios. Included six climate scenario models, three with 1ºC increase and three with 2ºC increase in temperatures (see Section 3 for modelling methods). Historical temperature records were also included for comparison.	Historical Major Change 1 Major Change 2 Moderate Change 1 Moderate Change 2 Minor Change 1 Minor Change 2	All models
Management Scenarios	Provided input for the impact of different flow management adaptation alternatives. Included four management alternatives (see Section 2 for an outline) and a default option of no action taken.	SM1 (C1) SM2 (C2) SM3 (C3) SM4 (C4) No action taken	All models Management alternatives only applicable to regions Upper and Lower Cotter, Upper and Mid Murrumbidgee and Queanbeyan.
Effluent Control	Provided input for the impact of four different effluent treatment options in the Lower Molonglo region (at the LMWQCC) and a default value of baseline. For selection of effluent treatment options see Section 2.	Baseline scenario Total dissolved solids reduction Upgrade LMWQCC Membrane filtration Salt and sewage control	All models except Whole community (riffle) and Fish. Only applicable to the Lower Molonglo region.

Table 35. Intermediate node descriptions, states and applicable models. Appendix L gives more detailed descriptions.

Intermediate Node	Description	States	Applicable models
Flow Distribution	Flows based on percentile flows from the historical time series at each gauging station.	Very high flow High flow Intermediate flow Low flow Very low flow	All models
HF days per year	The number of days of high flow over one year (365 days).	0–10.5, 10.5–30, 30–60, 60–366 0–3, 3–366	AUSRIVAS O/E — Riffle habitat Macroinvertebrate community — Riffle habitat
LF days per year	The number of days of low flow over one year (365 days).	0–25, 25–108, 108–366 0–12, 12–49, 49–366	Thermophobic abundance — Riffle habitat Macroinvertebrate community — Edge habitat
CTF days per year	The number of days of cease-to-flow over one year (365 days).	0–20, 20–43, 43–80, 80–366	AUSRIVAS O/E — Edge habitat
Daily flow variability (1 year)	The variability of daily flow over one year. Calculated using the coefficient of variation over 365 days.	0–2.3, 2.3–4, 4–6, 6–15	Thermophobic abundance — Edge habitat Fish
Mean daily flow (1 year)	The average daily flow calculated over 365 days.	0–249, 249–400, 400–800, 800–8000 0–100, 100–300, 300–500, 500–3000	AUSRIVAS O/E — Riffle habitat AUSRIVAS O/E — Edge habitat
Mean Max daily temp (1 year) ºC	The average maximum daily temperature at the sample site, calculated over 365 days.	Varies between models but all models include a 19 ºC threshold. 2–3 states per model.	All invertebrate models in edge habitat AUSRIVAS O/E — Riffle habitat Fish
Mean Max daily temp (1 month) ºC	The average maximum daily temperature at the sample site, calculated over 30 days.	8–19, 19–21, 21–40	Thermophobic taxa abundance — Riffle habitat
Mean Min daily temp (1 year) ºC	The average minimum daily temperature at the sample site, calculated over 365 days.	0–7, 7–22	AUSRIVAS O/E — Riffle habitat
Daily Rainfall Variability (1 year)	Calculated using the co-efficient of variation for the mean daily rainfall for each sampling site.	0–3, 3–8	Macroinvertebrate community — Riffle habitat Fish
Mean Annual Rainfall (mm)	Calculated using the mean daily rainfall for each sampling site and multiplying by 365 days.	0–965, 965–3500	AUSRIVAS O/E — Riffle habitat Thermophobic taxa abundance — Riffle habitat Fish

Intermediate Node	Description	States	Applicable models
Elevation (m)	The proportion of land at a given elevation above sea level (by region).	Discretisation of states varied between models, and included elevations between 300 and 1400 m (encompassing the whole study area). 2 to 4 states per model.	All models
Water temperature ^{o}C	The temperature of the water taken at the sampling site during macroinvertebrate sampling	0–13, 13–22 0–12, 12–30 0–12.5, 12.5–35	AUSRIVAS O/E — Riffle habitat Macroinvertebrate community — Riffle habitat Thermophobic abundance — Riffle habitat
EC (uS/cm)	The salinity (electrical conductivity) value of the water taken at the sampling site during macroinvertebrate sampling	Discretisation of states varied between models, and included EC values between 0 and 2200. 6 to 7 states per model. Thresholds of 390 and 420 were included to enable the effect of different effluent controls on salinity levels to be shown in the models.	All invertebrate models, except Macroinvertebrate community — Riffle habitat
DO (mg/L)	The amount of dissolved oxygen in the water measured at the sampling site during macroinvertebrate sampling	0–6, 6–7.44, 7.44–9, 9–15	AUSRIVAS O/E — Edge habitat Fish
Agriculture (%)	The proportion of land under agricultural land use, including grazing, cropping and horticulture, in the surrounding sub-catchment above a sampling point	0–2, 2–10, 10–50, 50–70, 70–100	Thermophobic abundance — Edge habitat Fish
Natural (%)	The proportion of land under natural land uses, including conservation reserves and State forests, in the surrounding sub-catchment above a sampling point		Fish
Intensive Land Use (%)	The proportion of land under intensive land uses, including urban and industrial land use, in the surrounding sub-catchment above a sampling point	0–2, 2–7, 7–9, 9–100 0–2, 2–10, 10–50, 50–100	Macroinvertebrate community — Edge habitat AUSRIVAS O/E — Edge habitat

Intermediate Node	Description	States	Applicable models
Sandstone (%)	The percentage of sandstone bedrock in the surrounding sub-catchment above a sampling point	0–25, 25–50, 50–75, 75–100	Thermophobic abundance — Riffle habitat
Volcsed (%)	The percentage of volcsed bedrock in the surrounding sub-catchment above a sampling point	0–3, 3–100	Macroinvertebrate community — Riffle habitat
Granite (%)	The percentage of granite bedrock in the surrounding sub-catchment catchment above a sampling point	0–5, 5–10, 10–50, 50–70, 70–100	AUSRIVAS O/E — Riffle habitat
Other Geology (%)	The percentage of other geological bedrock in the surrounding sub-catchment above a sampling point	0–3, 3–10, 10–40, 40–70, 70–100	AUSRIVAS O/E — Riffle habitat
Habscore	The sum of different habitat scores for stream habitat (see see Appendix L for details of habitat scores)	0–84, 84–300	Thermophobic abundance — Edge habitat
Shading of River (%)	Estimation of the proportion of stream shaded when the sun is directly overhead.	0–3, 3–25, 25–100	Macroinvertebrate community — Edge habitat
% cover of reach by Riffle Periphyton	The percentage of the stream reach in which sampling was taking place covered in periphyton (riffle only)	0–35, 35–100	Macroinvertebrate community — Riffle habitat
% Cover of riparian zone by shrubs	The proportion of land covered by shrubs > 3m within the riparian zone of sampling site	0–23, 23–60, 60–100	AUSRIVAS O/E — Edge habitat

Table 36. Child node descriptions, states and applicable models

Child Node	Description / definition	States	Model
AUSRIVAS O/E score (Riffle)	Observed taxa / expected taxa as predicted by the Upper Murrumbidgee catchment AUSRIVAS model (riffle)	0–0.4, 0.4–0.8, 0.8–1.4	AUSRIVAS O/E — Riffle habitat
AUSRIVAS O/E score (Edge)	Observed taxa / expected taxa as predicted by the upper Murrumbidgee catchment AUSRIVAS model (edge)	0–0.4, 0.4–0.8, 0.8–1.4	AUSRIVAS O/E — Edge habitat
Thermophobic taxa relative abundance (Riffle)	The relative abundance of thermophobic species at the site in riffle habitat	0–1, 1–5, 5–20, 20–100	Thermophobic taxa relative abundance — Riffle habitat
Thermophobic taxa relative abundance (Edge)	The relative abundance of thermophobic species at the site in edge habitat	0–1, 1–5, 5–20, 20–100	Thermophobic taxa relative abundance — Edge habitat
Leptophlebiidae (E) (%)	The relative abundance of Leptophlebiidae at the sampling site	0–2, 2–10, 10–100	Macroinvertebrate community — Riffle and Edge habitat models
Gripopterygidae (P) (%)	The relative abundance of Gripopterygidae at the sampling site	0–2, 2–10, 10–100	Macroinvertebrate community — Riffle and Edge habitat models
Leptoceridae (T) (%)	The relative abundance of Leptoceridae at the sampling site	0–2, 2–10, 10–100	Macroinvertebrate community — Edge habitat
Conoesucidae (T) (%)	The relative abundance of Conoesucidae at the sampling site	0–2, 2–10, 10–100	Macroinvertebrate community — Riffle habitat
Elmidae (Coleoptera) (%)	The relative abundance of Elmidae at the sampling site	0–2, 2–10, 10–100	Macroinvertebrate community — Edge habitat
Scirtidae (Coleoptera) (%)	The relative abundance of Scirtidae at the sampling site	0–2, 2–10, 10–100	Macroinvertebrate community — Riffle habitat
Oligochaeta (worms) (%)	The relative abundance of Oligochaeta at the sampling site	0–2, 2–10, 10–100	Macroinvertebrate community — Edge habitat
Chironomidae (Diptera) (%)	The relative abundance of Chironomidae at the sampling site	0–2, 2–10, 10–100	Macroinvertebrate community — Riffle habitat
Macquarie Perch	The presence or absence of Macquarie Perch at the sampling site	Present / Absent	Fish
Trout Cod	The presence or absence of Trout Cod at the sampling site	Present / Absent	Fish
Two-spined Blackfish	The presence or absence of Two-spinned Blackfish at the sampling site	Present / Absent	Fish
Australian Smelt	The presence or absence of Australian Smelt at the sampling site	Present / Absent	Fish
Mountain Galaxias	The presence or absence of Mountain galaxias at the sampling site	Present / Absent	Fish
Western Carp Gudgeon	The presence or absence of Western carp gudgeon at the sampling site	Present / Absent	Fish
Golden Perch	The presence or absence of Golden perch at the sampling site	Present / Absent	Fish
Fish species richness	Number of native fish species at the sampling site	1, 2, 3, 4, 5	Fish

6. SCENARIO ASSESSMENT

This section describes the use of the Bayesian Networks (BNs) to inform adaptation initiatives. It describes a series of 'story lines' that demonstrate the use of the models to explore adaptation initiatives. The consequences of adaptation initiatives for future water security and waste water management for water quality and ecological response are also covered. Priorities for adaptation initiatives based on probabilities of adverse effects are also discussed.

6.1 Scenario Assessment: Story Line Results

In the following sections we describe the consequences for the selected ecological end points, for the various management alternatives and/or climate change scenarios, in five regions within the Upper Murrumbidgee catchment. We selected these regions to illustrate the range of management adaptation alternatives:

- Goodradigbee region — primarily conservation with little/no management;
- Upper Cotter region — river regulation;
- Upper Murrumbidgee region — river regulation;
- Lower Molonglo region — salt discharge from water quality treatment centre;
- Yass region — agriculture and salinity.

We show the results for the climate scenarios associated with a 2^{o}C increase in temperature ('Minor change 2', 'Moderate change 2' and 'Major change 2') only, because the results for the 1^{o}C increase in temperature mirror the 2^{o}C changes, but are less marked.

In each region, with the exception of Lower Molonglo, the responses of the O/E score, the Thermophobic taxa relative abundance and the relative abundance of selected macroinvertebrates were investigated for both edge and riffle habitats. In the Molongolo region, we only investigated the response of the O/E score, which is a commonly used indicator of river condition in that region. In addition, we investigated Two-spined Blackfish in the Upper Cotter, because this species displayed a relatively significant response to climate change and is of management interest for fish conservation in the Upper Cotter region.

6.1.1 Goodradigbee Region

The Goodradigbee River catchment is located in the Brindabella mountain range on the eastern side of the Upper Murrumbidgee catchment (see Section 4, Figure 29). Most (95%) of the 1101 km^2 catchment is forested, and the upper parts of the catchment located within Kosciuszko National Park. The remainder of the catchment is used for grazing purposes. The Goodradigbee River (105 km long) is considered to be unregulated, although there is a small diversion on the upper reaches of the river diverting water into Tantangara Dam (Lintermans 2002). In the Goodradigbee region, no management options were relevant, so all results from the BN are for the baseline management strategy — 'no action taken' (Figure 50).

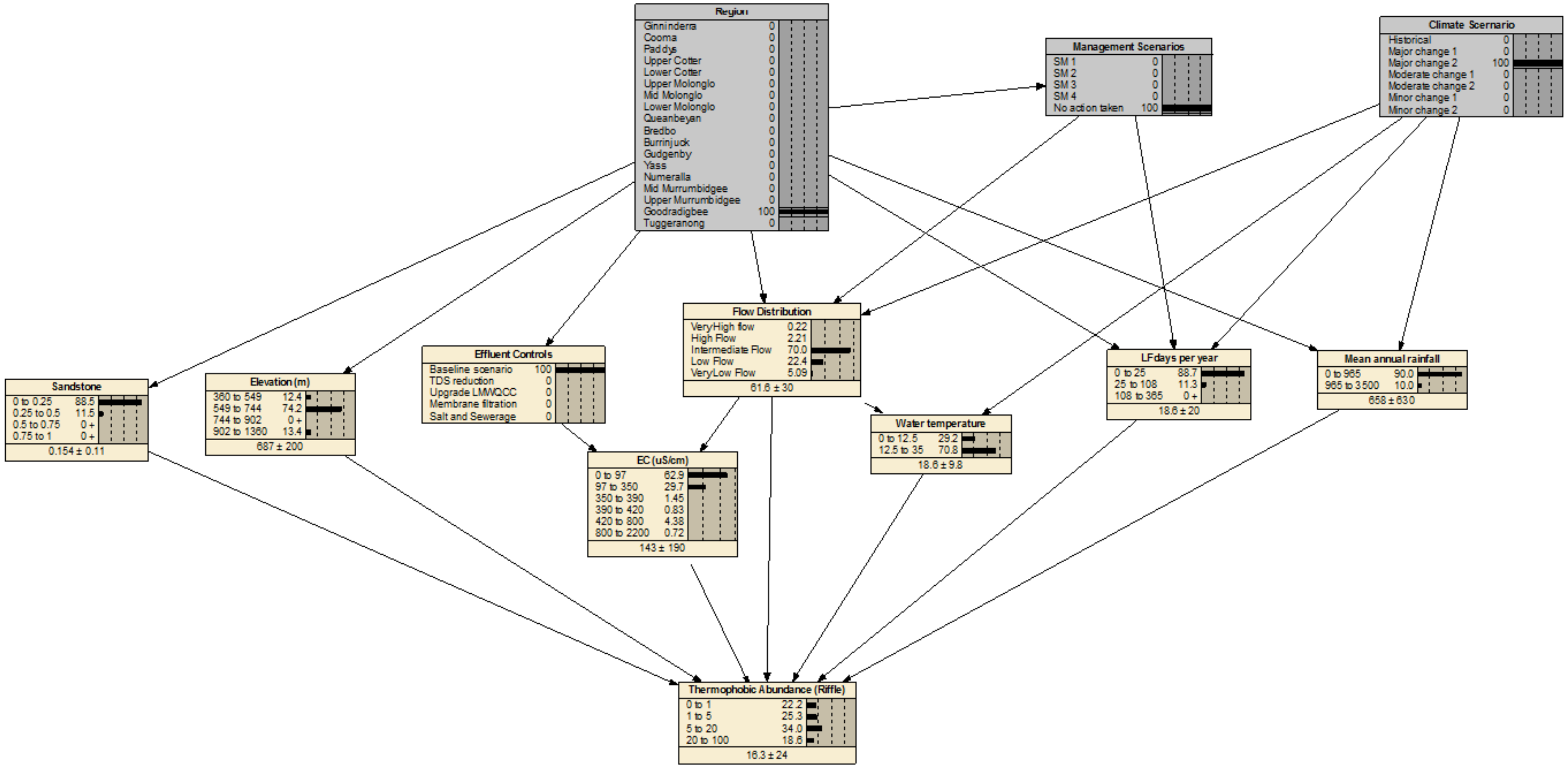


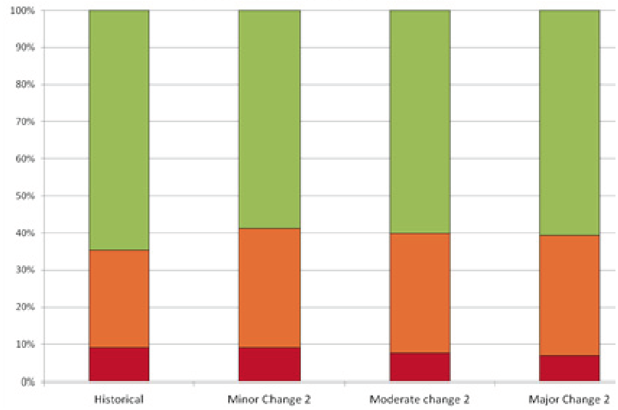
Figure 50. Example BN for O/E score in edge sites under the management scenario of no action taken, and the Major change 2 Climate scenario in the Goodradigbee region

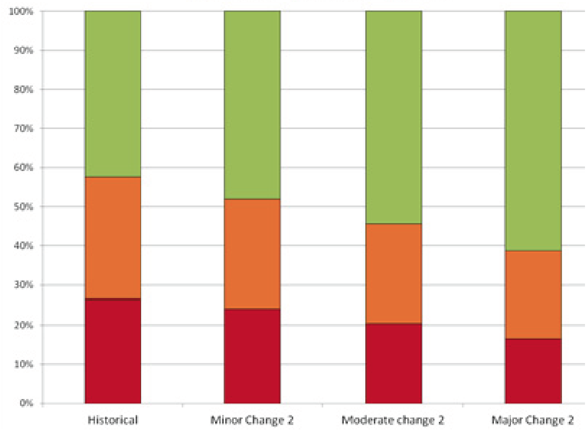
(i) O/E scores

In the Goodradigbee region, the probability of O/E scores being in a particular state (i.e. condition) showed little change (only 5% for the 0.8–1.4 O/E score category) in edge habitats between different climate scenarios (Figure 51). In riffle habitats the models predicted an increasing likelihood of O/E scores being in the highest category (0.8–1.4) as climate change impacts increase (i.e. moving from historical to Minor, Moderate and Major climate change scenarios) (Figure 52). There was an almost 20% increase in the likelihood of O/E scores being in the highest state under the Major change 2 climate scenario compared to Historical conditions (Figure 52).

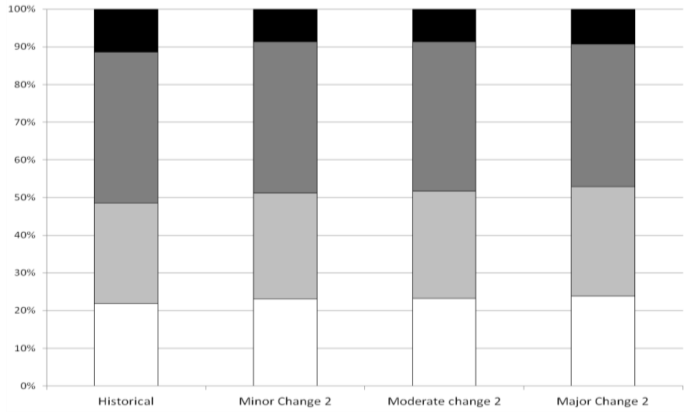
(ii) Thermophobic taxa relative abundance

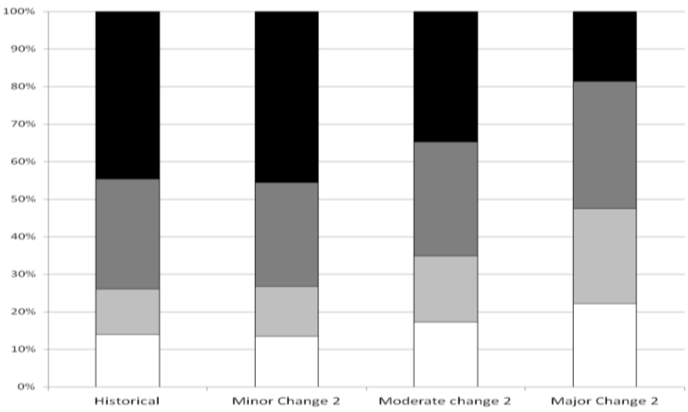
As with O/E scores, there was little change in the relative abundance of Thermophobic taxa under different climate change scenarios in the edge habitats (Figure 53). In riffle habitats, the likelihood of the Thermophobic taxa relative abundance being high (in the 20–100 state) was greatest under the Historical and Minor change 2 scenarios and least under the Major change 2 scenario (Figure 54). Relative to the Historical scenario, under the Major change 2 scenario Thermophobic taxa were 25% less likely to be highly abundant (Figure 54).





Figures 51 & 52. Predicted probabilities of O/E scores in edge and riffle habitats in the Goodradigbee region, for four climate change scenarios (l–r): Historical climate conditions, Minor change 2, Moderate change 2, Major change 2. Edge = top graph; Riffle = lower graph. Probabilities (y axis): 0% to 100%. Categories: low scores: 0–0.4, red; moderate scores: 0.4–0.8, orange; high scores: 0.8–1.4, green.





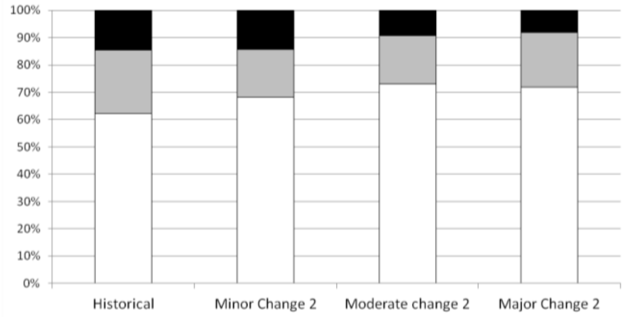
Figures 53 & 54. Predicted probabilities of Thermophobic taxa relative abundance in edge and riffle habitats in the Goodradigbee region, for four climate change scenarios (l–r): Historical climate conditions, Minor change 2, Moderate change 2, Major change 2. Edge = top graph; Riffle = lower graph. Probabilities (y axis): 0% to 100%. Categories: 0–1, white; 1–5, pale grey; 5–20, dark grey; 20–100, black.

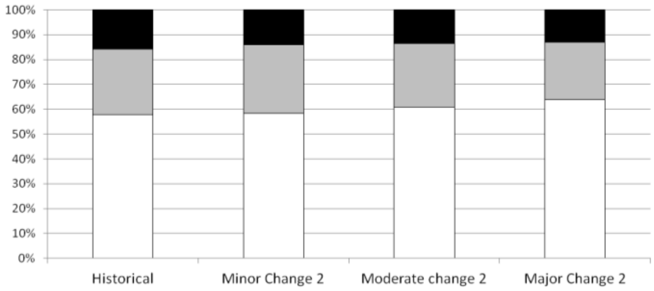
(iii) Macroinvertebrate community indicators

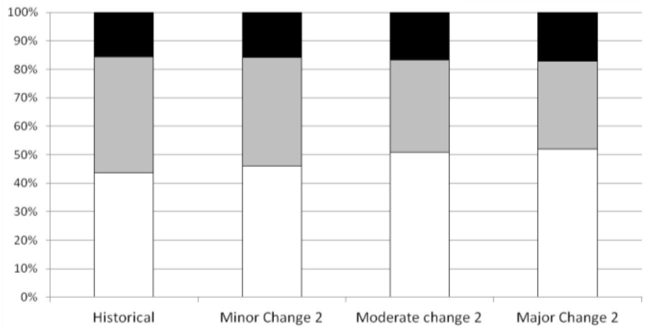
In edge habitats, the probability of representative macroinvertebrate community taxa being in a particular state changed minimally (<10%) under the different climate change scenarios (Figures 55–59). However, in riffle sites some taxa showed more pronounced changes (Figures 60–44). In riffle sites Leptophlebiidae and Gripopterygidae were 10% less likely to be highly abundant (present in numbers of 10–100) under the Major change 2 climate change scenario than the Historical scenario (Figure 60 & Figure 64), while under Moderate change 2, Chironomidae was 20% less likely to be highly abundant (Figure 64).

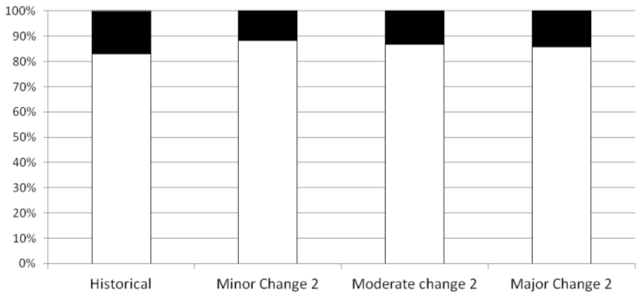
6.1.2 Goodradigbee region conclusions

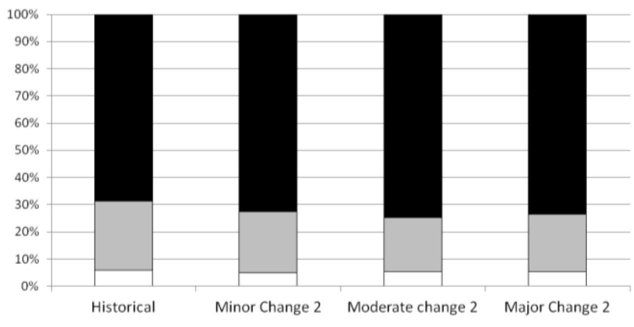
In the Goodradigbee region the results suggest that climate change may cause a slight decline in the Thermophobic taxa relative abundance and some taxa representative of the macroinvertebrate community (e.g. Leptophlebiidae) (as explained in Section 4). In contrast, predicted changes in the O/E scores suggest that the condition, as defined by the macroinvertebrate community, may improve slightly under climate change.



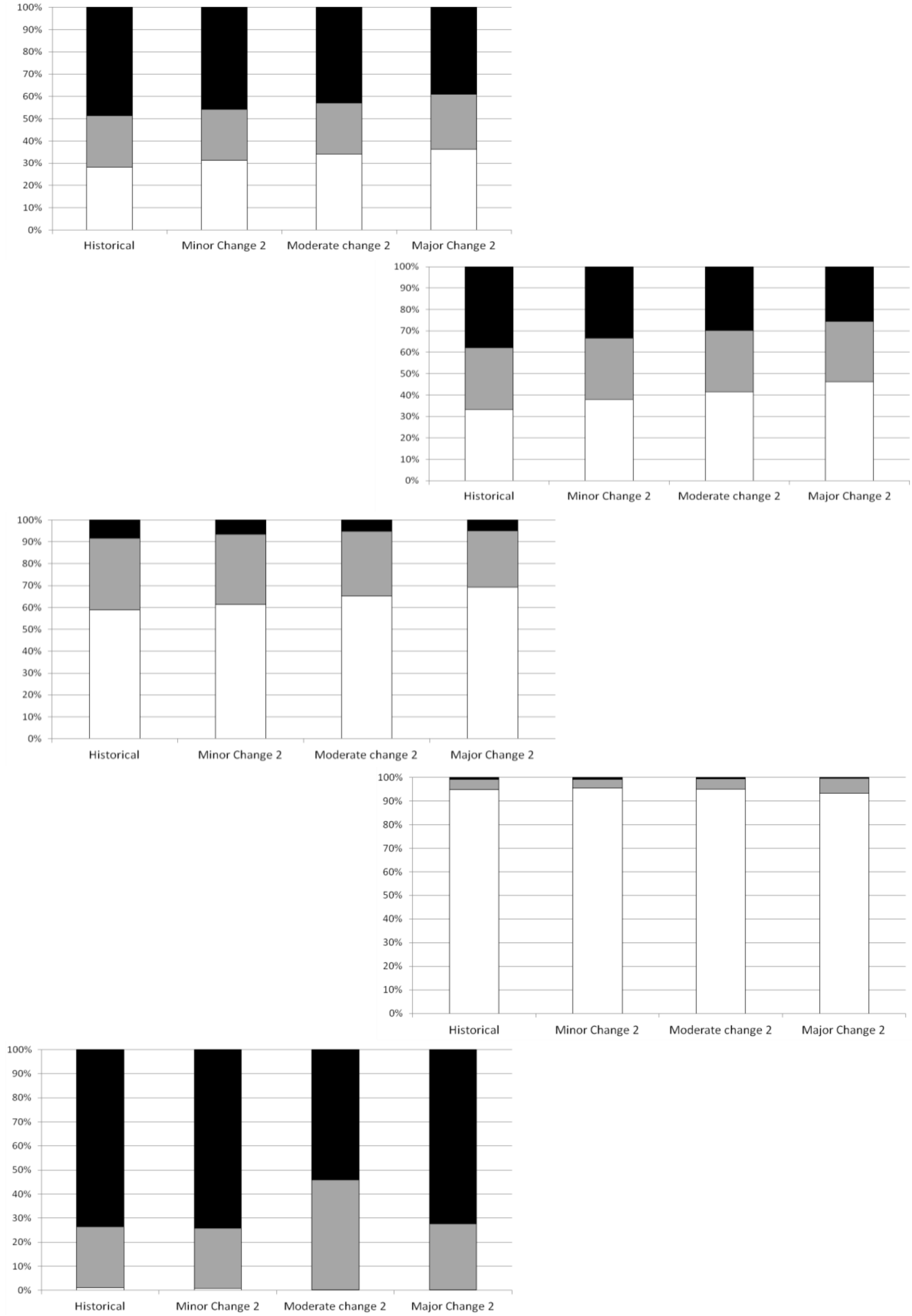








Figures 55 (top)–59. Predicted probabilities of the relative abundance in edge habits in the Goodradigbee region, of Leptophlebiidae (top), Gripopterygidae, Leptoceridae, Elmidae, Oligochaeta (bottom), for four climate change scenarios (l–r): Historical climate conditions, Minor change 2, Moderate change 2, Major change 2. Probabilities (y axis): 0% to 100%. States: 0–2, white; 2–10, grey; 10–100, black.



Figures 60 (top)–64. Predicted probabilities of the relative abundance in riffle habitats in the Goodradigbee region, of Leptophlebiidae (top), Gripopterygidae, Conoesucidae, Scirtidae, Chironomidae (bottom), for four climate change scenarios (l–r): Historical climate conditions, Minor change 2, Moderate change 2, Major change 2. Probabilities (y axis): 0% to 100%. States: 0–2, white; 2–10, grey; 10–100, black.

These results appear contradictory and may be an artefact of the model for this region. The Goodradigbee region has been minimally affected by human alterations and is generally considered to be in "reference" condition and the distribution of O/E scores are highly skewed. We have generally found weak relationships between O/E scores and the drivers of macroinvertebrate communities, particularly those that are climate related. These weak relationships may mean that our capacity to predict changes is limited with these models, and contradictory results are produced.

Nonetheless, thermophobic and particular representative taxa did show a negative response to the declines in water availability predicted under the climate change scenarios tested. This suggests that even though Goodradigbee is a reference region minimally impacted by human activities some taxa may be vulnerable to climatic changes in the future.

6.1.3 Upper Cotter Region

The Cotter River is located on the western side of the Upper Murrumbidgee catchment and is regulated by three dams that supply water for Canberra. The Upper Cotter River comprises the area upstream of Cotter Dam (approximately 480 km^2) and is covered with native forest (see Section 2 Figure 4 or Section 4 Figure 27). For the Upper Cotter region, management alternatives SM1 to SM4 (see Section 2) were relevant, so all results from the BN are given in terms of each management alternative (Figure 65).

(i) O/E scores

No large differences were predicted for O/E scores in any of the management alternatives or climate scenarios for the edge and riffle habitats (Figures 66, 67). Only a slight decrease (~5%) in the probability of high O/E scores (0.8–1.4) occurring was predicted for the edge habitat in all climate scenarios (Figure 66).

(ii) Thermophobic taxa relative abundance

No large differences were predicted for Thermophobic taxa relative abundance in any of the management alternatives or climate scenarios (Figures 68, 69). In edge habitat a slight decrease (~4%) in the probability of thermophobic taxa being in the highest abundance category was detected across climate scenarios (Figure 68), but for the riffle habitat the opposite trend was observed (Figure 69).

(iii) Macroinvertebrate community indicators

The models predict that climate change will decrease the probability that Leptophlebiidae are highly abundant in both edge and riffle habitats (Figures 70, 75). This response was more prominent under Major change 2 for the edge habitat (Figure 70) and under Minor change 2 for the riffle habitat (Figure 75). In all cases, management alternative SM3 (increased water demand) reinforced this response (Figures 70, 75). Leptophlebiidae is a family which belongs to the Ephemeroptera order and together with the Plecoptera and Trichoptera order they make up the EPT metric. EPT is widely used as an indicator of river health; therefore it is expected that severe climate change will cause a decrease in the abundance of these orders. An increase in water demand (i.e. SM3) will amplify this response.

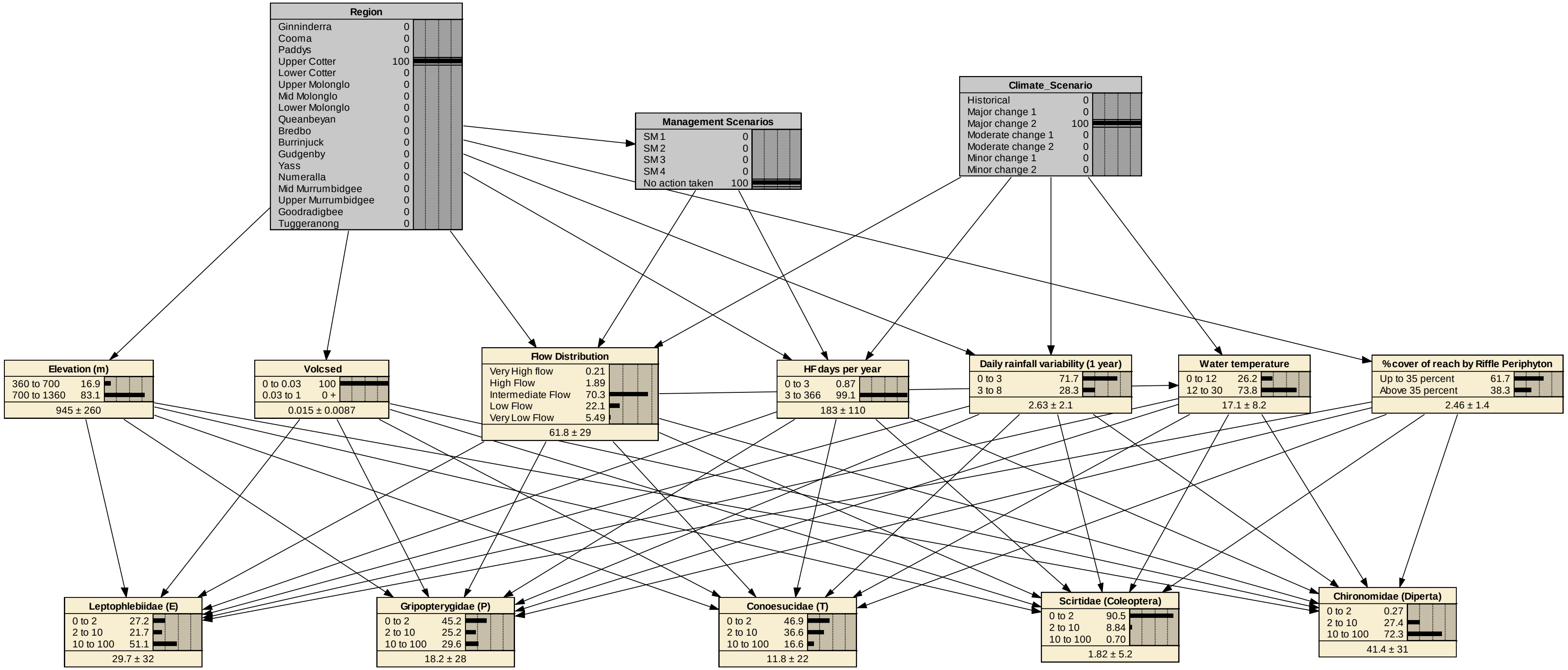
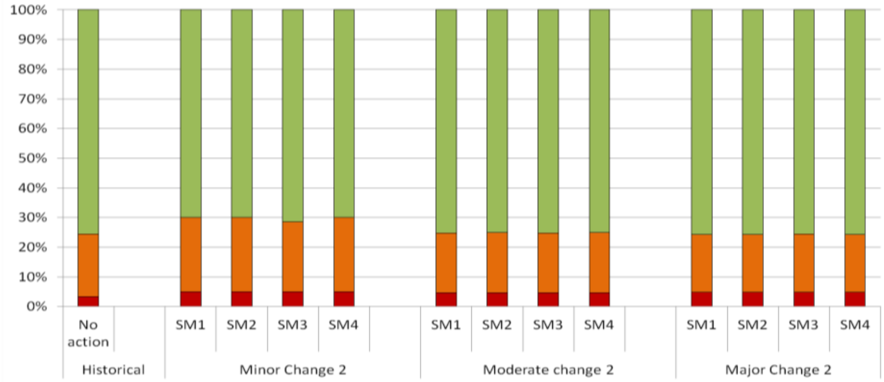
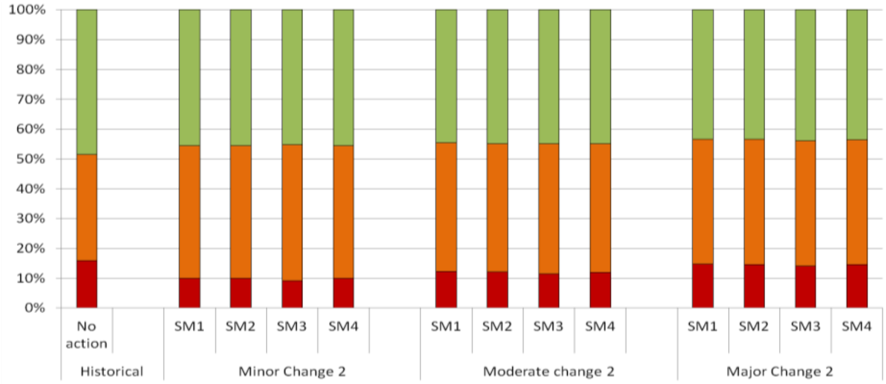
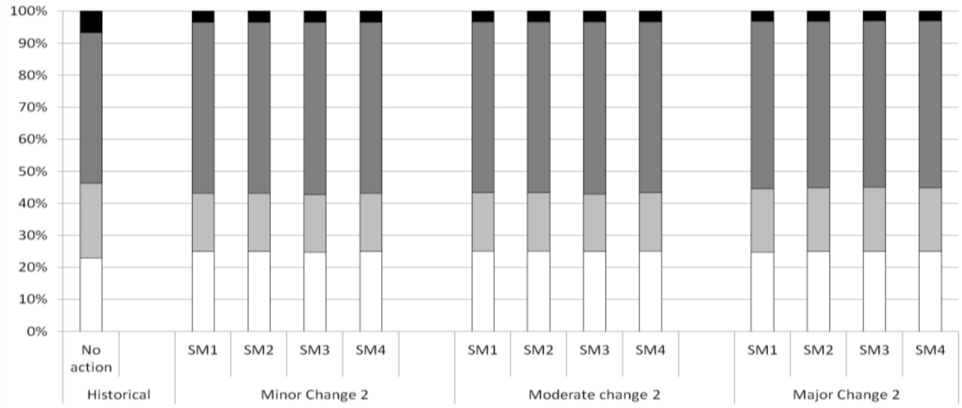


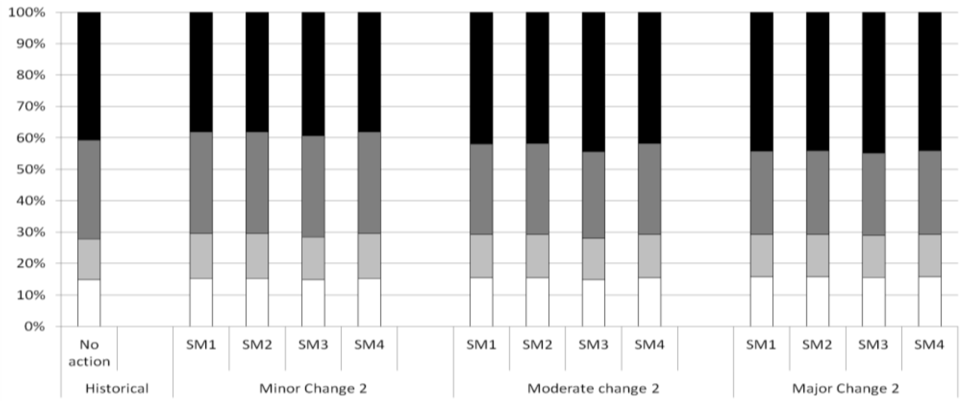
Figure 65. Example BN for macroinvertebrate representative taxa in Riffle sites under the Major change 2 Climate scenario in the Upper Cotter region



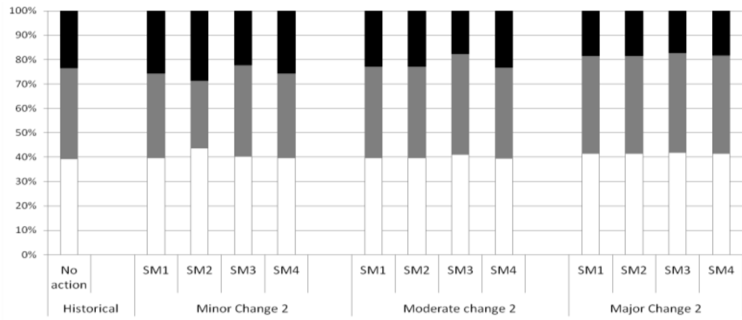


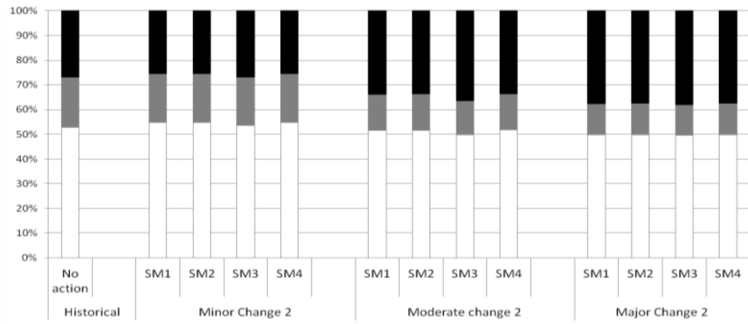
Figures 66 & 67. Predicted probabilities of O/E scores in edge and riffle habitats in the Upper Cotter region, for 4 climate change scenarios and 4 management adaptation alternatives (l–r): Historical climate conditions + current management practices (no action); SM1–SM4 in each of Minor change 2, Moderate change 2 and Major change 2. Edge = top graph; Riffle = lower graph. Probability (y axis): 0–100%. Score categories: low: 0–0.4, red; moderate: 0.4–0.8, orange; high: 0.8–1.4, green.

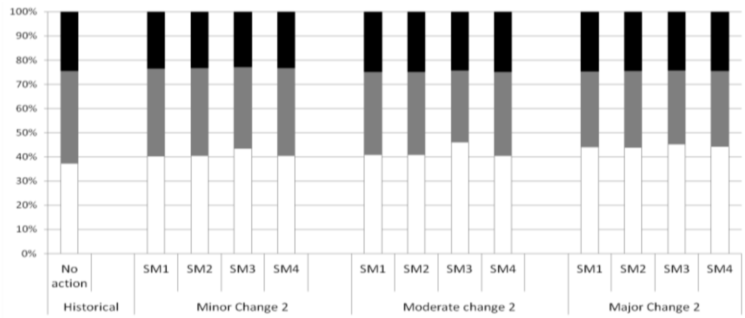


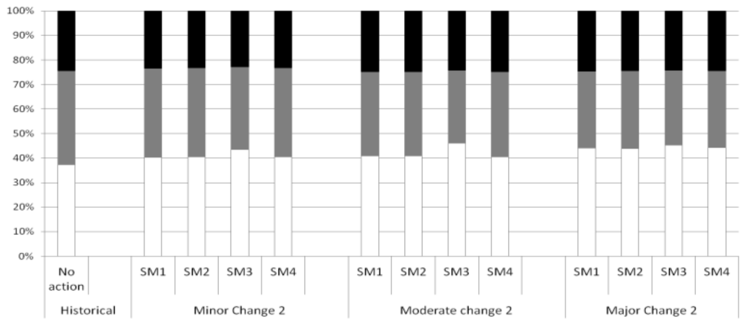


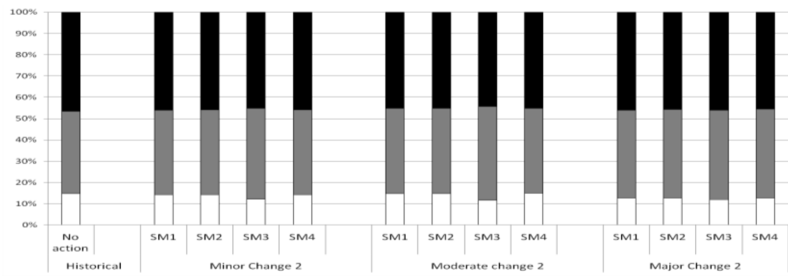
Figures 68 & 69. Predicted probabilities of Thermophobic taxa relative abundance in edge and riffle habitats in the Upper Cotter region, for 4 climate change scenarios and 4 management adaptation alternatives (l–r): Historical climate conditions + current management practices (no action); SM1–SM4 in each of Minor change 2, Moderate change 2 and Major change 2. Edge = top graph; Riffle = lower graph. Probability (y axis): 0% to 100%. Categories: 0–1, white; 1–5, pale grey; 5–20, dark grey; 20–100, black.



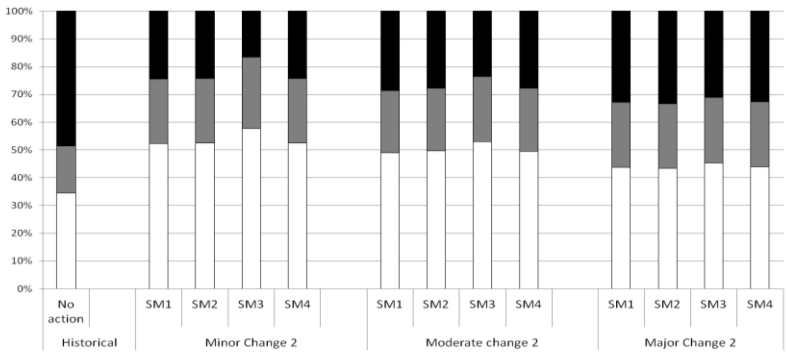


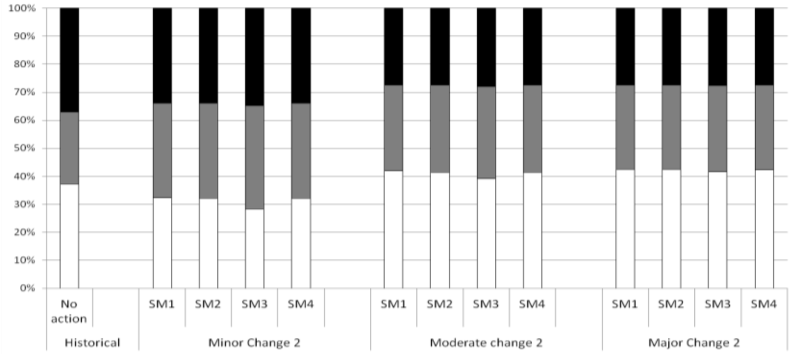


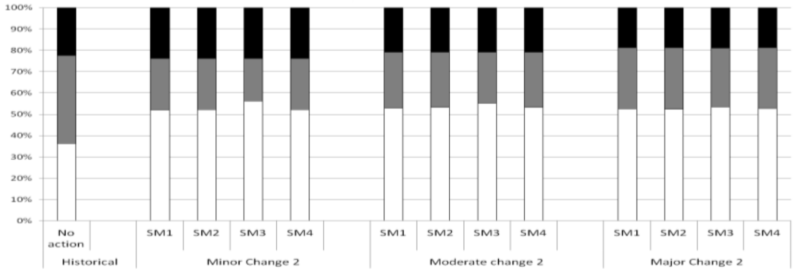


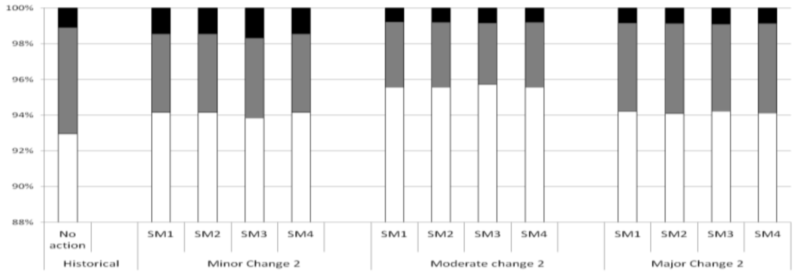


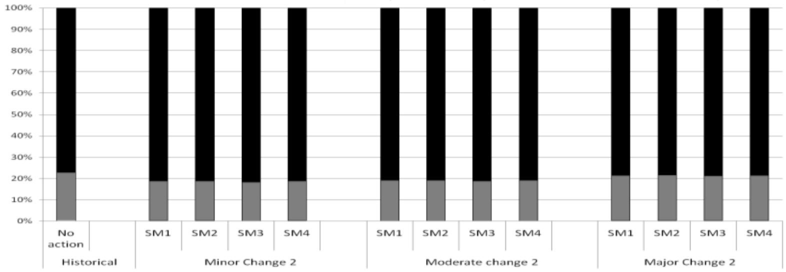
Figures 70 (top)–74. Predicted probabilities of the relative abundance in edge habits in the Upper Cotter region, of Leptophlebiidae (top), Gripopterygidae, Leptoceridae, Elmidae, Oligochaeta (bottom), for 4 climate change scenarios and 4 management adaptation alternatives (l–r): Historical climate conditions + current management practices (no action); SM1–SM4 in each of Minor change 2, Moderate change 2 and Major change 2. Probability (y axis): 0% to 100%. States: 0–2, white; 2–10, grey; 10–100, black.











5 figures, one page, vertical.

Figures 75 (top)–79. Predicted probabilities of the relative abundance in riffle habitats in the Upper Cotter region, of Leptophlebiidae (top), Gripopterygidae, Conoesucidae, Scirtidae, Chironomidae (bottom), for 4 climate change scenarios and 4 management adaptation alternatives (l–r): Historical climate conditions + current management practices (no action); SM1–SM4 in each of Minor change 2, Moderate change 2 and Major change 2. Probability (y axis): 0% to 100%. States: 0–2, white; 2–10, grey; 10–100, black.

A similar response to Leptophlebiidae was predicted for Leptoceridae (Trichoptera), Conoesucidae (Trichoptera) and Gripopterygidae (Plecoptera). A slight increase in the probability of Leptoceridae being present in low numbers (the lowest relative abundance category) was observed with climate change. This response was again more prominent under Major change 2 (~8%) and slightly reinforced by management alternative 3 (SM3, Figure 72).

A similar response was observed for Conoesucidae (Figure 77). Climate change produced an increase in the probability of Conoesucidae being present in low numbers (Figure 77). This response was more prominent under Major change 2 (~16 %) (Figure 77). In all cases, management alternative SM3 slightly reinforced this response (Figure 77).

Finally, Gripopterygidae (Plecoptera) showed the same response as other EPT families, but only for riffle sites. In the riffle, the probability of Gripopterygidae being highly abundant decreased ~10% under the Major change 2 (Figure 76). However, in edge sites the probability of Gripopterygidae being highly abundant increased with climate change (Figure 71).

Decreases in the abundances of the families Elmidae and Scirtidae (Coleoptera) were also observed. This response was strongest in the case of Elmidae with the probability of being highly abundant decreasing under Major change 2 by ~10% (Figure 73). Meanwhile for Scirtidae the decrease was very small (~3%) (Figure 78). This response was slightly reinforced by an increase in the water demand (management alternative SM3) in the case of the family Scirtidae. Although Coleoptera is not part of the EPT metric, some families of the order Coleoptera such as Elmidae and Scirtidae are also identified as sensitive. Therefore, a decrease in their abundances, as predicted by our models, might also be expected as a result climate change.

By contrast, taxa traditionally identified as tolerant are expected to either increase with climate change, or at least be minimally impacted. Our predictions are consistent with this expectation, with tolerant families Chironomidae and Oligochaeta predicted to change little between climate or management scenarios (Figures 74, 79).

(iv) Two-spined blackfish

A decrease in the probability of Two-spined Blackfish being present across climate scenarios was predicted (Figure 80). This decrease was most prominent for the severest climate scenario (Major change 2) and slightly exacerbated by an increase in water demand under the SM3 management scenario. Two-spined Blackfish is a flow sensitive species and particularly sensitive to changes in flow variability. The scenarios tested are likely to change flow variability, which may explain, in part, the predicted negative impact of climate change on the species. However, future research is needed to further investigate this issue.

6.1.4 Upper Cotter region conclusions

In the Upper Cotter region the results suggested that climate change may cause a slight decrease in the probability of achieving the highest categories of O/E scores (i.e. worsening river condition). In addition, most of the sensitive macroinvertebrate taxa (EPT and coleopterans) may be negatively impacted (except for Gripopterygidae in riffle sites).On the other hand, tolerant taxa (Chironomidae and Oligochaeta) are unlikely to be affected. Thermophobic taxa relative abundance appeared to be unaffected by climate

change. The presence of the vulnerable native fish species, the Two-spined Blackfish, is expected to decrease.

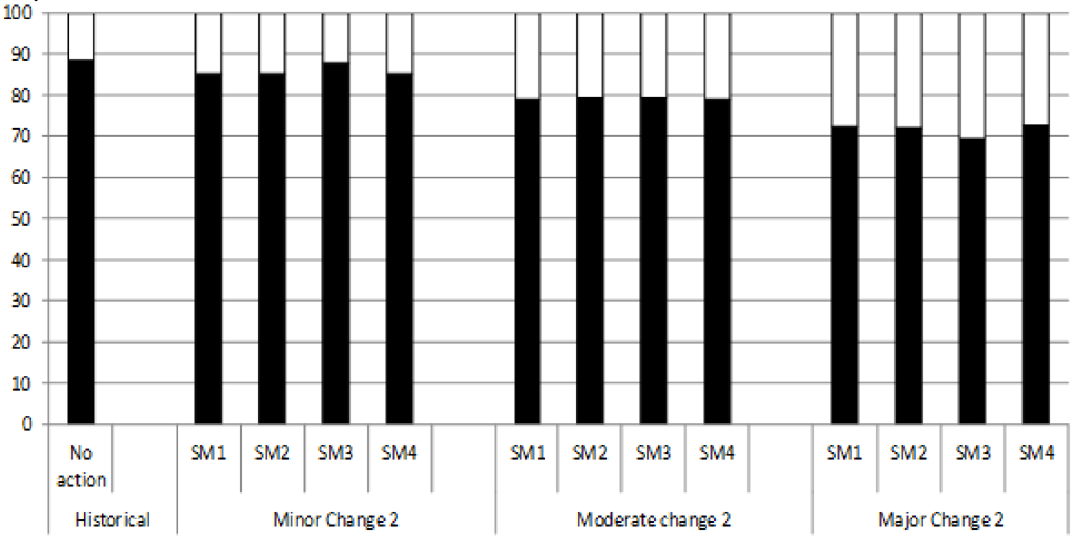


Figure 80. Predicted probability of Two-spined Blackfish presence/absence in the Upper Cotter region, for 4 climate change scenarios and 4 management adaptation alternatives (l–r): Historical climate conditions + current management practices (no action); SM1–SM4 in each of Minor change 2, Moderate change 2 and Major change 2. Probability (y axis): 0% to 100%. Present, black; absent, white.

Key macroinvertebrate taxa are predicted to be adversely affected by climate change, and in some cases this is reinforced by an increase in the demand for water. Disappearance of these key taxa could have important implications for the way in which the aquatic ecosystems function. Macroinvertebrates are commonly used as measures of river health because of their important role in food webs and their sensitivity to disturbances. Consequently, a loss of macroinvertebrate diversity can seriously affect river ecosystem functioning. Generally, in this study, the effect of climate change on individual taxa was more evident than the effect of climate change on aggregate community indicators (O/E scores or Thermophobic taxa relative abundance). This suggests that studying individual taxa that are key elements in the ecosystem may be more useful than studying aggregate metrics for detecting climate change impacts on aquatic ecosystems.

Within the Upper Cotter there also appeared to be little or no effect of the management alternatives on the ecological responses tested. Only an increase in water demand under management alternative SM3 resulted in an amplification of the negative effect of climate change. These results are not a surprise, because the Upper Cotter has been subject to environmental flows for many years and in the applied management alternatives we tested here there was little change to these conditions.

6.1.5 Upper Murrumbidgee Region

The Upper Murrumbidgee region forms the headwaters of the broader study area (the Upper Murrumbidgee River catchment as a whole) and encompasses the area upstream of Tantangara Dam through to the confluence with the Numeralla River (see maps at Figures 4 in Sections 2). The area upstream of Tantangara Dam is forested, and most of it is within Kosciuszko National Park. Downstream of Tantangara, the land use is a mix of forest (including national park and nature reserves), cropping and grazing. Tantangara Dam is a major regulating structure on the Murrumbidgee River, diverting in excess of 99% of the flows from the Murrumbidgee River into the Snowy Hydro scheme. For the Upper Murrumbidgee region, management alternatives SM1 – SM4 were relevant, so here we give the results from the BN for each option (Figure 81).

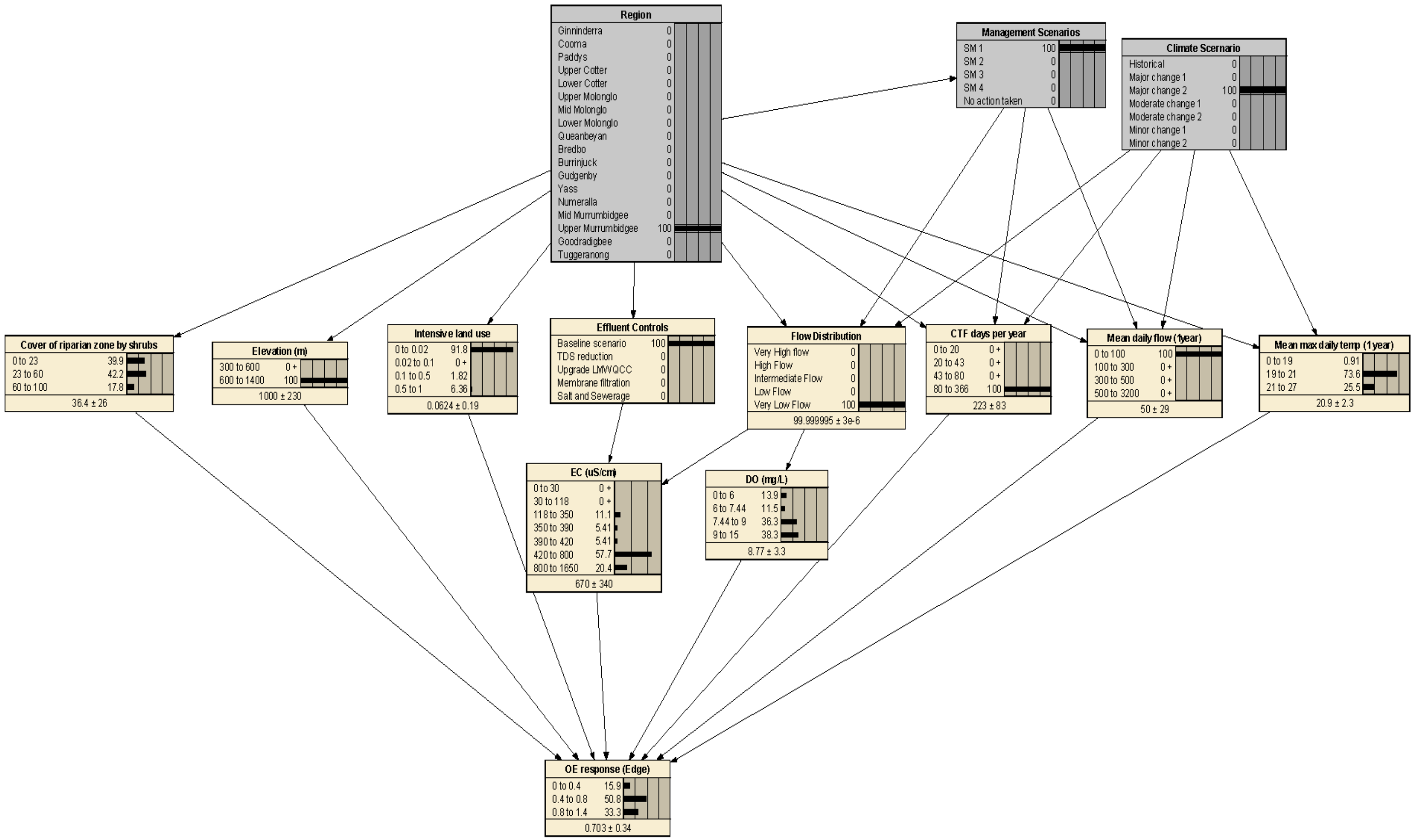


Figure 81. Example BN for O/E score in edge habitat under Management scenario SM1 and the Major change 2 Climate scenario in the Upper Murrumbidgee region.

(i) O/E scores

Regardless of the climate change severity, the probability of O/E scores being high was much lower for the edge habitat under management alternative SM1 (no water being released from Tantangara Dam), and when only purchased ACT water was released from Tantangara Dam (management alternative SM3, Figure 82). For the riffle habitat when no water was being released from Tantangara Dam (management alternative SM1) there was a slight decrease (1–2%) in the probability of getting a high O/E score across the three climate scenarios (management alternative SM1, Figure 83).

(ii) Thermophobic taxa relative abundance

High Thermophobic taxa relative abundance (20–100%) for the edge habitat decreased under each climate scenario compared to historical conditions (Figure 84). However, regardless of climate scenario under management alternatives SM1 (no water released) and SM3 (minimal water released), there was an increased likelihood of thermophobic taxa being absent (0–1% relative abundance category, Figure 84).

Similarly for the riffle habitat, regardless of climate change and when no water was being released from Tantangara Dam (management alternative SM1), high Thermophobic taxa relative abundance (20–100%) was less likely, compared to the probability under other management alternatives (Figure 85). There was also an increased absence of thermophobic taxa (0–1% relative abundance category) compared to other scenarios (Figure 85).

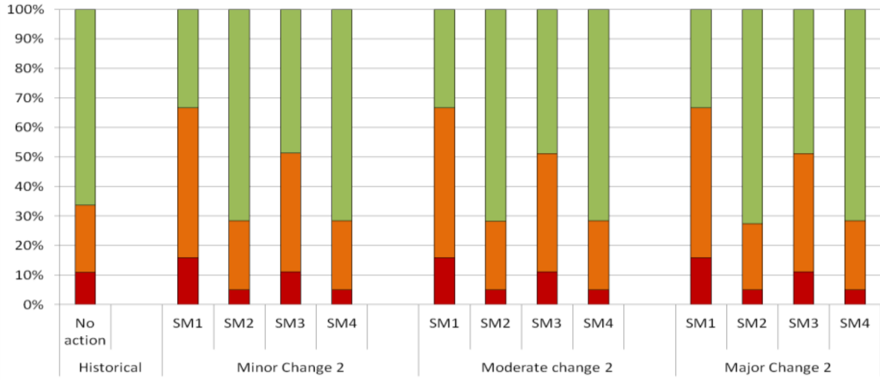
(iii) Macroinvertebrate community indicators

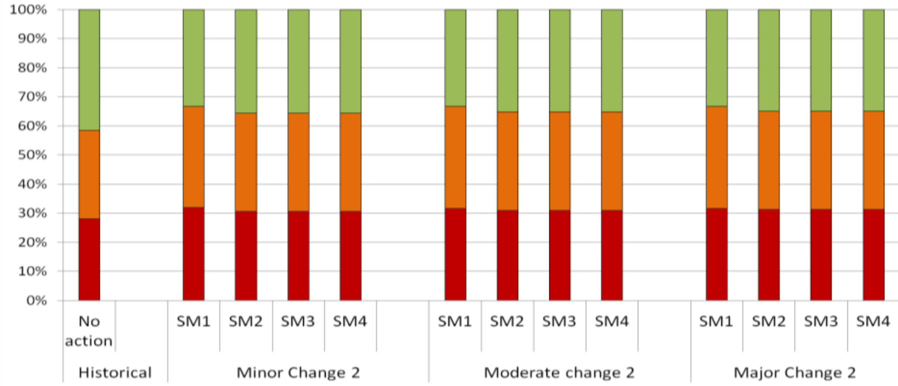
For the edge habitat the relative abundances of Leptophlebiidae, Gripopterygidae, and Leptoceridae were dependent upon water being released from Tantangara Dam (Figures 86–88). Regardless of the climate scenario, when no water was being released from Tantangara Dam (management alternative SM1), absences (0–1% category) of Leptophlebiidae, Gripopterygidae and Leptoceridae increased (Figures 86–88). Conversely, the relative abundance of tolerant Oligochaeta increased regardless of climate scenario when no water was being released from Tantangara Dam under scenario SM1 (Figure 90).

Elmidae was the only taxon in the edge habitat model not to show a response to management; it possibly showed a small response to climate change (Figure 89). Across each climate change scenario there was a small increase in absences of Elmidae (0–1% category) (Figure 89). However, it should be noted that this relationship may have been limited by low Elmidae abundance in the historical data set (Figure 89).

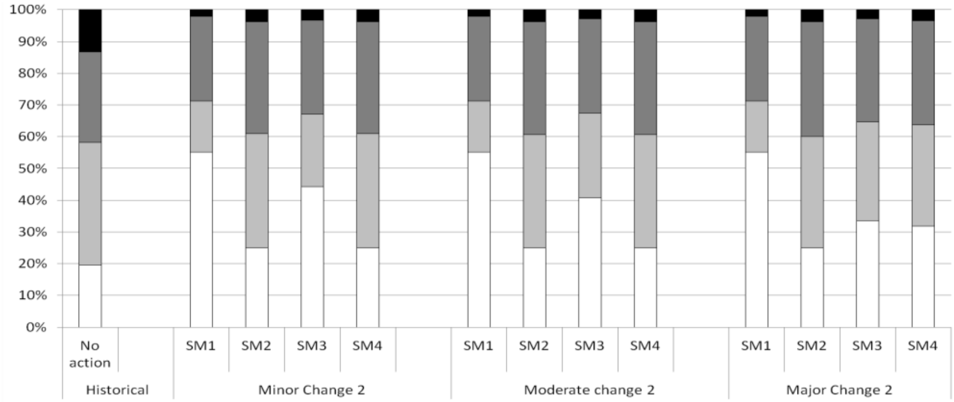
Macroinvertebrate community results for the riffle habitat also showed a minimal response to climate change (Figures 91–95). Across all climate change scenarios, the relative abundance of Scirtidae increased regardless of the management alternatives applied (Figure 94).

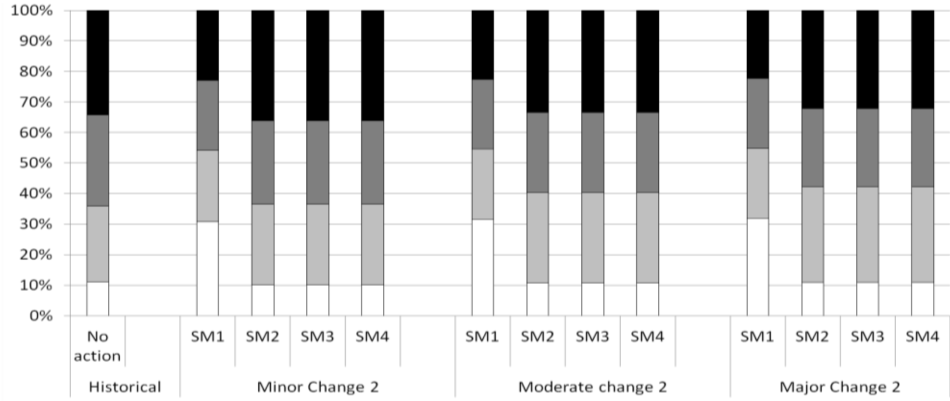
Conoesucidae was the only taxon to show a negative response to no water being released from Tantangara Dam under management alternative SM1 (Figure 93). The relative abundances of Leptophlebiidae, Gripopterygidae and Chironomidae unexpectedly all increased under management alternative SM1 (Figures 91, 92, 95). Further investigation is needed, to determine the influencing factors behind the relationship for the relative abundance of each taxon with management alternative SM1.



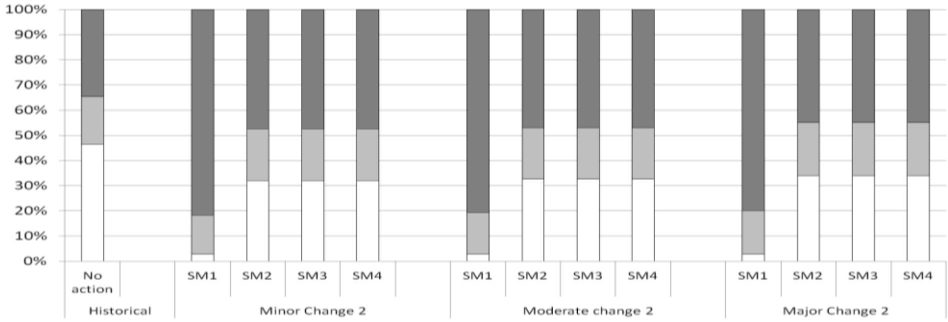


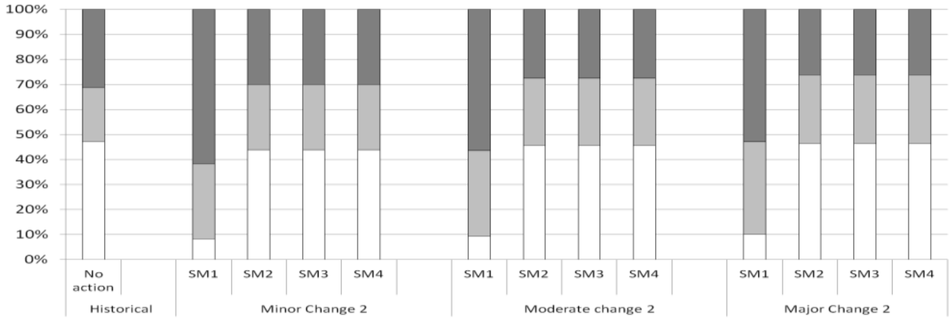
Figures 82 & 83. Predicted probabilities of O/E scores in edge and riffle habitats in the Upper Murrumbidgee region, for 4 climate change scenarios and 4 management adaptation alternatives (l–r): Historical climate conditions + current management practices (no action); SM1–SM4 in each of Minor change 2, Moderate change 2 and Major change 2. Edge = top graph; Riffle = lower graph. Probability (y axis): 0–100%. Score categories: low: 0–0.4, red; moderate: 0.4–0.8, orange; high: 0.8–1.4, green.

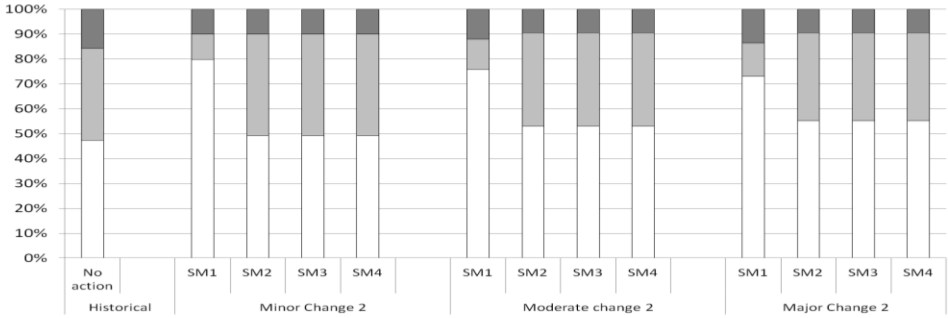


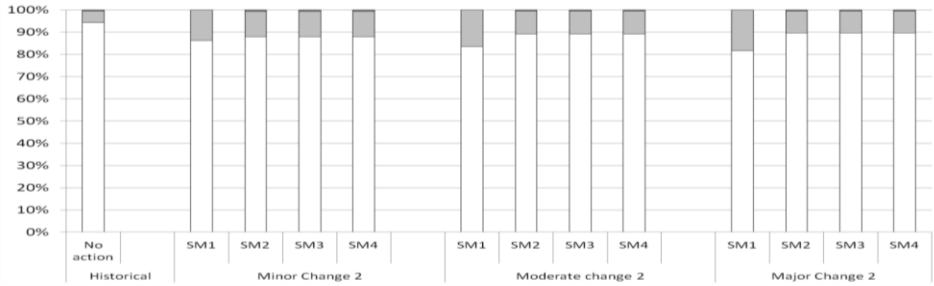


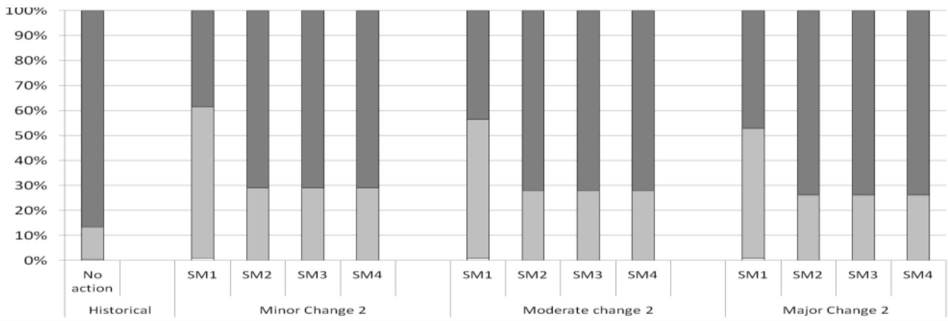
Figures 84 & 85. Predicted probabilities of Thermophobic taxa relative abundance in edge and riffle habitats in the Upper Murrumbidgee region, for 4 climate change scenarios and 4 management adaptation alternatives (l–r): Historical climate conditions + current management practices (no action); SM1–SM4 in each of Minor change 2, Moderate change 2 and Major change 2. Edge = top graph; Riffle = lower graph. Probability (y axis): 0% to 100%. Categories: 0–1, white; 1–5, pale grey; 5–20, dark grey; 20–100, black.











Figures 91 (top)–95. Predicted probabilities of the relative abundance in riffle habitats in the Upper Murrumbidgee region, of Leptophlebiidae (top), Gripopterygidae, Conoesucidae, Scirtidae, Chironomidae (bottom), for 4 climate change scenarios and 4 management adaptation alternatives (l–r): Historical climate conditions + current management practices (no action); SM1–SM4 in each of Minor change 2, Moderate change 2 and Major change 2. Probability (y axis): 0% to 100%. States: 0–2, white; 2–10, grey; 10–100, darker grey.

6.1.6 Upper Murrumbidgee region conclusions

The influence of climate change on macroinvertebrate communities is relatively small compared to that of flow management in the Upper Murrumbidgee Region. Changes in macroinvertebrates communities are largely driven by the release of flows from Tantangara Dam and an absence of flow from the dam results in large negative changes to macroinvertebrate communities, especially in the edge habitat. Differences in responses between edge and riffle habitats in the region require further investigation to determine the factors influencing the responses of macroinvertebrates in each habitat in response to flow management and further modifications of the riffle models may be required.

6.1.7 Lower Molonglo Region

The Lower Molonglo River is located to the north-east of Canberra and encompasses the area downstream of Canberra's major constructed waterbody, Lake Burley Griffin. The river receives runoff from urban streams, and the land use is a mixture of urban and peri-urban development combined with grazing operations. The river receives discharge from the Lower Molonglo Water Quality Control Centre (LMWQCC, the main sewage treatment plant for Canberra); hence all effluent management options were relevant. Results from the BN for this region are given in terms of each effluent management scenario (Figure 96).

(i) O/E scores

The four effluent management scenarios resulted in almost identical predictions of O/E score distributions for both edge and riffle habitats, and they all differed from the Historical, 'no action', scenario. Relative to historical conditions, in riffle habitat it is predicted that the probability of O/E scores being high will increase, while in edge sites a decrease is predicted (Figures 97, 98).

(ii) Lower Molonglo conclusions

A slight improvement in the O/E scores in riffle may be associated with the positive effect of salt reduction in the water because of the effluent management options being implemented. However, more data is required for the region to test the potential effects of the different effluent management scenarios here, because conditions predicted for climate change and effluent management scenarios are outside the range of the limited amount of data used to construct the models.

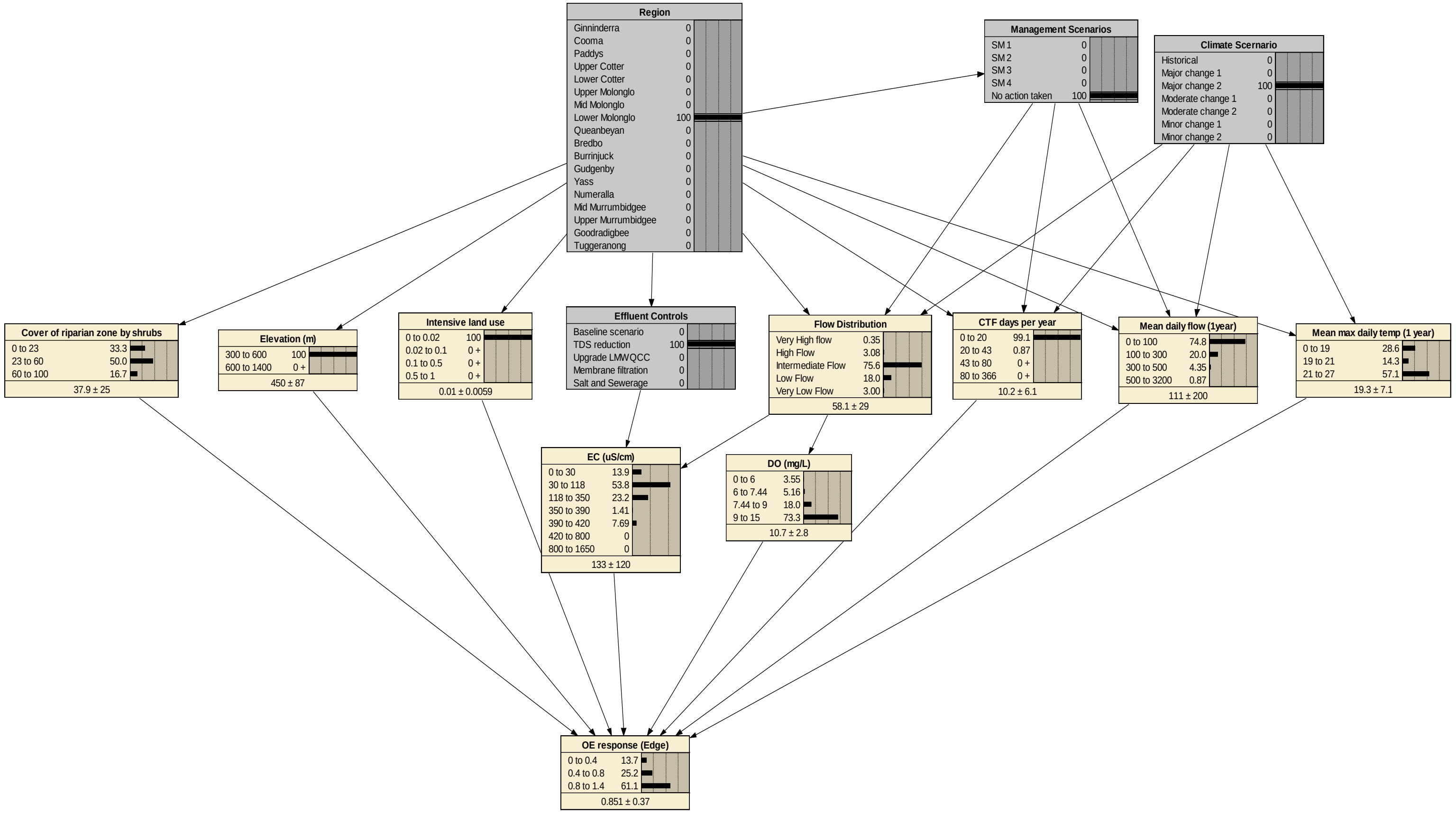
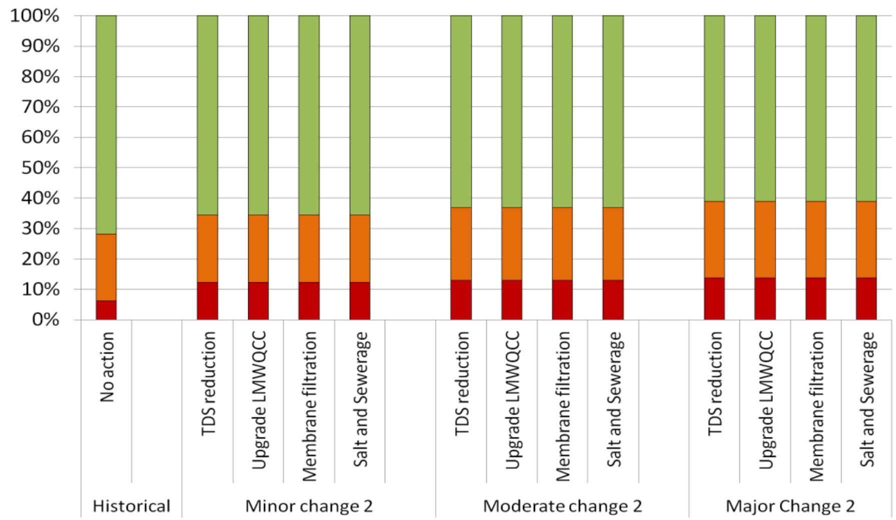
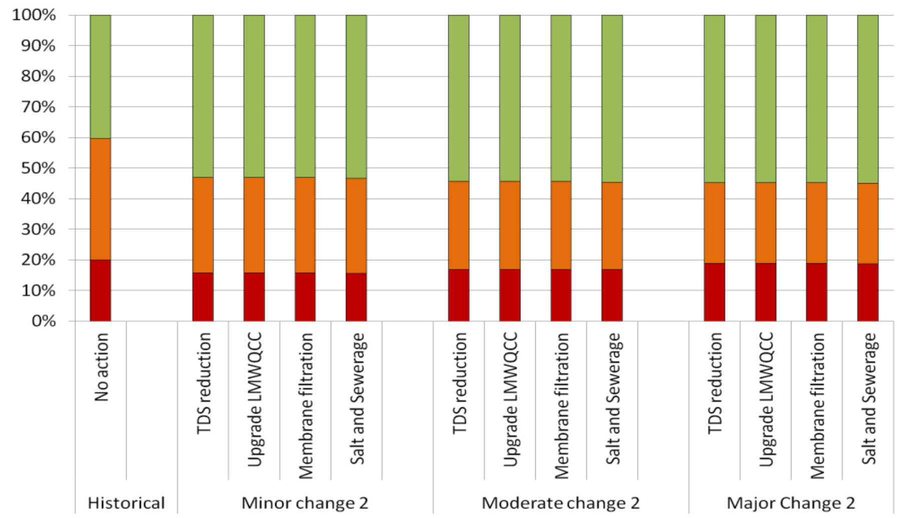


Figure 96. Example BN for OE taxa score in edge site under the TDS reduction scenario and the Major change 2 Climate scenario in the Lower Molonglo region.





Figures 97 & 98. Predicted probabilities of O/E scores in edge and riffle habitats in the Lower Molonglo region, for 4 climate change scenarios and 4 effluent management scenarios. (L–r): Historical climate conditions + current management practices (no action); Minor change 2, Moderate change 2 and Major change 2, each including TDS reduction, Upgrade LMWQCC, Membrane filtration, Salt and sewage.
Edge = top graph; Riffle = lower graph. Probability (y axis): 0–100%.
Score categories: low: 0–0.4, red; moderate: 0.4–0.8, orange; high: 0.8–1.4, green.

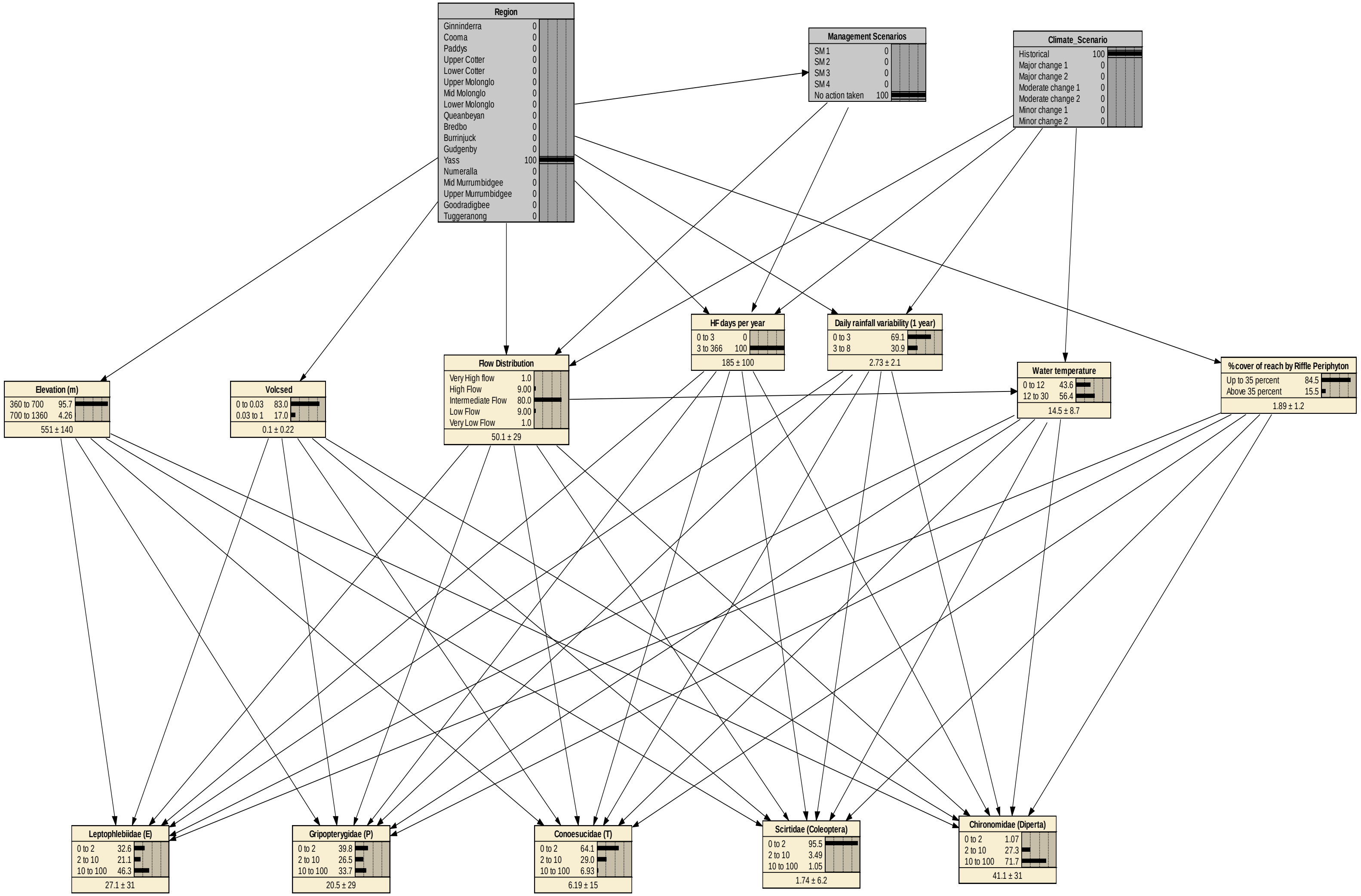
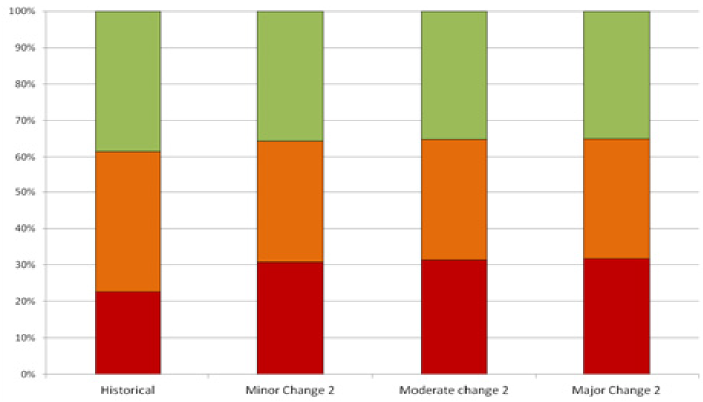
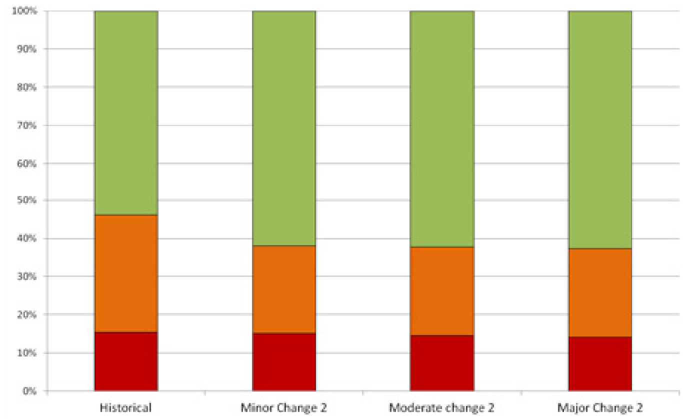
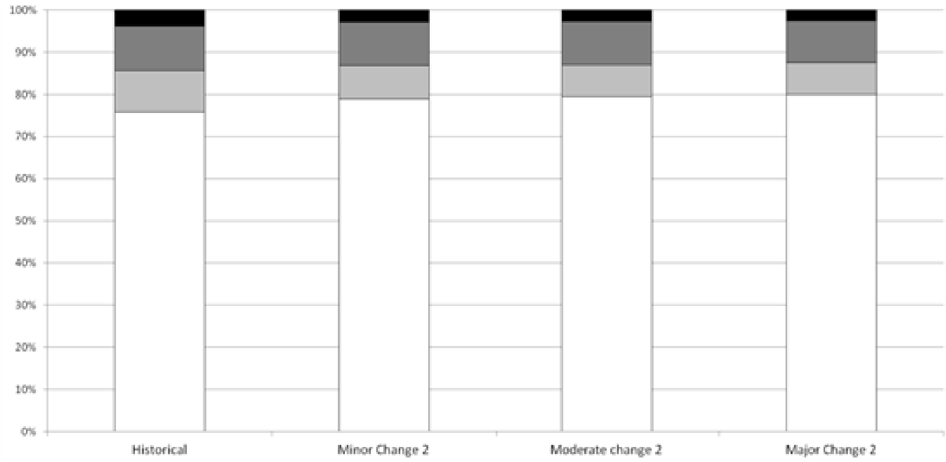


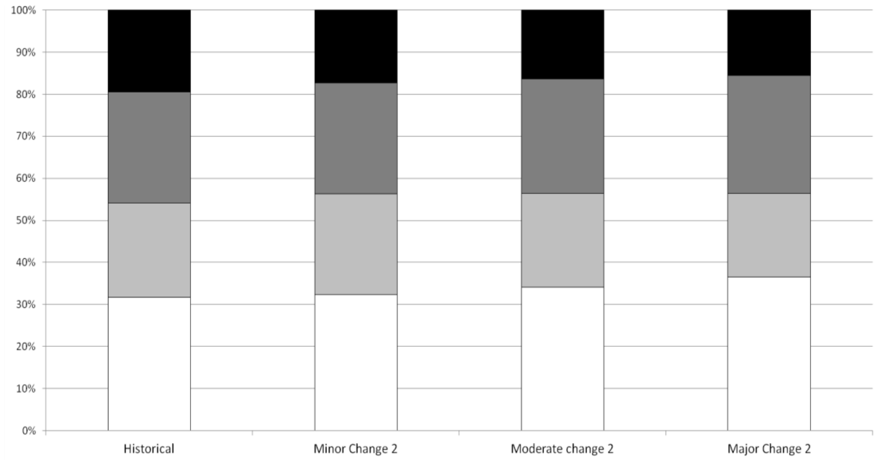
Figure 99. Example BN for the macroinvertebrate community (Riffle) showing the results for Yass region, no management action taken and historical climate scenario.



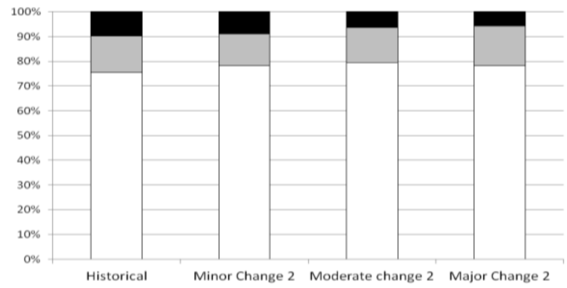


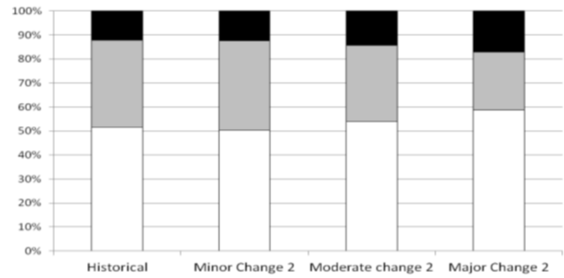
Figures 100 & 101. Predicted probabilities of O/E scores in edge and riffle habitats in the Yass region, for four climate change scenarios (l–r): Historical climate conditions, Minor change 2, Moderate change 2, Major change 2. Edge = top graph; Riffle = lower graph. Probabilities (y axis): 0% to 100%. Categories: low scores: 0–0.4, red; moderate scores: 0.4–0.8, orange; high scores: 0.8–1.4, green.

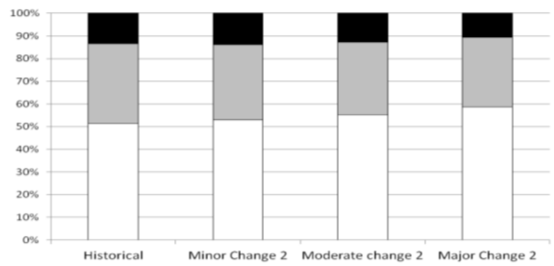


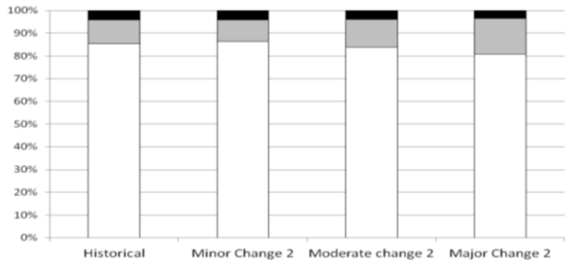


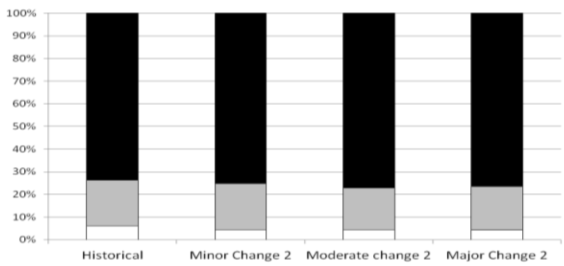
Figures 102 & 103. Predicted probabilities of Thermophobic taxa relative abundance in edge and riffle habitats in the Yass region, for four climate change scenarios (l–r): Historical climate conditions, Minor change 2, Moderate change 2, Major change 2. Edge = top graph; Riffle = lower graph. Probability (y axis): 0% to 100%. Categories: 0–1, white; 1–5, pale grey; 5–20, dark grey; 20–100, black.



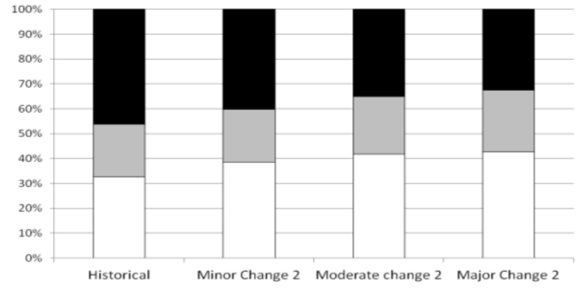


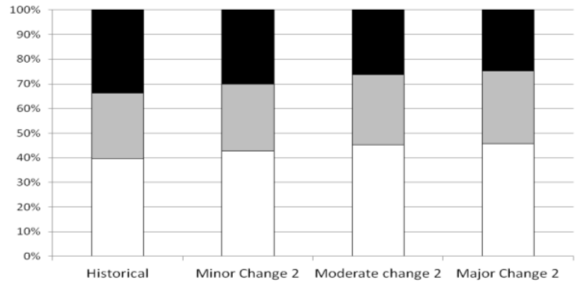


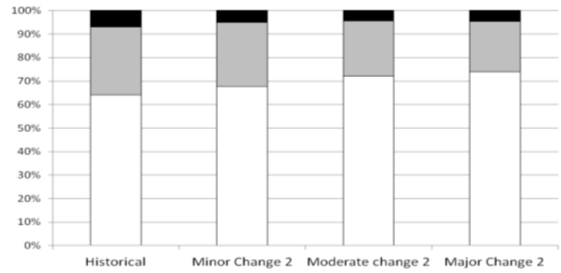


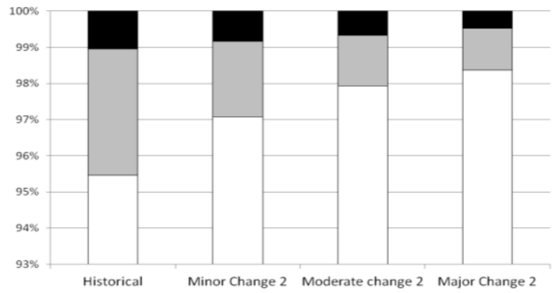


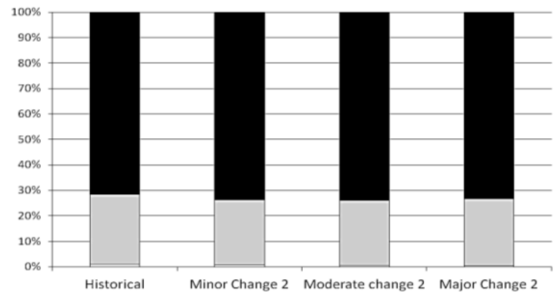
Figures 104 (top)–108. Predicted probabilities of the relative abundance in edge habits in the Yass region, of Leptophlebiidae (top), Gripopterygidae, Leptoceridae, Elmidae, Oligochaeta (bottom), for four climate change scenarios (l–r): Historical climate conditions, Minor change 2, Moderate change 2, Major change 2. Probability (y axis): 0% to 100%. States: 0–2, white; 2–10, grey; 10–100, black.











Figs 109-113, full page, 5 figs

Figures 109 (top)–113. Predicted probabilities of the relative abundance in riffle habitats in the Yass region, of Leptophlebiidae (top), Gripopterygidae, Conoesucidae, Scirtidae, Chironomidae (bottom), for four climate change scenarios (l–r): Historical climate conditions, Minor change 2, Moderate change 2, Major change 2. Probability (y axis): 0% to 100%. States: 0–2, white; 2–10, grey; 10–100, black.

6.1.8 Yass Region

The Yass region is located in the north of the Upper Murrumbidgee catchment. Most of the 1600 km^2 catchment is used for agricultural purposes: the dominant land uses are peri-urban development and grazing. Some intensive land uses (viticulture and horticultural activities) also exist within the catchment. The Yass River is unregulated; extractions for Yass town water supplies occur at Yass weir. For the Yass region, no management alternatives were relevant, so all results from the BN are given for the baseline management strategy "no action taken" (Figure 99).

(i) O/E scores

For the edge habitat, the probability of O/E scores being low (0–0.4) increased under the climate change scenarios (Figure 100). For the riffle habitat, a very minor but positive response to climate change occurred in O/E scores (Figure 101).

(ii) Thermophobic taxa relative abundance

Thermophobic taxa relative abundance in the edge and riffle habitats showed slight negative responses to increasing climate change severity (Figures 102**,** 103). As climate change severity increased there were slight increases the probability of absence of thermophobic taxa (0–1% relative abundance category) (Figures 102, 103).

(iii) Macroinvertebrate community indicators

For both the macroinvertebrate community indicator models (where select taxa were used to represent the macroinvertebrate community in both edge and riffle habitats), sensitive taxa (EPT) were predicted to show a negative response to climate change, while more tolerant common taxa were predicted to show a positive response to climate change.

In the edge model for community response, the sensitive species Leptophlebiidae, Gripopterygidae, Leptoceridae and Elmidae were predicted to show an overall negative response to climate change; however, for Gripopterygidae, no clear response was predicted (Figures 104–107). Meanwhile, there was a predicted increase in the more common tolerant Oligochaeta (worms) with changing climate scenarios (Figure 108).

In the whole community riffle model, Leptophlebiidae (Figure 109), Gripopterygidae (Figure 110), Conoesucidae (Figure 111) and Scirtidae (Figure 112) showed a predicted decrease in the probability of occurrence in the highest relative abundance category (20–100%) and a predicted increase in the probability of absences (0–1% category), with increases in air temperatures associated with changing climate scenarios. Conversely, tolerant Chironomidae showed a predicted increase in the probability of occurrence in the highest relative abundance category (20–100%) and a predicted increase in the probability of absences (0–1% category) with increasing severity of climate change (Figure 113).

(iv) Yass region conclusions

Increasing severity in climate change was predicted to have a negative effect on macroinvertebrate communities in the Yass Region. BN models still showed, for the most part, a negative response of sensitive taxa to increasing climate impacts, and a positive association by more common, tolerant macroinvertebrates (Oligochaeta and

Chironomidae) to increasing temperatures (air and water), thus indicating a possible decline in river health in the Yass region with climate change. Flow variability may play a role in the decline in river health, as as a result of changes in rainfall total or variability and increases in water temperature.

6.2 Storyline synthesis

The predicted responses of macroinvertebrate communities to theoretical scenarios of climate change and/or alternative management adaptations varied across the five regions examined. The strongest predicted effects of management occurred in the Upper Murrumbidgee region where there was the greatest direct impact of management alternatives on flows downstream of Tantangara Dam. In this region, the impacts of the theoretical adaptation alternatives appeared to outweigh the effects of climate change. In the Lower Molonglo region there was a potential effect from management alternatives relative to baseline conditions, illustrating the direct effects of effluent control options for this region.

Responses to climate change were predicted to occur in the Upper Cotter, Goodradigbee and Yass regions, with the latter two regions not subject to management adaptation alternatives.

Future investigation of macroinvertebrate community responses will need to consider regional differences. In some cases different responses between the edge and riffle habitat will also need to be taken into account. The most appropriate macroinvertebrate community condition indicator will also need to be considered, given the differing responses that were found across the five regions examined. In general, the predictions showed there was a consistent decrease in EPT taxa with climate change, except for Gripopterygidae in some regions (Yass and Upper Cotter regions).

The outputs provided by the BN assisted in addressing the three components proposed in Objective 3 (see section 1, Table 1), which are discussed below.

6.2.1 Evaluation of the consequences of alternative management adaptations in future water security and waste water management for water quality and ecological responses

We can make two key observations as a result of running the scenario models for the alternative management adaptations.

(i) There are different predicted responses to climate change and management between regions; and

(ii) ecological responses are predicted to vary within and between regions.

The differing responses between regions suggest that within the Upper Murrumbidgee River catchment as a whole, alternative management adaptations should not be uniformly applied in response to climate change. We observed that in the Upper Cotter region the alternative management adaptations appeared to have minimal, if any, influence. However, in other regions, alternative management adaptations could be highly beneficial.

For example, this project predicts that alternative management adaptations to secure water for Canberra from Tantangara Dam would have positive effects on the ecological condition of the Upper Murrumbidgee region. Similarly, if alternative management

adaptations involving effluent management were to be applied in the Lower Molonglo region, these should be beneficial for river health. This is an example where multiple benefits can be achieved from alternative management adaptations.

Our results suggest that it would be wise to consider the feasibility of alternative management adaptations, and adjust them to the local and spatial conditions within the catchment.

Secondly, the predicted differential ecological responses also suggest that management will need to consider their objectives in applying alternative management adaptations. It may not be possible to make general management adaptations, aiming simply to 'improve or maintain' ecological condition under climate change, because only some taxa may show positive responses.

This situation will be an issue both within and between regions in this overall Upper Murrumbidgee catchment, and possibly even at local scale (e.g. between edge and riffle habitats). Varying ecological responses are likely to play an important role in how alternative management adaptations are prioritised.

6.2.2 Prioritising for alternative management adaptations, based on probabilities of adverse effects

The predicted variations in regional and ecological responses have significant consequences for the prioritisation of alternative management adaptations. At the regional level, priorities for adaptation should take into account both the predicted impact of climate change and the consequences of different management actions.

In regulated rivers, where future climate and management scenarios are predicted to involve high water demand and declines in macroinvertebrate indicators (e.g. in the Upper Murrumbidgee region), then a focus on alternative management adaptations should be prioritised.

In contrast, in the unregulated, but stressed rivers of the region (e.g. Yass River), this project predicts that climate change will worsen the ecological health of the river, and would be likely to amplify current negative effects of catchment management practices. There are currently no alternative management adaptations for the Yass River, but its vulnerability to climate change suggests that some should be developed. Natural Resource Management activities (such as restoring the riparian zone) to help mitigate the effects of climate change are one option.

Finally in the rivers that are in 'reference condition', where there may be some slight negative climate change impacts for selected taxa (e.g. thermophobic taxa in the Goodradigbee River), and no management options, then it may be sufficient to continue assessment of effects of future climate change by using new climate scenarios. Continued protection of 'reference condition' areas also means that we have valuable places within the landscape where biota can adapt to climate change without the additional stressors of land use change.

In addition to considering the interplay between climate and management impacts at the regional level, ecological responses should also be considered when prioritising alternative management adaptations. Two possible approaches are

(i) prioritising action towards those ecological responses that show the highest vulnerability to climate change (e.g. thermophobic taxa), or

(ii) prioritising action towards taxa which are of the highest social ‘value’ (e.g. fish species). In practice, a combination of these approaches, among others, will likely be used to prioritise alternative management adaptations.

Regardless, prioritisation will need to take into account that ecological responses to initiatives may vary significantly, depending both on the species or community focused on in the study, and the region of interest.

7. TRANSFERABILITY AND REFLECTIONS

In this section we assess the transferability of the model framework to other regions, and we identify revisions the model requires for it to be broadly applicable. The section presents a framework for a linked modelling approach, and concludes with key learnings from the project, which will inform future use of the framework.

Managing freshwater resources for people and the environment requires an understanding of the relationship between climate change, water quality and ecological responses. Most research into the potential impacts of climate change on water resources has focused on the volumes of water available. In contrast, little attention has been directed at potential water quality changes, and consequent impacts on aquatic ecosystems.

This research project developed a modelling framework to link ecological and water quality responses to stream flow patterns and management activities (Figure 64). The framework will improve understanding about the consequences of climate change, including interactions between water management and freshwater ecosystem responses. The study has used the Upper Murrumbidgee River catchment as its context and source of data and models. Therefore, the modelling framework created for the Upper Murrumbidgee catchment will be of immediate use to the ACT Government and ACTEW Water, and regional local government. It should help them understand the effects of climate change on salinity and ecological response in streams.

Water resource managers from other regions of Australia have shown interest in this project, and we have therefore partnered with the Goulburn Broken Catchment Management Authority (CMA) to test the transferability of our approach within the Goulburn Broken (GB) Catchment (Figure 114). Results of the testing will help us refine the models and ensure that the approach is applied more broadly. The transparency of Bayesian Networks and the ease of updating models as new knowledge becomes available together make this project's approach easily accessible to stakeholders and other end users.

In this section we give an overview of the Goulburn Broken catchment as context for the transferability testing. The discussion is structured around the same aspects we covered for the Upper Murrumbidgee catchment, including the hydrological profile of the catchment, present and future adaptation options, water quality and ecological conditions. Then, we present the concepts of transferability, including cognitive and technical transferability, and use this to discuss how our modelling framework can be directly or indirectly transferred.

7.1 Goulburn Broken catchment

The Goulburn Broken (GB) catchment is located in northern central Victoria and covers area of 24,000 km^2 (Figure 115). The catchment extends from the Great Dividing Range (near the outskirts of Melbourne) up to the River Murray on the New South Wales border, and covers around 10.5% of the total area of Victoria. The area supports a population of around 215,000 people. Over 90% of the population reside in the major centres including Shepparton, Nagambie, Benalla, Kyabram and Tatura (CSIRO 2008a).

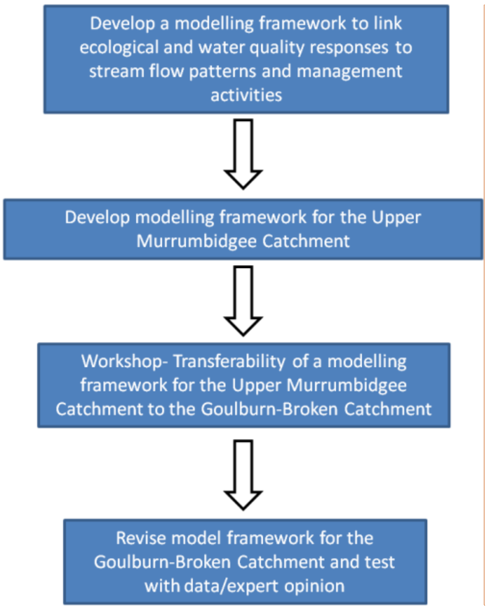


Figure 114. Project framework for model construction for the Upper Murrumbidgee catchment and transferability testing in the Goulburn-Broken Catchment[fc2]

Figure 115. A map of the Goulburn Broken Catchment (GBCMA 2012)

The catchment itself sits within the wider Murray Darling Basin (MDB) and makes up around 2.1% of the total area of the MDB, but produces about 11% of the MDB's total water resources (GBCMA 2005). Average annual rainfall in the GB region is 764 mm, varying between 1600 mm per year in the southern high country of the Great Dividing Range, to only 450 mm per year in the north-western plains (Miles et al. 2010).

The dominant land-use across the region is dryland agriculture, which covers 1.4 Mha (approximately 62% of the catchment) (GBCMA 2005), and irrigated agricultural production of, for example, perennial pasture, cereal, orchards and vine fruits, covers a further 9% of the region. The dryland area of the Goulburn-Broken Catchment generates around $1.9 billion in economic activity each year, and irrigated food production contributes to 25% of Victoria's export earnings (GBCMA 2005). Most of the low lying regions have been cleared of native vegetation for agricultural production; yet native vegetation still remains in the mountains and covers just over 30% of the total area of the Goulburn Broken catchment (CSIRO 2008a). Further land uses include tourism, recreational services, urban centres, as well as approximately 800,000 ha of public land. Based on 2009 figures, the gross regional output was $15.9 billion, with the gross value of agricultural production contributing $1.16 billion in 2009–10 and the value of tourism and other services reaching $2.29 billion, 30% of the gross regional output (increasing from only 3.9% in 2001) (Monticello 2012). Rivers and streams in the upper Goulburn catchment are highly valued for tourism and recreation, with many visitors from outside the catchment taking advantage of the high quality rivers and wetlands.

The Goulburn Broken catchment includes the two major basins of the Goulburn River and the Broken River. The Goulburn River basin is the largest in Victoria, occuping over 1.6 million hectares (7.1% of the state's total area) (GBCMA 2005). The river originates on the northern slopes of the Great Dividing Range near Woods Point, and flows in a north and westerly direction before joining the River Murray just upstream of Echuca on the NSW border. The river is approximately 570 km long, with an annual discharge of 3040 GL (1.8 ML/ha), representing 13.7% of the state's total discharge. The river is heavily regulated, and supports extensive irrigation regions. Lake Eildon, the largest of the regulatory features on the river, has a capacity of 3390 GL and supplies more than half of the water used in the Shepparton Irrigation Region.

The Eildon reservoir has altered the flow regime of the river so the winter/spring flows are reduced and summer/autumn flows are increased, essentially reversing the natural flow patterns. The Goulburn Weir also regulates the river and, along with its associated diversion channels, has reduced the average annual downstream flow to only 1340 GL, less than half the pre-regulated flow (GBCMA 2005). Water is diverted into the Cattanach and Stuart Murray canals and delivered to the Waranga Basin which is also connected to the Loddon and Campapse rivers by the Waranga Western Channel (CSIRO 2008a).

The Broken River basin occupies an area of around 772,000 ha, and is the major tributary of the Goulburn River, which it joins at Shepparton. Most of this basin has been cleared of native vegetation for agricultural purposes and a large part of the basin lies within the Murray Valley irrigation district which supports intensive horticulture, dairy and livestock production (GBCMA 2005). The mean annual flow within the Broken River itself is 325,000 ML (0.42 ML/ha), although over half of this is recorded between the months of July and September. Annual flow has varied greatly over the years, from only 5000 ML in

1943, under drought conditions, to 1,000,000 ML in 1917 and 1956, under flood conditions. The major water storage on the Broken River is Lake Nillahcootie, with a capacity of 40,000 ML which supplies water for stock, domestic and irrigation (GBCMA 2005). Lake Mokoan, now called the Winton Wetlands, was decommissioned as an active reservoir in 2010 and its natural wetland habitat is now being restored (GBCMA 2005).

The Broken-Boosey National Park, a unique linear corridor in the north of the basin covering 1030 ha, was declared a National Park after the passing of the Box-Ironbark Bill on 30 October 2002. This unique area stretches from the Barmah Forest (a Ramsar-listed wetland) to nearly the Broken River in the south.

(i) Water quality

The Goulburn Broken Catchment Management Authority (CMA) last prepared a Water Quality Strategy in June 1996, which outlined the priorities for water quality in the catchment over the following 20 years. The document was reviewed in 2002 (GBCMA 2002) and 2007 (GBCMA 2008). The strategy focuses on reducing the supply of excess nutrients to waterways, to minimise the impacts of blue-green algal blooms.

In 2010 the Regional River Health Strategy was reviewed (GBCMA 2010). The Progress report stated that water quality in 1990 had been poor, and it had since improved to a satisfactory level in 2009, with an improving trend. In particular, however, the 5-year rolling average for phosphorus load remained below its target. The decreased loads resulted from substantially decreased discharges from the region’s waste water treatment plants and irrigation drains. At the time of the Regional River Health Strategy review (GBCMA 2010), the Goulburn Broken catchment was still in the early stages of recovery from the 2006 and 2009 bushfires which had burnt substantial areas within the catchment. Short-term impacts on water quality were evident post-fire, with large increases in stream turbidity (Feehan 2012b). However, the effects of the fires on stream turbidity now seem to dissipating and there is no indication of a long-term effect on water quality (Feehan 2012b).

More recently, Newall et al. (2008) assessed the ecological risks to native fish communities in the Broken Creek and River. The assessment was called for in these parts of the catchment because of consistent failure to meet the State Environment Protection Policy (WoV; Government of Victoria 2003) guidelines on water quality, in particular the levels of turbidity, phosphorus and dissolved oxygen. Newall et al. (2008) present the status of these three parameters, as well as the targets and timeframe that was set to improve the conditions (Table 37).

Goulburn-Murray Water (G-MW) is the main regulator of the waterways in the Goulburn Broken catchment, and is responsible for the water quality within its storage facilities at Lake Eildon, Goulburn Waranga Basin and Lake Nillahcootie. Some of the major sources of risks to water quality in these areas are sedimentation, erosion, waste water management, stock access, and extreme weather events. These can lead to high turbidity, decreased dissolved oxygen, increased nutrient load (especially nitrogen and phosphorus) in water ways, and blue-green algae blooms (G-MW 2011). G-MW has developed BGA Response Plans to deal with algae blooms.

Table 37. Current status and condition targets for turbidity, phosphorus and dissolved oxygen in Broken Creek and Broken River (adapted from Newall et al. 2008)

Parameter	Site	WoV objective	Current status	Resource Condition Target (75th percentile)	Time frame	Uncertainty (confidence level to achieve target in timeframe)
Turbidity (NTU)	Broken Creek	<30	>100	<100 WoV (<30)	5 yrs (15 yrs)	Low to moderate
	Broken River		>80	<50 WoV (<30)	5 yrs (15 yrs)	Moderate
Phosphorus (mg/L)	Broken Creek	<0.045	>0.15	0.1 WoV (0.045)	5 yrs (15 yrs)	Low to moderate (Moderate)
	Broken River			0.1 WoV (0.045)	2 yrs (15 yrs)	Low to moderate (Moderate)
Dissolved oxygen (%sat)	Broken Creek	>85% <110%	60–100%	Maintain current conditions	Ongoing	Moderate to high. High below Flynns Weir.
	Broken River			Maintain current conditions	Ongoing	Moderate

Each year, Goulburn-Murray Water is required to produce a water quality report in compliance with the *Safe Drinking Water Act 2003*. One aim of this report is to highlight any incidents which may have posed a threat to the water quality of its storage facilities. In the 2010–11 Annual Water Quality Report (G-MW 2011) it was reported that during the period of wet weather over the summer, two regional water businesses were unable to dispose their treated waste water through irrigation of land, and eventually the businesses filled their onsite waste water storages to capacity. Emergency discharge licences under Section 30A of the *Environmental Protection Act 1970* were granted to G-MW for the business to discharge the excess treated waste water into the waterways within the catchment. The businesses had to undertake water quality and quantity monitoring and all downstream users were notified (G-MW 2011). Such discharges pose significant threats to water quality by increasing the nutrient load in the waterways and altering turbidity levels, amongst other risks, which can also have indirect impacts such as developing untimely blue-green algae blooms which need to then be managed and controlled.

At the time of the Regional River Health Strategy review (GBCMA 2010) the Goulburn Broken catchment was still in the early stages of recovery from the 2006 and 2009 bushfires which had devastated the landscape. These natural disasters had significant impacts on water quality; fire recovery management has been of utmost importance in the development of catchment strategies over recent years.

(ii) Ecological conditions

The Goulburn Broken catchment has significant ecological value in its pockets of native vegetation — which include River Redgum forest and woodland communities, and provide breeding ground for many waterbirds. The wetlands within the catchment are nationally and internationally recognised as important environment assets.

The Barmah-Millewa Forests in the Goulburn Broken catchment is listed under the Ramsar Convention as a wetland of international importance, and it is also listed in the Register of National Estate. These wetlands cover 29,000 ha. The Directory of Important Wetlands in Australia (EA 2001) identifies 10 wetlands (including the Barmah-Millewa Forests) in the Goulburn Broken catchment as nationally important. They include Broken Creek, Kanyapella Basin and the Lower Goulburn River Floodplain, which spans 13,000 ha from 150 km downstream of the Goulburn Weir to the Murray confluence. Six of the wetlands identified in the Directory are also listed under the Japan–Australia Migratory Birds Agreement (JAMBA) and/or the China–Australia Migratory Birds Agreement (CAMBA) (GBCMA 2005).

Three rivers in the catchment are identified as Heritage Rivers due to their environmental and social importance. Goulburn River supports intact River Redgum open forest and woodland and contains significant habitat for vulnerable wildlife including Squirrel Gliders and Murray cod. Big River and Howqua River both support habitat for the Spotted Tree Frog. They are also valued places for social activities such as fishing and canoeing.

The GBCMA's Regional River Health Strategy (GBCMA 2005) found there to be rare and threatened fauna species along 85% of the length of each of the Goulburn and Broken basins. A total of 60 species were identified, including three amphibian, 34 bird, 11 fish, two invertebrate, seven mammal and three reptile species. Rare and threatened flora species were much less widespread, being found along only 14% of the waterways in the Goulburn basin, and 50% of the length of the Broken basin. A total of 42 rare or threatened flora species were identified.

The condition of macroinvertebrate communities in the catchment has previously been rated "medium"–"excellent" (GBCMA 2005). It is a different story however for fish communities. The ratings for native fish communities varied across the catchment, with 31% of lengths of streams having a "good"–"excellent" rating, and around 28% of stream having "poor"–"very poor" rating for native fish. Very little of the catchment was found to be unaffected by introduced fish, with only 2–3% of the basins' streams rated to be in "good" or "excellent" condition (GBCMA 2005). The Murray-Darling Basin Commission Sustainable Rivers Audit rated the overall health of river systems across the catchment as being "very poor" (MDBC 2008).

Riparian vegetation was also assessed in the Regional River Health Strategy (GBCMA 2005); about 25% of the reaches in the Goulburn Broken catchment were found to have "poor" or "very poor" Riparian width, around 45% were found to have "poor" or "very poor" Riparian continuity, and most of the basin was found to have "medium" to "poor" structural intactness.

In 2010–11, 212.2 GL of the catchments' Environmental Water Allocation was released to manage dissolved oxygen levels in the Broken Creek and lower Goulburn River, and to maintain low flows in the lower Goulburn River, and to aid the breeding of waterbirds in the Barmah-Millewa Forest (GBCMA 2011).

During 2012–13, up to 59,000 ML and 231,937 ML of Commonwealth Environmental Water were made available for use in the lower Broken Creek and Goulburn River, respectively. This has hugely increased from the 2011–12 period, when only 30,000 ML and 96,900 ML were respectively made available because of the increased amount of rainfall in the catchment. This environmental water will help to facilitate fish movement, especially during migration and breeding, and maintain native fish habitat by marinating optimal levels of dissolved oxygen and restricting excessive aquatic plant growth (CEWO September 2012).

Major threats to the river systems in the Goulburn Broken catchment are identified in the Regional River Health Strategy, including (GBCMA 2005):

- stock access — with 76% of the Goulburn basin and 89% of the Broken basin affected,
- hydrological deviation found to be “extreme” or “extensive” over 23% and 43% of the total length of the Goulburn and Broken basins respectively. Hydrological deviation is based on the hydrology deviation index in the Index of Stream Condition, which consists of three key indicators, namely Amended Annual Proportional Flow Deviation which looks at the difference between natural and existing monthly flows, and catchment permeability, and the presence of hydroelectric power stations.
- barriers to fish migration — 62% and 69% of the length of the Goulburn and Broken basins were rated as “High” or “Very High” threat,
- condition of the streamside zone — threat level was rated “High” or “Very High” over 27% and 30% of the Goulburn and Broken basins, respectively. The condition of the streamside zone is defined in terms of width, longitudinal continuity, structural intactness, cover of exotic vegetation, regeneration of native species, and billabong condition.

As a part of a national assessment of water stress in catchments, SKM (2012) categorises the Broken River as one of the ‘least water stress-impacted basins’, whereas the Goulburn River is categorised as one of the ‘most water stress-impacted basins’. The basins were ranked in terms of hydrological impact, which was calculated using either the average or minimum score of the hydrological indicators used in the study, such as mean annual flow, monthly variation, high flood flows, low flow duration and seasonal period.

(iii) Legislation and regulations

The Goulburn Broken Catchment Management Authority (GBCMA) is a statutory authority established under the Victorian *Catchment and Land Protection Act 1994.* The CMA has developed and implemented a Regional Catchment Strategy, which provides a strategic framework for the alignment of sub-strategies which are used to action the long-term objectives for improving and maintaining natural resources in the Goulburn Broken catchment (Table 38).

(iv) Hydro-climate conditions

Over the past 15 years the Goulburn Broken catchment has experienced highly variable rainfall and river flows. From 1997 to 2006, average annual rainfall in the Goulburn Broken catchment has been about 15% lower than the long-term average (1895 to 2006) (CSIRO 2008a). Water flows were lower than the worst case predicted as recently as 2004, so water availability has been low for rivers and wetlands which has put aquatic

environments under severe pressure; threatened native species have experienced declines (Miles et al. 2010). However, from late 2010 onwards to early 2012 the catchment has experienced increased rainfall and flooding with La Nina weather conditions present.

Table 38. Summary of relevant Regional strategies in the Goulburn Broken Catchment (adapted from Miles et al. 2010)

<table>
<tr><td colspan="2">Goulburn Broken Regional Catchment Strategy 2003 (Updated 2011)</td><td>Overall</td></tr>
<tr><td>Dry Landscape Strategy
2008–11
An integrating strategy</td><td>Shepparton Irrigation Region Catchment Implementation Strategy
1990–2020
An integrating strategy and action plan</td><td>Integrating</td></tr>
<tr><td colspan="2">Issue-based sub-strategies and action plans:
• River Health Strategy 2005–15 (including Addendum 2010)
• Floodplain Management Strategy 2002
• Water Quality Strategy 1997–2027
• Shepparton Irrigation Region Farm and Environment, Surface and Sub-surface Action Plans 2007
• Dryland Salinity Management Plan draft 1990–2050
• Biodiversity Strategy 2010–15
• Native Vegetation Management Strategy 2000–30
• Strategic Plan for Integrating Native Biodiversity 2004–07
• Climate change position paper 2007
• Invasive Plants and Animals Strategy draft 2010
• Soil Health Strategy Draft 2002
• Landcare support Strategy 2004–09
• Monitoring, Evaluation and Reporting Strategy 2004–09</td><td>Theme or issue</td></tr>
</table>

The climate in the Goulburn Broken catchment is expected to get hotter and drier with the effects of climate change. This is likely to lead to lower average rainfall, but also great climate variability with more rainfall events, fewer frosts, and more hot days (Miles et al. 2010). With greater climate variability there is a higher risk of fire, floods and drought. High water temperatures and reduced stream flows could affect water quality, habitat values for aquatic and riparian species, and productive and recreational use (GBCMA 2012).

Climate changes are expected to affect surface water flows, the composition, abundance and distribution of weed species, and the process of pollination. These alterations will influence the compositions of ecosystems and the spatial distribution and abundance of species and communities (Miles et al. 2010).

Table 39 summarises the impacts of different possible future scenarios for the Goulburn Broken region. The scenarios are developed to represent different climate conditions combined with either current or possible future development levels. For Scenario ‘D’, the

projected growth in commercial forestry plantations is negligible. The total farm dam storage volume is projected to increase by 8.7 GL (8%) by ~2030 which would have minor impact (<1%) on river inflows. Groundwater extraction is expected to grow by 67% by 2030 to become around 16% of total water use. Average stream flow reduction of 37 GL/year (12 GL/year from groundwater extraction outside the Southern Riverine Plains area) is projected.

Table 39: Summary of CSIRO climate scenarios (adapted from CSIRO, 2008)[fc3]

	Mean Annual Rainfall, mm	**Modelled Runoff, mm**	**Evapo-transpiration**	**Surface Water Available**	**Water Diversions**	**Relative Level of Water Use**
Scenario 'A' – Baseline: Historical Climate & Current Development						
	764 (1500 south to 450 north) CV: 0.22	149 (400 south to 15 north) CV: 0.52		3233 GL/year	1099 GL/year to GB, plus 507 GL/year to Campaspe &Loddon-Avoca	50%
Scenario 'B': Recent Climate & Current Development						
Climate similar to that of last 10 years. Floodplains would cease to receive large flood events and there would be an increase in undesirably low flows.	15% lower	41% lower		-41%	-25% to G-B, and -25% to other regions	63% (+13%)
Scenario 'C': Future Climate & Current Development						
Climate projections for 2030, from 15 GCMs – median 'best estimate', wet variant and dry variant. - CDry: continuation of recent conditions, - CMid: Reduction in flooding and increase in low flow, - CWet: little change from current.	CDry: -19% CMid: -4% CWet: 0%	CDry: -44% CMid: -13% CWet: 2%	CDry: -12% CMid: -2% CWet: 0%	CDry: -45% CMid: -14% CWet: -3%	CDry: -32% out of region. CMid: -6% to G-B, and -5% to other regions. CWet: 0% out of region.	CDry: 21% (-29%) CMid: 54% (+4%) CWet: 49% (-1%)
Scenario 'D': Future Climate & Future Development						
Scenario 'C' plus developments including expansion of farm dams and commercial plantation forestry, increase in GW extraction, and also improvements in water use efficiency.	DDry: -19% DMid: -4% DWet: 0%	DDry: -44% DMid: -14% DWet: -3%	DDry: -12% DMid: -1% DWet: 1%			

[FD4]

At present surface water extraction within the catchment is very high, with 50% of available surface water being diverted for use or transferred for use in other water regions (CSIRO 2008a). Under the 'best estimate' (CMid) 2030 climate conditions, the average surface water availability would be reduced by 14% and diversion would decrease by 5% (Table 40).

(v) Bushfire impact

Over a third (325,823 ha) of the Goulburn Broken catchment's woody native vegetation was burnt in the 2006 and 2009 bushfires (according to spatial data from the Dept of Sustainability & Environment); the upper part of the catchment is still recovering from the 2009 'Black Saturday' bushfires. The 2009 fires burnt 190,000 ha, 8% of the entire Goulburn Broken catchment, including 2845km of stream frontage and 212 km of native vegetation along major rivers (Miles et al. 2010).

Since the 2009 bushfires, community and institutional reactions are heavily influencing planning regulations, in particular with regard to the positioning of new buildings in the landscape and the clearing of native vegetation for fire protection (Miles et al. 2010). These influences pose a threat to biodiversity in areas which are now deemed fire hazards to the local community.

In future climatic conditions, the risk of occurrence and severity of bushfires is likely to increase because of expected warmer and drier weather, as well as human impacts (Miles et al. 2010).

(vi) Urban water demand

Urban water use in the Goulburn Broken catchment is very small compared to irrigation water use (CSIRO 2008a). Current annual urban water use is around 40 GL (CSIRO 2008a). However, the population is growing rapidly in the region, at a rate of 1.23% (or 2600) each year, which is marginally higher than the average Victorian population growth rate. By 2026 it is expected to reach 255,500 (Monticello 2012). The Goulburn Valley Water's Water Supply Demand Strategy 2012–2060 (GVW 2012a) highlights the way water demand in the Goulburn Valley region is forecasted to change over the next 50 years (Table 40). It may also be relevant to consider the impacts of wider population growth and climate change on the use of the 'Sugarloaf' pipeline to transfer water resources from the catchment to Melbourne (MW 2012).

Table 40. Changes in water demand in the Goulburn Valley between 2011 and 2060 (adapted from GVW 2012b)

Demand Component	2011 (ML)	2060 (ML)
Residential	13,634	23,158
Commercial	3,836	5,125
Major Customer	6,479	6,479
Distribution System Non-Revenue	2,392	3,563
Headworks Non-Revenue	2,941	4,056
Total	**29,281**	**42,380**

(vii) Land-use changes

Storage capacity of small farm dams used for irrigation and stock and domestic purposes is estimated at 105 GL across the catchment (CSIRO 2008a), and future development is thought to be minimal because of efficiency improvements.

Land use changes can be both a threat and advantage for biodiversity. For example, the GBCMA believes that the general landholder in the catchment now has a better understanding of the needs of biodiversity, and there has been a parallel improvement in the uptake of environmental projects and work on private land (Miles et al. 2010). Social adjustment of farming practices to incorporate biodiversity enterprises, such as integrated farm forestry, may help to improve the condition of the local environment; emerging markets for carbon may manifest in carbon sequestration schemes which support biodiversity conservation.

Changes to energy markets and resources may also impact on the land use within the catchment because the search for alternatives to fossil fuels may result in changes to agricultural practices as well as products. They may also influence living preferences and tourism (Miles et al. 2010).

(viii) Adaptation policies

(a) Irrigation efficiency programs

Implementation of water delivery and on-farm irrigation efficiency measures that save water which can be used to meet environmental, economic and social needs (GBCMA 2012) has occurred within the catchment over recent years. These savings have been part of the Connections Project (formerly the Northern Victoria Irrigation Renewal Project, NVIRP) and the Farm Water Program.

The Victorian and Australian Governments have made a $2 billion investment to upgrade and rationalise irrigation water delivery systems in the catchment, resulting in water saving for the benefit of irrigators and the environment (Miles et al. 2010). In July 2012, the NVIRP merged into Goulburn-Murray Water to deliver that company's Connections Project (GVW 2012a).The project was originally set up in 2007 to manage and implement the modernisation of the irrigation system supplying water to the Goulburn Murray Irrigation District (GMID), covering 68,000 km^2. It was estimated that up to 900 GL of water was being lost in the irrigation system due to leaks, evaporation and inefficiencies before this investment. The work aims to save up to 425 GL per year to increase the water efficiency of the irrigation system from 70% up to 85%. In the Central Goulburn region, 902 gates, 1576 meters and 50 km of channel remediation had been completed between 2008 and July 2012, and 95 meters and a further 16 km of channel remediation were planned for the remainder of 2012 (G-MW 2011).

The GBCMA has led the Farm Water Program, and in Round 1 of the program (2010–11), 149 irrigation projects were funded across the Goulburn Murray Irrigation District. The Australian Government contributed $21.1 million, and the NVIRP contributed a further $16.4 million (GBCMA 2011). A total of 9 GL of water was saved and transferred to the Australian Government and NVIRP. This water saving is being used to address salinity issues, environmental water flows and improving river health.

(b) Urban Demand management

Water Efficiency Labelling and Standards (WELS) Scheme: Australia's water efficiency labelling campaign, WELS, allows consumers to compare the water efficiency of products such as showers, taps, toilets and washing machines. The scheme is designed to encourage consumers to buy water saving products (GVW 2012b).

Permanent Water Savings Rules (PWSR): These Rules are set out as a series of common sense rules that customers of Victorian Water Authorities are obliged to follow so they do not waste water on a daily basis. The rules are in place to encourage efficient use of water at all times, without restricting consumers' choice and flexibility. The rules include using fitted trigger nozzles on hoses, using watering systems for gardens and lawns between the hours of 6 pm and 10 am, and not permitting hosing down of driveways, paths, concrete, paved areas or timber decking except in limited circumstances (GVW 2012b).

Water Restrictions: Goulburn-Valley Water implements water restrictions by law when necessary. These restrictions vary from Stage 1 to Stage 4 (GVW 2012b).

Use of Recycled Water: Recycled water is used by agricultural enterprises managed by Goulburn Valley Water as well as other third party users. There is increasing interest in the use of recycled water for irrigation, and third party agreements for the use of recycled water have been created with farming enterprises and golf clubs. To maximise this use, winter storage lagoons have been constructed at all Wastewater Management Facilities (WMFs) and land has been made available for irrigation; as a result of this most WMFs now experiences full re-use of recycled water (GVW 2012b).

(ix) Water quality and aquatic ecosystems protection

Water buybacks: The proposed Basin Plan from the Murray Darling Basin Authority (MDBA 2012a) included the concept of water buybacks. The Basin Plan aims to re-balance the water needs of the environment alongside other uses, and to do this it has defined new volumetric limits (sustainable diversion limits) on water use (MDBA 2012b).

The Basin Plan, in the context of the Goulburn Broken catchment area: Between 2009 and 2011, a total of 245 GL/year has been recovered from the Goulburn Broken catchment: 75 GL/year from NVIRP Stage 1, 166 GL/year by Australian Government water purchases and the remaining 4 GL/year by Australian Government infrastructure projects. Further work is still needed, and the Basin Plan suggests that a further 99 GL/year needs to be returned to the environment in the Goulburn Broken catchment. The NVIRP Stage 2 is, however, expected to recover an extra 214 GL/year which will meet the Basin Plan environmental water requirements (MDBA 2012b).

The water recovered through the Basin Plan buybacks and the NVIRP efficiency improvements will benefit the lower floodplain, with its wetlands of national and international importance, as well as benefiting the river channel itself. For the lower floodplain, the aim is to increase the frequency of flows to the key wetlands to improve the health of reed beds and River Redgums, provide habitat for migratory and native waterbirds and help to support native fish populations, including the Murray Cod and Trout Cod. Note that these increased floods are not planned to impact upon communities built upon the floodplains (MDBA 2012b).

(x) Education programs

GVW and GBCMA have been long term sponsors of the Goulburn Broken Waterwatch program (GVW 2012b) which raises awareness of water quality issues and helps to encourage practical solutions from within the community, to help to improve the water quality in the catchment. GVW is also involved in programs such as National Water Week which also helps to raise awareness of water saving solutions and water quality issues (GVW 2012b).

(xi) Regional Catchment Strategy

The strategic objectives of the GBCMA Regional Catchment Strategy (GBCMA 2012) include:

- adapt to water policy reform,
- adapt to land-use change,
- adapt to climate variability,
- adapt to increased farm productivity, and
- strengthen partnerships.

Table 41 outlines the long-term objectives (20–30 years) of the Regional Catchment Strategy regarding waterway and floodplain condition. Table 42 highlights some examples of the adaptive management measures which may be put in place, through the relevant sub-strategies and in partnership with the relevant authorities, in order to address the strategic objectives. These measures are some of those that refer directly to the health of waterways in the Goulburn Broken region.

(xii) Climate change research

As part of a larger project to undertake ecological risk assessments, funding has been provided to GBCMA to undertake a project looking at impacts of climate change on waterway values, in particular water quality. The project is exploring the impact and location of changes on waterway values and current programs (Feehan 2012a). The risks of crossing water quality thresholds in an environment of climate change, and their effect on waterway values and current programs, are also being considered from a number of perspectives:

- likelihood of crossing the threshold
- consequence of crossing the threshold
- managing the likelihood and consequence.

The project is also exploring the concepts of resilience and thresholds to assist in answering risk management questions and what the consequences of crossing thresholds are for ecosystems.

Table 41. Long term (20–30 years) objectives of the GBCMA Regional Catchment Strategy regarding waterway condition (Miles et al. 2010).

Salinity	Keep increases to salinity levels of the River Murray at Morgan at or below 8.9 EC. Ensure no net increases in stream salinity in the Goulburn Ricer upstream of Goulburn Weir. Prioritise protection of foothills and river valleys of highland areas from salinisation threatening significant terrestrial and aquatic assets.
River Health	350km of river maintained in excellent or good condition. Ecological flow objectives met n high value reaches. Nutrient loads reduced or improved. Riparian condition protected or enhanced along 550 km of river. Instream habitat enhanced or reinstated along 140 km of river.
Wetlands	Maintain extent of all wetlands types at 2003 levels. Improve conditions of 70% of wetlands by 2030 (benchmark 2003). Strategic approach to managing the threat of saline GW on wetlands within the SIR documents about protecting the assets from rising watertables.

Water Quality	1996 target: reduce potential phosphorus load by 65% by 2016, by reducing phosphorus loads from: —irrigation drains by 50%, —dryland and diffuse sources by 20%, —waste water management facilities by 80%, —urban stormwater, —intensive agricultural industries and local water quality issues.
Floodplains	Reduce the impact of flooding on the built environment. Provide the ecosystems with natural flooding patterns where appropriate.

Table 42. Strategic objectives, Strategic priorities and Adaptive management measures directly relating to waterways and wetlands (GBCMA 2012)

Management measures	**What may these look like?**	**Relevant sub-strategy**	**Partners**
Adapt to water policy reform			
Water delivery to waterways and wetlands			
Plan, deliver and monitor environmental water delivery to improve the condition of priority waterways and wetlands	Meet Ramsar obligations for Barmah National Park including provision of natural flooding patterns	RRHSS	Commonwealth and Victorian Environmental Water Holders, DSE, Parks Victoria, Yorta Yorta Nation, MDBA, G-MW
Modernise water delivery on irrigated land to provide ecological and productivity benefits	Continue to implement the Farm Water Program for ecological and agricultural production benefits	SIRCIS	Landholders, North Central and North East CMAs, DPI, DSE, G-MW, Dairy Australia, Northern Victorian Irrigators, Murray Dairy, SEWPaC
Adapt to land-use change			
Managing land develop risks			
Manage wastewater treatment and stormwater runoff to minimise pollutants to urban waterways and wetlands		WQS	GVW, EPA, local government
Flood management			
Understand more about the nature of flooding to manage its impacts on the natural and built environments		FMS	G-MW, local government, GVW
Adapt to climate variability			
Adapting to climate variability risks			

Management measures	What may these look like?	Relevant sub-strategy	Partners
Factor risks of climate variability and identify adaptation strategies in GBCMA plans	Undertake risk assessment for specific biodiversity assets to determine priorities for investment. Seek research into the effects of climate change on native forests.	CCIS	
Contribute to public land fire management plans to minimise loss of biodiversity	Input into a strategic approach to planned burning in Barmah that considers ecological values. Develop an education campaign that targets new landholders. Identify where the CMA can have influence and build partnerships with public land managers, and nurture partnerships with DSE and Parks Victoria so as to have input as appropriate.	CCIS	DSE, Parks Victoria, Traditional Owners, CFA, local government, community
Climatic events response and recovery			
Partner in planning and implementing flood, fire and drought response and recovery	Introduction of new landholder packages for land management. Identify risks to drought refuge areas and develop mitigation plans.	CCIS	DSE, DPI, local government, water authorities, Parks Victoria, CFA, SES
Low carbon future opportunities			
Identify where carbon sequestration activities provide environmental economic and social benefits	Identify areas no longer under irrigation that may be suitable for biodiversity planting for carbon sequestration and diversification of income. Implement the NRM Planning in Climate Change Initiative.	CCIS	Landholders, community groups, Australian Government, DPI, industry
Adapt to increased farm productivity			
Increasing biodiversity in agricultural land use			
Create awareness and acceptance of land management practices that protect and improve terrestrial and aquatic habitat	Provide education opportunities through Conservation Management Networks	BS, IPAS	Community groups, landholders, local government
Identify environmental stewardship opportunities for land managers	Support community-led projects like the Superb Parrot, the Grey Crown Babbler and Regent Honeyeater projects.	BS	Community groups, Trust for Nature, local government, landholders
Work with landholders to protect and improve biodiversity on private land and build understanding of its contribution to the landscapes.	Education campaigns to target high value assets Implement a biodiversity tender that protects and enhances habitat, and provide landholders with incentives to improve the condition of terrestrial, riparian and wetland habitat.	BS, IPAS, SHS	DPI, G-MW, DSE, landholders, community groups and networks, Trust for Nature.
Strengthening Partnerships			
Partnering public land managers			

Management measures	What may these look like?	Relevant sub-strategy	Partners
Undertake works with public land managers to improve waterways and wetlands, including private landholders managing Crown land frontage	Provide landholder with incentives to improve riparian vegetation management. Research trials on different riparian management regimes in collaboration with private land managers on river reaches.		

7.1.1 Similarities and differences between catchments

There are a number of similarities between the Upper Murrumbidgee catchment and the Goulburn Broken catchment.

- Both catchments have highly variable ecological and water quality. Upland areas are in good ecological condition and have good water quality, while lowland areas have poorer ecological condition and water quality.
- Reduction in native fish species abundance is an issue in both catchments.
- Rivers in both catchments are highly regulated.
- Management authorities in both catchments are actively involved in climate change research and have put, and / or are putting, policies in place for climate change adaptation.

There are also differences between the Upper Murrumbidgee catchment and the Goulburn Broken catchment.

- The Goulburn Broken catchment is considerably larger than the Upper Murrumbidgee catchment: 24,000 km^2 vs 13,144 km^2.
- Urban land use is higher in the Upper Murrumbidgee catchment. Therefore, effluent discharge and water quality issues associated with urban land use are a bigger issue in the Upper Murrumbidgee catchment.
- Irrigation land use is much higher in the Goulburn Broken catchment. Therefore, changes in river flow and water quality issues associated with irrigation are a bigger issue in the Goulburn Broken catchment.
- Lowland wetlands are important ecological assets in the Goulburn Broken catchment (e.g. Barmah Millewa Forests). These types of wetlands do not exist in the Upper Murrumbidgee catchment.
- The purpose of river regulation differs between the two catchments. In the Upper Murrumbidgee catchment, river regulation is primarily for urban water supply, while in the Goulburn Broken, flow regulation is primarily for irrigation supply. Hence, the purpose, timing and magnitude of flow releases from water supply dams are also different between the two catchments.

7.2 Methods

7.2.1 Workshop

A workshop was held on the 31st October 2012 in Shepparton with attendees from the Goulburn Broken CMA, Goulburn Murray Water, Goulburn Valley Water and Department of Sustainability & Environment. The objectives of the workshop were to:

1. identify the transferability of the approach and learnings from the development of this project's model framework (cognitive transferability);
2. review the draft BN models developed for the Upper Murrumbidgee catchment for their applicability to the Goulburn Broken catchment and identify model modifications required to make them more broadly applicable (technical transferability);
3. identify data sets and information that will enable revised models to be populated.

In the future the models will be populated using data sets supplied by the Goulburn Broken CMA and model tested using sensitivity and uncertainty analysis techniques to

determine the confidence in the model predictions. The workshop provided an opportunity to discuss and share knowledge on the transferability of the modelling framework designed for the Upper Murrumbidgee catchment to the Goulburn Broken catchment.

Two aspects of transferability were considered in the process of designing the workshop (Grafton et al. 2012):

- cognitive transferability focuses on the transferability of the reasoning and judgement behind the model development; and
- technical transferability focuses on the transferability of the model structure, endpoints and scenarios.

Both cognitive and technical transferability are seen as important components of assessing model transferability between catchments (Grafton et al. 2012). During the workshop, two key questions were addressed:

1. Cognitive transferability — are the key learnings from developing the models in the Upper Murrumbidigee catchment transferable and relevant to the Goulburn Broken catchment?
2. Technical transferability — does the model structure (both the detailed conceptual model and the refined model) need to be adjusted to be specific to the Goulburn Broken catchment?

7.2.2 Modelling

A more broadly applicable model framework was proposed based on input from the Shepparton workshop. This model was compared with the Upper Murrumbidgee modelling approach and key differences highlighted.

7.2.3 Cognitive transferability

Question: Are the key learnings from developing the models in the Upper Murrumbdigee Murrumbidgee River Catchment transferable and relevant to the Goulburn Broken Catchment?

Scenario-based approach

The scenario approach has been used within the Goulburn Broken region and is supported because of its ability to manage complexity and provide input to management decisions. Previous scenario planning approaches include the irrigation futures project (Robertson et al. 2007). In terms of climate models, more than a few models (GCM outputs) are considered to provide too much information and complexity to consider within a management context (similar to the Upper Murrumbidgee catchment).

There is preference to use four scenarios: Wet, Average, Dry and worst on record, and compare these with historical behaviour. Learning from post-fire rainfall impacts on water quality has come from the 2003, 2006 and 2009 fires in the two catchments.

Endpoints

Ecological endpoints need to be framed in a way that is directly relevant to management. Identifying shifts in community structure is interesting, but not particularly relevant from a management perspective. For instance, if macroinvertebrates are used as the endpoints, then scoring systems that indicate ‘health’ (such as AUSRIVAS O/E scores; or Index of

Stream Condition scores) are more helpful to managers from the perspective of identifying strategies. For the Goulburn Broken catchment, a range of endpoints might be chosen, with a range of ecological assets of interest within regions:

- Fish — both species and communities
- Platypus
- Water rats
- River redgums
- Blue Green Algae
- Macroinvertebrates

Given that tourism within both catchments is also important, recreational use, drinking water and aesthetic features of the water are also of interest as an endpoint to the modelling framework.

It is likely that the drivers of the ecological responses will vary depending on both catchment and possibly region — particularly in an area like the Goulburn Broken catchment where regional differences are more pronounced than in the Upper Murrumbidgee catchment. For example, in the Goulburn Broken catchment there are differences in land use between upland and lowland areas (lowland areas have irrigation). Furthermore the Goulburn River is highly regulated compared to the Broken River. This means that a different model structure will be defined each time the framework is used.

Other points arose during the workshop.

- To enhance model transferability a generic model of water quality responses with ‘add on’ ecological response models was suggested.
- Understanding the water quality changes from a global perspective has greater value to the CMA and the water authorities in their regional planning activities.

7.2.4 Technical transferability

Question: Does the model structure (both the detailed conceptual model and the refined model) need to be adjusted to be specific to the Goulburn Broken catchment?

Model simplification based on identification of key drivers (bottom-up model structuring) was seen as an appropriate approach and important if the endpoints were clearly defined, but it results in models that are not easily transferred and therefore the technical transferability was considered to be limited.

The current model framework for the Upper Murrumbidgee catchment, and the modified framework for the Goulburn Broken catchment, are shown in Figures 66 and 67, at the end of this section.

Details of model framework additions, removals, other modifications and considerations are outlined below.

Model framework additions

- Irrigation drain returns

- Intensive land use will need to be modified to reflect regional land use characteristics. Irrigation activities will need to be separated to identify horticulture and dairy farming as separate categories.
- The source of water:
    - Water from a release from Lake Eildon is different in quality from water that results from runoff from the Acheron tributaries or the Sunday/Mollison's tributaries (i.e. Goulburn River tributaries below Eildon Dam).

Model framework removals

- Effluent management options are not important within the region.
- pH removed — not seen as a water quality issue in the Goulburn Broken catchment.

Other model framework modifications and considerations

Regions: A regionally based approach is appropriate: Eight regions

1. Alpine
2. Upper Goulburn
3. Goulburn Tablelands
4. Goulburn Irrigation
5. Upper Broken River
6. Lower Broken River
7. Broken Creek Anabranch
8. Broken Creek Irrigation

Water quality attributes needed for a generic model:

- DO
- TN
- TP
- EC
- Turbidity

Water quality thresholds:

- State Environment Protection Policy (Water of Victoria) defines water quality objectives that would be useful to use.
- ANZECC guidelines are also helpful.
- Tiller & Newell (1995) — Victorian nutrient guidelines.

Endpoints of interest:

- Ecological asset based.

Management scenarios to include:

- environmental flows,
- riparian vegetation management,
- barriers (fish),
- management of exotic species (fish and riparian weeds),
- sediment management (erosion control; pasture management; riparian areas).

Additional drivers to consider:

- SKM report identified ‘Days since last rainfall’ as being a predictor of water quality (suspended solids) in the Goulburn Broken Region. This was not tested in the Upper Murrumbidgee catchment.

Data considerations for the Goulburn Broken catchment:

- Accurate land use data for the catchment may be difficult to obtain.

7.3 Conclusions

The conceptual thinking behind the modelling framework constructed for the Upper Murrumbidgee catchment (Figure 68) is generally transferable to the Goulburn Broken catchment. However, in terms of technical transferability some changes need to be made to the modelling framework to make it applicable to the Goulburn Broken catchment.

In the Goulburn Broken catchment there are differences in land use between upland and lowland areas. The upland area of the Goulburn Broken catchment is very similar to the Upper Murrumbidgee catchment, while the lowland areas used for irrigation are vastly different. Furthermore, the Goulburn River is highly regulated compared to the Broken River. This means that at a regional scale a different model structure will be defined each time the framework is used.

The bottom-up model structuring method applied in the Upper Murrumbidgee catchment is a scientifically valid method, which leads to a parsimonious and manageable model structure. However, this does not facilitate model transferability, because of regional differences in management.
Managers in the Upper Murrumbidgee catchment are interested in ecological thresholds and consider it an important part of the process. Managers in the Goulburn-Broken are less interested in ecological threshold identification and more interested in using endpoints that can be easily interpreted by management (e.g. AUSRIVAS O/E scores; or Index of Stream Condition scores). The next step for the modelling framework to be transferable to the Goulburn Broken is to focus on applying a top-down model with all water quality endpoints, which can then be linked into component ecological response models. However, before this can take place, river flows under different climate scenarios for the Goulburn Broken need to be calculated to understand the effects of climate change on river flows and water quality within the catchment.

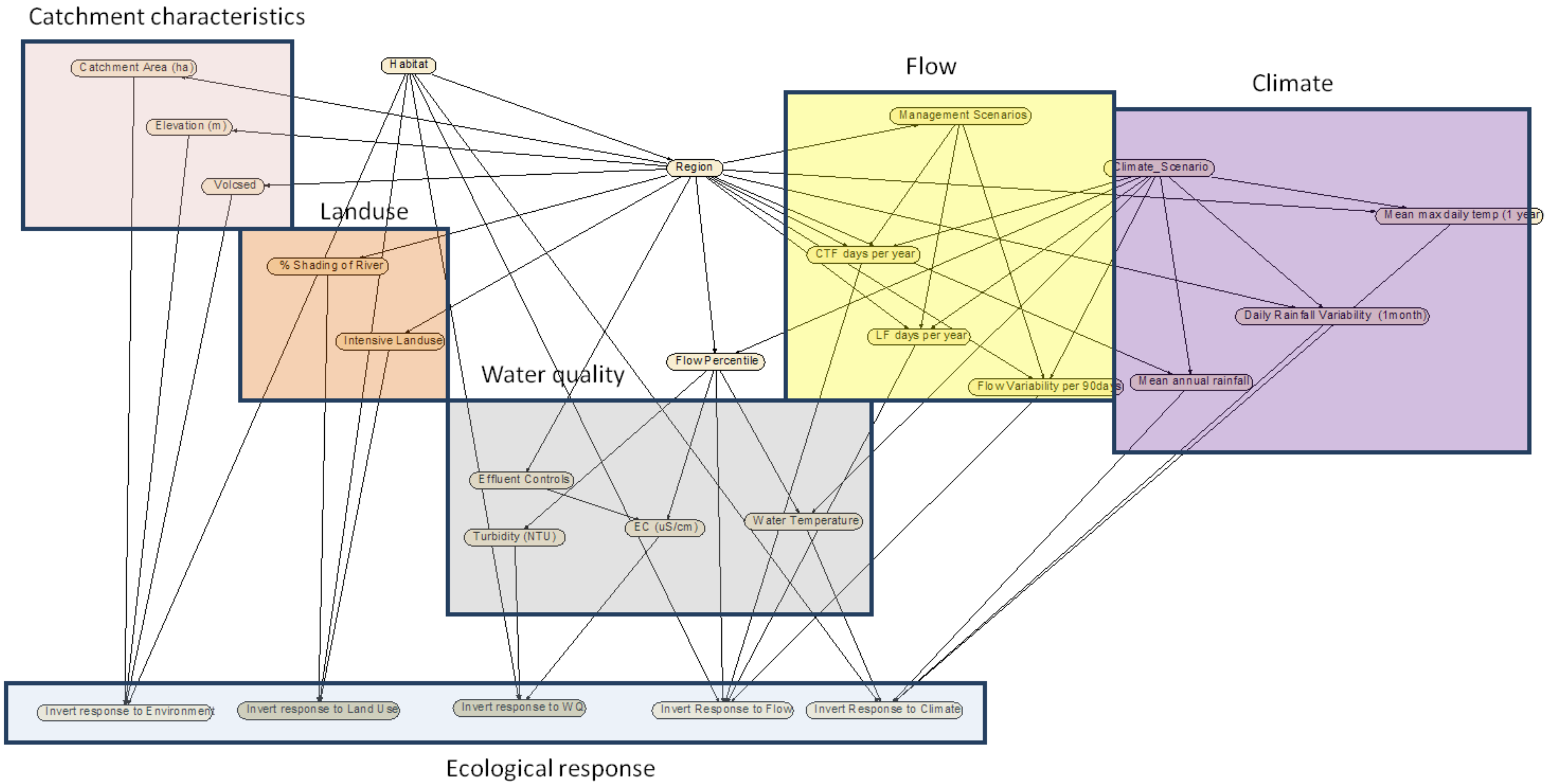


Figure 116. Example of the Upper Murrumbidgee Catchment modelling framework- invertebrate communities.

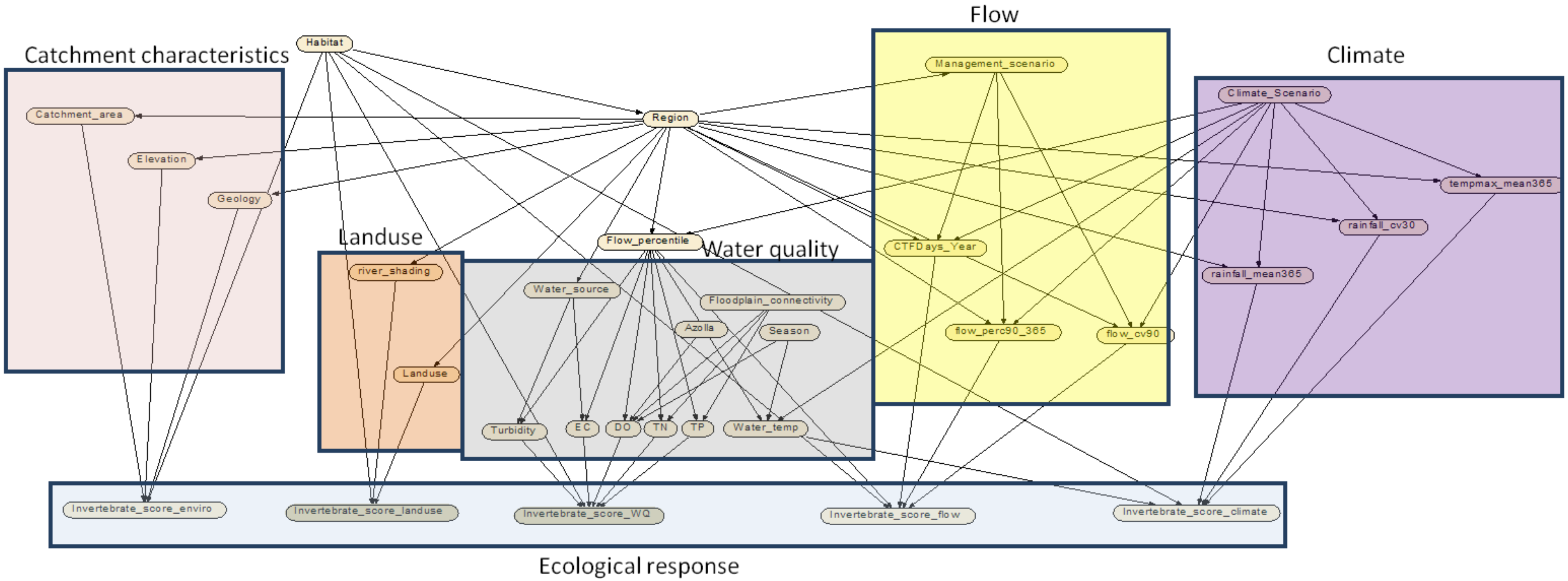


Figure 117. Modified modelling framework for the Goulburn Broken catchment- invertebrate communities

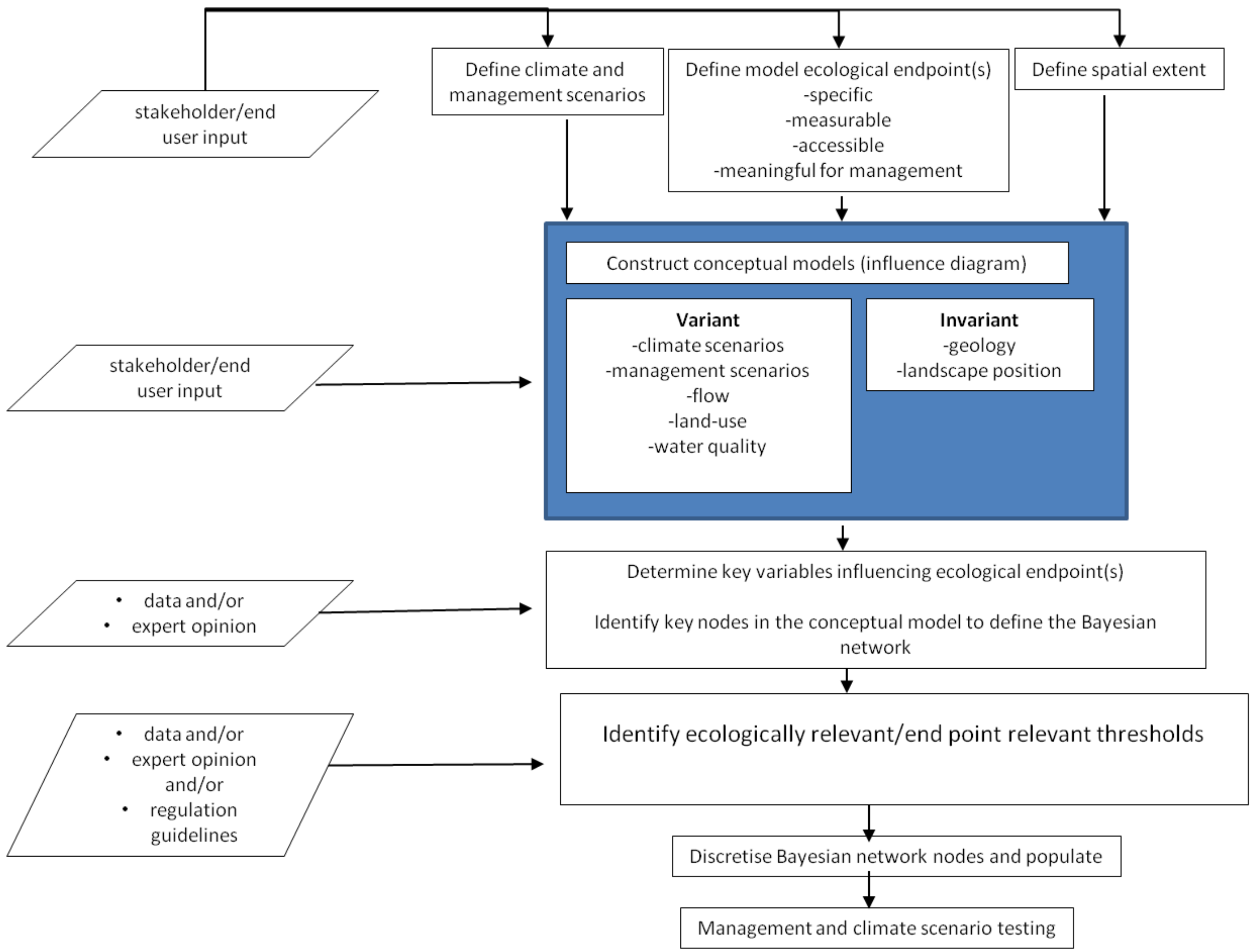


Figure 118. Modelling framework of the project to be transferred other regions.

8. KNOWLEDGE GAPS & FUTURE RESEARCH DIRECTIONS

We have made a number of significant advances, such as identifying thresholds, predicting effects of adaptation initiatives, and developing a comprehensive modelling framework for ecological response to climate change. In the process, we have identified a number of limitations and knowledge gaps. In this section we highlight limitations to methods used and key areas that should be prioritised for future research.

8.1 Improved hydrological prediction

Current methods for predicting variations in flow regimes in response to climate change rely on a scaling approach. Such methods are unable to capture potential long-term changes in seasonality, or the long-term fluctuations in the frequency and magnitude of extreme events that may arise as a consequence of changing climate (Francis & Hengeveld 1998; Jackson et al. 2001). As such, we do not have the models that enable us to develop plausible flow time series that can be used to predict the ecological consequences of such changes. The use of current models that predict variation in flow may limit the ability to represent realistic ecological response to climate change.

8.2 Limits of data sets

A key element of this project was to select predictor variables that were significantly related to the ecological responses of macroinvertebrate and fish communities. Several challenges that that were identified during this process remain unaddressed.

(i) Limits to using historical data sets for predictive modelling

Historical data sets for climatic variables (rainfall, temperature, flow) were used to provide information about ecological responses and thresholds. In using such data sets, there is the assumption that the range of historical conditions observed is sufficient to capture patterns that can be used to predict future conditions. However long one’s historical datasets, that assumption does not hold if the future climate widely differs from historical conditions (e.g. if there are major seasonal shifts or changes in the magnitude and frequency of extreme events). To better understand the ecological responses that might occur as a result of climate change, research needs to include experiments that take account of conditions outside of those experienced historically.

(ii) Spatial scale of predictor variables and model analysis

A further constraint was the availability of spatial and temporal data at consistent scales for use in predictive models. Climatic and hydrological variables differed in temporal scales; other variables (such as water quality, land use and habitat variables) varied in spatial scales. It is challenging to combine these variables into one working model: a particular example was combining small-scale response variables (e.g. the relative abundance of thermophobic taxa) with broad-scale predictor variables (e.g. land use).

8.3 Limits to methods

(i) Identifying the optimum number of predictor variables for BN use

For univariate responses (e.g. O/E scores or Thermophobic taxa relative abundance), the shape of the relationship between the ecological response and the predictor variable determines the best approach to estimating the threshold (Brenden et al. 2008). In our project, we applied Quantile Piecewise Linear regression as a general approach, because

most results showed a scattered relationship that varied across quantiles. However, for future work, different methods, which adjust better to the specific relationship between the predictor and response, should be investigated.

(ii) Threshold selection and scale

The selection of predictor variables and relevant thresholds for discretisation was a key element determining both the validity and strength of the Bayesian Networks (BNs) in predicting ecological response. Changes in thresholds may ultimately change ecological outcomes predicted by the models. One important consideration is that thresholds were produced taking into account all regions together. When ecological outputs were examined by region, some thresholds became inappropriate (for example air temperature in the Yass region, see Section 6). From a management perspective it could be more useful to explore these thresholds at regional scales. However, this will be dependent on appropriate data sets being available. This could be important when considering the transferability of this approach (see Section 7).

(iii) Impossible scenarios and uniform probabilities in BN models

The Bayesian Network models were developed to link climate scenarios and management adaptation alternatives with changes in water quality and quantity and relevant ecosystem responses. One issue with using a large number (>5) of input variables into the response node is the creation of "impossible" scenarios, or scenarios for which historical data do not exist. In these cases the BN prescribes uniform probabilities to the conditional probability tables, which in turn limits the ability of the model to predict outcomes under such conditions.

(iv) Inability to capture and model the occurrence of extreme events

The BN models were found to be effective in modelling relationships between environmental predictors and response variables using historical and modelled data. One limitation of BN models is that they are ineffective at modelling low probability events, such as extreme weather events, which are predicted to increase with changing climate. Although these events have low probability associated with them, their impact on the environment is likely to be significant in the long and short term.

8.4 Knowledge gaps and future research directions

During the project, much time was invested in methods for identifying thresholds for predictor variables. We believe we have made a significant contribution to identification and selection of thresholds, and we have been invited to prepare a paper for a respected journal on this work.

However, approaches and methods used for such calculations remain an emerging science and there are still uncertainties around how to apply thresholds in integrated modelling frameworks.

The selection of ecological response indicators was also a challenge. For example, choosing invertebrate taxa or ecological indices that efficiently capture responses to adaptation initiatives and climate change was difficult because of the wide range of responses observed. In the future the development of a standardised technique for dealing with these issues would be grand!

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