# Predicting the effects of water abstraction and land use intensification on gravel bed rivers

a Bayesian network approach

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# Executive summary

Regional Plan changes affecting large gravel-bed rivers are currently underway in several regions of New Zealand, and more are likely to begin within the next few years. These plan changes have been prompted and shaped by the New Zealand Government's package of freshwater reforms (MfE 2013) including the National Policy Statement for Freshwater Management (2014) (NPS-FM). Plan changes for large gravel-bed river catchments are highly complex, seeking good outcomes for multiple values through use of a wide variety of policy and management tools. Ministry for the Environment has commissioned NIWA to develop a tool and guidance document to support decision-making processes for gravel bed rivers and their catchments.

This report describes a Bayesian network that has been developed to support decision-making for large New Zealand gravel bed rivers where freshwater planning needs to balance increasing demands for irrigation water to support economic growth with in-stream ecological, social and cultural values. The Bayesian network shows the relationships between in-river attributes and environmental drivers, underlying catchment characteristics, and common management actions. It predicts, in general terms, the likely effects of increased water abstraction, land use intensification and dam construction on stream macroinvertebrate communities, river birds, trout biomass, periphyton (attached algae) and health risks from secondary contact recreation in a "generic" river catchment. The Bayesian network is available from the author or Ministry for the Environment as a single model that requires the purchase of Netica™ software (www.norsys.com) to run. It is also available as several discrete modules that relate to individual values and are small enough to run on a free downloadable version of the software.

The Bayesian network predicts that increased water abstraction and associated land use intensification without mitigation actions are likely to increase periphyton growth, reduce the Macroinvertebrate Community Index and decrease trout biomass in the river mainstem through increased dissolved nutrient concentrations, summer water temperatures and reduced density of grazing invertebrates. Dam construction may lead to reduction in densities of several river bird species. The risk of infection from waterborne diseases through secondary contact recreation increases if land use intensifies without use of on-farm mitigation measures to reduce runoff of E. coli.

This report describes the sources of information used to derive the structure of the Bayesian network, the "states" of the key variables (nodes) and the predictive relationships between them, including comments on the limitations and appropriate application of the network. Resources are also provided so that a user may develop the Bayesian network to predict outcomes for a specific river catchment. These resources include identifying models that can be used to quantify specific components of the river system and the cause-effect relationships between them, sources of information for locating site-specific data, and experts who can help to complete the inputs required for specific situations. The resources are identified in the relevant subsections of the Results, and listed in the Reference section.

In addition, a linkage diagram shows relationships between a wider range of values than is included in the predictive Bayesian network. The linkage diagram is a conceptual model only, as it does not quantify the relationships between variables. It serves as an additional form of guidance to indicate ways in which the predictive network could be developed further with the use of other models (e.g. economic models) and information sources.

# 1 Introduction

Regional Plan changes affecting large gravel-bed rivers are currently underway in several regions of New Zealand, and more are likely to begin within the next few years. These plan changes are occurring in response to a number of factors, in particular the New Zealand Government's package of freshwater reforms (MfE 2013) that includes the National Policy Statement for Freshwater Management (2014) (NPS-FM). The NPS-FM specifies that regional councils must "make or change regional plans to the extent needed to ensure the plans:

a) establish freshwater objectives in accordance with Policies CA1-CA4 and set freshwater quality limits for all freshwater management units in their regions…., and

b) establish methods (including rules) to avoid over-allocation."

With respect to water quantity, the NPS-FM specifies a similar requirement for the purpose of "setting environmental flows and/or levels for all freshwater management units in its region."

Through the National Objectives Framework (NOF), the NPS-FM identifies ecosystem health and human health for recreation as "compulsory national values" that must be maintained in regional plans. Within these two values it specifies condition bands for several attributes with the lowest band representing a national "bottom line" that regional plans must equal or exceed.

As a result of the NPS-FM, many regional councils are undertaking plan changes for large river catchments that involve setting limits on a variety of aspects of water quality, water quantity and river ecosystem health. Councils need to base these limits on robust science that provides evidence that ecological values will be maintained, while ensuring the limits are practically achievable and economically viable. This task is made all the more challenging by the increasing demands on freshwater for out-of-river use, particularly irrigation, while farming intensification increases the difficulty of maintaining water quality.

In addition, the processes by which regional plans are being made are radically different from the past. The Government's freshwater reforms anticipate greater recognition of a wide range of community values (social, cultural, economic and ecological) in developing regional plans. They also envision greater involvement by local communities in developing or changing regional plans, and encourage regional councils to adopt a collaborative planning process. This may involve the council establishing a "collaborative stakeholder group" (CSG), with members representing groups in the community that have majors interests in freshwaters, whose mandate is to provide the council with recommendations on freshwater quality and quantity. To deliver appropriate recommendations, CSGs must base their decisions on good science. However the science is often highly complex, as different components of a river system interact in complex ways such that a single decision has effects on multiple values. CSG members need decision-support tools that enable them to determine and compare the effects of different management or policy options on a wide range of values. Council staff involved in freshwater planning may also benefit from such tools, as increasingly they are required to provide transparency to their decisions.

# 2 Bayesian networks

Bayesian networks (BNs) are a tool particularly well-suited for supporting decisions on environmental resource management. Their strength in helping to resolve complex environmental problems lies in their ability to incorporate the effects of multiple influences on a wide range of values (economic, social, cultural and ecological) and to include information from a variety of sources, including empirical data, scientific theory, various types of models and expert opinion (Quinn et al. 2013).

BNs represent the components of a river system in the form of “nodes” (shown as boxes or circles), with the cause-effect relationships (linkages) between them shown by arrows. Each node in a BN has two or more possible states, and the BN represents outcomes as a probability distribution between the possible states. The effect of the causative (parent) nodes on another (child) node is quantified in a “conditional probability table” (CPT). The CPT shows the probability of each state in the child node, given each combination of states in the parent nodes. In a causative chain with multiple linkages, a change in the state of the top node propagates through the entire network to all of the “descendant” nodes. In the context of resource management, the top nodes are typically management decisions and the final descendant nodes represent components of the system, such as species, aspects of the environment, economic indicators, etc., that are valued by the community. In this way, BNs are able to condense complex scientific information into an intuitive form that is appropriate for guiding stakeholder deliberations and supporting decision-making.

BNs allow users to visualise the interacting components of a river system and run “what-if” scenarios with different management options. Thus they are useful in group situations where a shared understanding of the system is important, and where all members need to see the effects of different management options. BNs provide transparency to decision-making, whether those decisions are made by councils or stakeholder groups.

BNs are not intended to replace mechanistic models. Rather, they summarise the key outputs of mechanistic models and integrate them with other sources of information. Because they describe outputs in terms of probabilities, BNs can incorporate highly precise forms of knowledge with others that are inherently less precise or subject to a range of influences outside the BN.

# 3 Scope of this report

The ecosystem considered in this report is a large New Zealand river ($6^{th}$ order or larger) with a mobile gravel bed. The river may be single thread or braided, but includes extensive gravel bars which naturally would migrate with high flow events. Although I acknowledge that most of the length of a large river network is made up of small and medium-sized tributaries, here I represent primarily the effects on the mainstem. I have also chosen not to include effects specific to river mouths or estuaries.

The scenario modelled in this Bayesian network is an increase in water abstraction resulting in a significant decrease in the river's flow. Since water abstractions are typically used for irrigation, and therefore likely to result in land use intensification in the catchment, the effects of land use intensification on the selected instream values are also included. Increasing water supply for irrigation is sometimes accompanied by construction of a dam on the river mainstem or one of the larger tributaries. I have attempted to incorporate the most significant impacts of damming, which include changes to flow regime (especially frequency, magnitude and timing of flood flows), reduction in sediment supply, alterations to water temperature, creating barriers to migration of fish and invertebrates. The river reach represented by this BN is downstream of any such dam.

The focus of this BN is on ecological values (invertebrates, periphyton and river birds), recreational values (trout fishing and secondary contact recreation) and aesthetic values (periphyton). Native fish could not be included within the timeframe. I acknowledge the importance of other values, such as Maori cultural values (including mahinga kai, taonga species and the mauri of the river), uses such as drinking water and industrial processing, specific recreational values, such as whitewater kayaking or rafting, economic and tourism values. All of these are important, and could be incorporated into future extensions of the BN.

Since this BN is a generic model, not representing a specific location, it can only include information that is applicable to all rivers of the type described above. To apply the BN to a specific river requires determining values for many of the nodes. Some of these values are available from State of Environment monitoring databases (held by regional councils or available on the Land and Water Aotearoa (LAWA) website www.lawa.org.nz), or can be calculated from these databases. Nodes requiring the user to input values or set states are shown in light blue. Some nodes and some of the cause-effect relationships between nodes can only be determined by the use of other models, with local data providing input parameters. Nodes requiring such models are shown in green and linkages requiring models are labelled with text boxes. This report gives references to the information sources and models that are required to fill these gaps.

In some parts of the BN (e.g., effects on trout size and abundance), many quantitative studies and models are available, and outcomes can be determined relatively precisely. In other parts (e.g., effects on river birds), the effects of influential factors are known only in general terms and outcomes can only be known in general terms.

# 4 Methods

The Bayesian network incorporating the components of a gravel bed river system relevant to the values and development scenario considered here, and the relationships between them, was created in two forms. The first form is a predictive Bayesian network. In this network, the states of each node are defined, and the cause-effect relationships between them are quantified. This network can be used to predict the state of outcome variables given the state of user-input variables. Because quantifying each cause-effect relationship requires a large amount of information, this network represents a subset of the second form. The second form is a linkage diagram, also known as an influence diagram or knowledge network. This diagram shows the different variables and the relationships between them. However, the states are not defined (or not shown) and the relationships between them are not quantified. Therefore, this diagram acts only as a conceptual model, helping the user understand the different components of a river system and which ones interact, but not able to provide predictions based on management or policy decisions. The linkage diagram has all the same nodes and linkages as the predictive Bayesian network, however it places these in the wider context of a river system by including outcome nodes for Ecosystem health and mahinga kai, native fish, suitability for swimming, natural character and gross domestic product. Some of the causative variables of these nodes are also additional to those included in the predictive model.

The Bayesian network was developed using the software package Netica 4.16 (Norsys, 2005). The structure, node definitions, states and cause-effect relationships of the BN were developed by drawing on a variety of information sources. The specific sources are cited in the relevant subsections of the Results section. Generally, they included published papers, reports, expert evidence related to recent Environment Court cases, large datasets from regional council State of Environment and NIWA National Rivers Water Quality Network monitoring, and expert opinion from scientists at NIWA, Cawthron Institute, universities and regional councils.

The method for developing each node and its probabilistic relationships with adjacent nodes is described in the Results section, but some general procedures are described here.

To make this BN as relevant as possible to planning under the NPS-FM, variables specified in the National Objectives Framework were assigned states that correspond to the condition bands in the NOF.

Parent nodes include only the most influential drivers, those that are likely to change under the scenarios being considered here and those whose relationship with the child node are known with at least some degree of precision. Therefore this Bayesian network does not represent all the ecological relationships that influence a particular variable or organism, and it may or may not include the same set of factors that have been shown to influence that variable in another situation or at another scale.

Nodes are colour-coded to aid usability and interpretation. Yellow nodes are those representing key values. These include periphyton (attached algae), Macroinvertebrate Community Index (a measure of stream ecological health based on the invertebrate assemblage inhabiting the river bed), trout biomass (a combination of fish abundance and size), abundance and richness (diversity) of river birds, risk of infection from secondary contact recreation, suitability for swimming, natural character, Gross Domestic Product, native fish and River Ecosystem Health and mahinga kai. Light blue nodes are variables requiring the user to set values or states. In the predictive Bayesian network, these include management/policy decisions (e.g., disturbance of bird habitat by vehicles), values drawn from monitoring data (e.g., current state of water clarity), or specifications of development scenarios (e.g., flood frequency and magnitude). In the linkage diagram, management/policy decisions are distinguished from other user settings by purple colour. Green nodes are values that can only be calculated with the use of other models, such as RHYHABSIM for % protection of trout habitat, or CLUES for increases in nutrient input with change in land use. Nodes in the default colour (light brown) are variables whose states are determined by Netica based on the values entered in the user setting nodes. Note, however, that a user may set the value of any node if it is known. The relationships between nodes are such that setting the state of a child node affects the state of a parent node as well as the reverse (for this reason, the state of a parent and child node cannot both be set to values that are incompatible with each other given the relationship defined between them).

# 5 Results: predictive Bayesian network

The predictive Bayesian network, which is provided along with this report, is shown in Fig. 1. This figure shows the network nodes with their states, but doesn't include any settings for those states, so readers should not give regard to the probabilities shown in this figure. Running scenarios with the Bayesian network requires the Bayesian networks themselves, which are available from the author or from Ministry for the Environment. It also requires Netica software. A free downloadable version of Netica is available from the Norsys website (https://www.norsys.com/download.html), but this version can only run networks consisting of 15 nodes or less. To make this BN accessible for use on the free version, it has been broken it into parts corresponding to the five values (periphyton, MCI, trout biomass, river birds and secondary contact recreation). Used individually, these parts lose one key benefit of Bayesian networks, that is to show the effects of single decisions on multiple values.

## A note for using this Bayesian network in specific catchments:

Bayesian networks are intended to reflect our best estimate regarding the state of different components in a system, based on what we know of the factors affecting them. They are intended to be updated with more precise knowledge as it becomes available. Since this Bayesian network was developed for a generic large gravel-bed river, when applied to a specific river catchment it may be appropriate to add or remove some nodes and/or modify the probability tables according to local knowledge of that catchment. Users are encouraged to modify the network, using what we have presented here as a starting point and a framework for thinking.

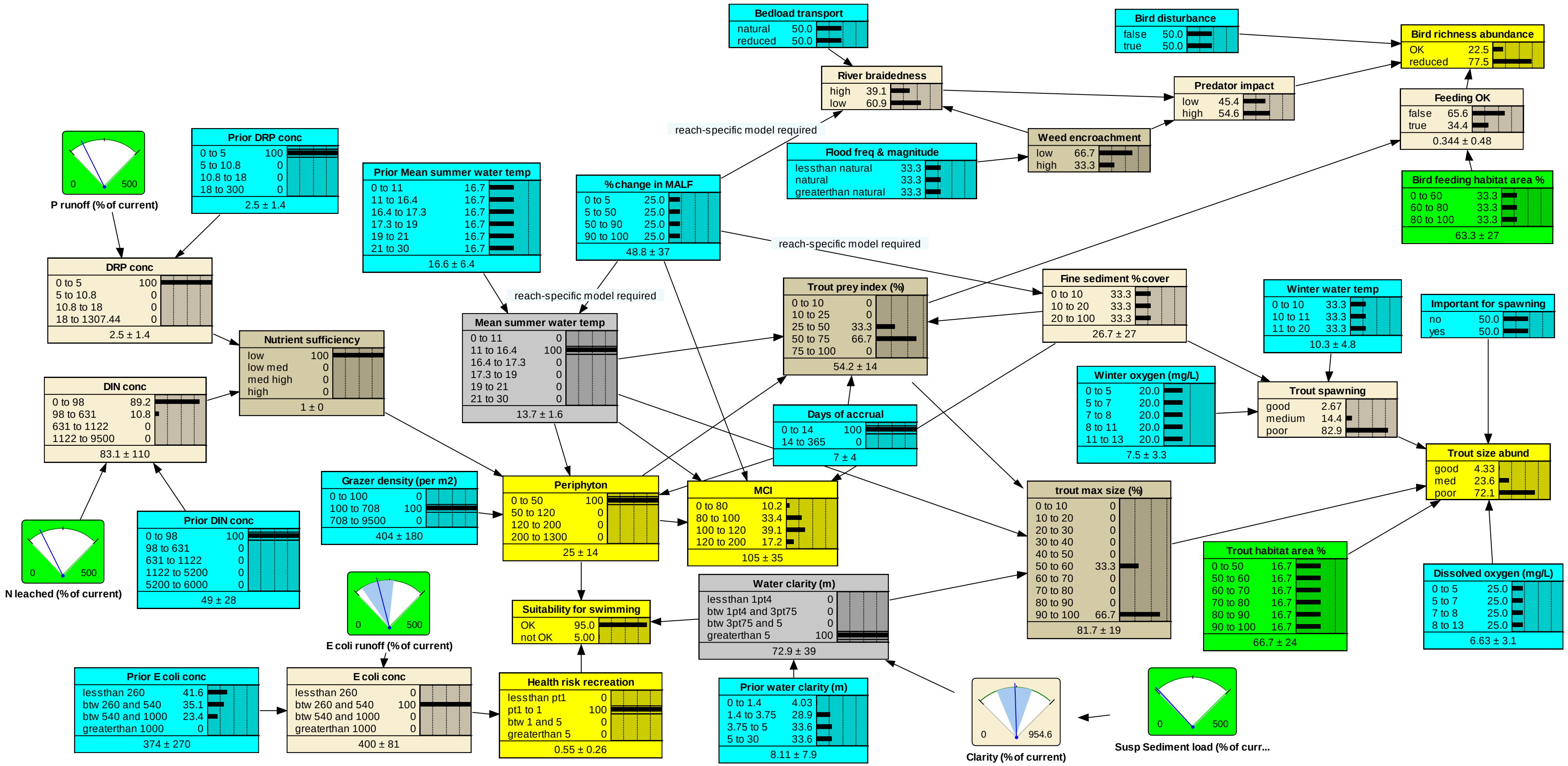


**Figure 1:** Bayesian network of a river catchment showing the factors affecting a number of ecological values under a scenario of increasing abstraction, dam construction and/or land use intensification. Colour codes are: yellow for outcome nodes, blue for user settings from data or management decisions, green for outputs of other models. States are shown but not the settings for those states, so the probabilities should not be interpreted.

However, if care is taken to ensure the user settings in each of the parts is identical to those in others, this benefit can largely be retained. Note, however, that the full version of Netica (available from https://www.norsys.com/order.html) costs only US$585 for a commercial licence and US$285 for an educational licence.

The following subsections describe each of the five values, including their definition, states, and the structure, node definition, and probabilistic relationships of all the factors influencing them. The assumptions and information sources used to develop the BN around each value are described. Knowing these is essential in order to know how the BN may and may not be used, and how to adapt it to particular river catchments. Some of the references may also allow the BN to be extended to include other values or causative factors that may be important in specific situations.

## 5.1 Periphyton biomass

### 5.1.1 Node description and states

*Node name:* Periphyton. *Units:* mg Chlorophyll a /$m^2$

Periphyton biomass is one of the nodes in this Bayesian Network that is specified as an attribute of Ecosystem Health in the National Objectives Framework. NOF condition bands for periphyton biomass as mg of chlorophyll *a* per $m^2$ were used to define the states for the periphyton node. Periphyton is classified in a particular state provided that its biomass does not exceed the upper bound for that state in more than 8% of monthly samples.

### 5.1.2 Node parents

Periphyton biomass results from a balance between its rate of growth and the frequency of biomass loss events (Snelder et al. 2014; Matheson et al. 2015). Rate of growth is controlled primarily by nutrient supply, light, temperature and grazing macroinvertebrates, whereas biomass loss is primarily due to high flow events that scour periphyton from the substrate. Here I assume that the amount of light reaching periphyton will not change greatly with the development scenarios under consideration, therefore, the drivers of periphyton biomass are temperature, nutrient supply, density of macroinvertebrate grazers and frequency of high flow events.

The conditional probability table for periphyton is quite long (6 rows) because each of the four parent states has between 2 and 4 states. Therefore, instead of showing the entire table, I describe the relationship between each parent and periphyton, and the method for combining these relationships.

#### Dissolved nutrients

*Node names:* DIN conc, DRP conc. *Units:* mg/$m^3$

Periphyton requires dissolved nutrients from the surrounding water, in particular dissolved inorganic nitrogen (DIN; consisting of ammonia, nitrate and nitrite) and dissolved reactive phosphorus (DRP), in order to grow. Therefore, provided other resources are not limiting, periphyton biomass is strongly correlated with the concentration of dissolved nutrients in river water (Snelder et al. 2014). Periphyton typically requires DIN and DRP in a concentration ratio of 15:1, therefore where the ratio is higher than this, growth is limited by availability of DRP and when lower than this it is limited by availability of DIN. In the summer focused dataset of Matheson et al. (2015), 85% of samples had periphyton biomass equalling the boundaries between NOF condition bands A-B, B-C and C-D at annual mean DIN concentrations of 98, 631 and 1122 mg/m$^3$, respectively. The 85%iles of the Matheson et al. (2015) summer-focused data were assumed to correspond approximately to the permissible 8% exceedance level for the NOF bands, because including winter data, when periphyton cover is typically low, would likely increase the 85%ile to close to 92%. In addition, 85% of samples had periphyton biomass corresponding to B-C and C-D boundaries at annual mean DRP concentrations of 10.3 and 18 mg/m$^3$, respectively. I added an additional DRP concentration of 5 mg/m$^3$ to approximate a likely boundary between A and B bands. In Matheson et al.'s (2015) dataset, the proportion of samples with periphyton in the higher biomass categories increased with increasing concentration of DIN and DRP (Tables 1 and 2).

**Table 1: Frequency of periphyton biomass in NOF categories with different concentrations of annual mean dissolved inorganic nitrogen (DIN) in summer- focused dataset of Matheson et al. (2015).**

| | Chl-a <50 mg/m$^2$ | Chl-a 50-120 mg/m$^2$ | Chl-a 120-200 mg/m$^2$ | Chl-a >200 mg/m$^2$ |
|---|---|---|---|---|
| DIN low (<98 mg/m$^3$) | 90% | 7% | 2% | 1% |
| Low-med (98-631 mg/m$^3$) | 65% | 23% | 6% | 6% |
| Med-high (631-1122 mg/m$^3$) | 53% | 22% | 10% | 14% |
| High (>1122 mg/m$^3$) | 61% | 21% | 7% | 12% |

**Table 2: Frequency of periphyton biomass in NOF categories with different concentrations of annual mean dissolved reactive phosphorus (DRP) in summer- focused dataset of Matheson et al. (2015).**

| | Chl-a <50 mg/m$^2$ | Chl-a 50-120 mg/m$^2$ | Chl-a 120-200 mg/m$^2$ | Chl-a >200 mg/m$^2$ |
|---|---|---|---|---|
| DRP low (<5 mg/m$^3$) | 76% | 16% | 4% | 5% |
| Low-med (5-10.8 mg/m$^3$) | 73% | 16% | 6% | 5% |
| Med-high (10.8-18 mg/m$^3$) | 53% | 24% | 12% | 11% |
| High (>18 mg/m$^3$) | 57% | 22% | 11% | 11% |

DRP and DIN were combined into an intermediate node called "nutrient sufficiency" which recognises that the nutrient that is in least supply will limit periphyton growth. Therefore, nutrient sufficiency takes on the state that is the lower of DIN and DRP.

*Changes in nutrient concentrations with abstraction and land use intensification*

If increases in water abstraction lead to land use intensification in the catchment, then concentrations of DIN and DRP are expected to increase unless compensatory mitigation strategies are employed. Determining the amount of increase is beyond the scope of this Bayesian network, and other models have been developed for this purpose, in particular, CLUES (Catchment Land Use for Environmental Sustainability; Semadeni-Davies et al. 2011; available for download at ftp://ftp.niwa.co.nz/clues/). CLUES provides annual average loads of Total Nitrogen, Total Phosphorus, *E. coli* and sediment in streams based on scenarios that specify land use, stock intensification and mitigation practices (such as riparian planting, treatment wetlands, etc.). Users can create land use scenarios by modifying maps of current land-use provided with CLUES or by importing new land-use maps. Land-use types that can be analysed include several types of sheep, beef, dairy, and deer farming along with arable, horticulture, and forestry. CLUES uses nutrient leaching rates calculated by a simplified version of OVERSEER® and scales them up to sub-catchment and catchment-integrated nutrient loads. Although the outputs of CLUES are loads rather than in-river concentrations, such concentrations can be estimated with reasonable accuracy based on known concentrations of nutrients prior to land use change (i.e., current state), and assuming that a proportional increase in catchment load with land use change results in the same proportional increase in concentration. This approach has been taken by the developers of CLUES in several reports (e.g., Semadeni-Davies and Elliott 2011, 2012). Current-state nutrient concentrations can be obtained from regional council State of Environment monitoring databases, which are now publicly available on the Land and Water Aotearoa (LAWA) website (www.lawa.org.nz). This Bayesian network includes nodes where users can enter prior (current state) DRP and DIN concentrations and the proportional increase in DRP and DIN runoff predicted by CLUES.

Nutrient concentrations also change along the course of a river as a result of attenuation as well as inputs from the catchment. A variety of plants and microbes may reduce nutrients via assimilation or denitrification, but the dominant process in large gravel-bed rivers is likely to be assimilation by periphyton (Chapra et al. 2014). This process creates a feedback loop in our model of nutrients and periphyton growth. Unfortunately Bayesian networks are not able to incorporate feedbacks, so I cannot include this process. This process also highlights that the nutrient concentrations in a river vary with location, typically decreasing with distance from source.

Increased water abstraction may also lead directly to changes in river nutrient concentrations. If water is removed from reaches upstream of areas of developed land, then less water will remain in the river to dilute inflows bringing elevated nutrient concentrations. In this case, increasing abstraction will result in higher nutrient concentrations in lower reaches. However, if water is abstracted from reaches already enriched in nutrients, then increasing abstraction may not cause any change in nutrient concentrations. Because the effect of abstraction on nutrient concentrations depends on the location of abstractions relative to nutrient sources, I have not included this effect in the Bayesian network. If it can be determined outside of the Bayesian network, the user may enter the calculated concentrations of DIN and DRP in the corresponding nodes of the network.

### Summer water temperature

*Node name:* Mean summer water temp. *Units:* °C.

*Definition:* average water temperature during mid-summer months January-February

Many metabolic activities in living organisms proceed more rapidly with increasing temperature, thus periphyton growth rates increase with water temperature (Snelder et al. 2014). Matheson et al.'s (2015) dataset includes several measures of water temperature, at different numbers of months prior to the date of periphyton sampling, as well as temperatures averaged over these time periods. Matheson et al. (2015) found that the strongest correlation between periphyton biomass and water temperature were for water temperatures averaged over 12 months prior to sampling. Logically, however, water temperatures closer to the time of sampling should have a stronger influence on periphyton biomass, since biomass is "re-set" several times per year by scouring floods. Since periphyton was typically sampled in late summer (timed with the period of maximum growth), I used water temperature averaged over the two months prior to sampling as an estimate of summer water temperature. This also made the water temperature node for periphyton consistent with that for macroinvertebrates and trout. In Matheson et al.'s (2015) dataset, the proportion of samples with periphyton in the higher biomass categories increased with increasing summer water temperature (Table 3).

**Table 3: Frequency of periphyton biomass in NOF categories with different summer water temperatures in Matheson et al.'s (2015) dataset.**

| | Chl-a <50 mg/m² | Chl-a 50-120 mg/m² | Chl-a 120-200 mg/m² | Chl-a >200 mg/m² |
|---|---|---|---|---|
| Temp <11 °C | 89% | 3% | 1% | 7% |
| <11-16.4 °C | 83% | 5% | 4% | 7% |
| 16.4-21 °C | 79% | 8% | 5% | 9% |
| >21 °C | 62% | 14% | 7% | 17% |

*Changes in water temperature with abstraction*

Removal of water from a river channel typically results in reduced depth and flow velocity, increased rates of heating and cooling, and therefore increased daily maximum temperature at a given location (Rutherford 2001). Although these effects are known in general terms, predicting the actual increase in water temperature in a given reach depends on specifying the reduction in river flow as well as a number of characteristics of the river reach, including local climate, channel morphology, and inflows from tributaries and groundwater. A physically-based model such as STREAMLINE (Rutherford 2001) can predict changes in water temperature with flow reduction based on values of these parameters. In this Bayesian network I indicate the role of such a model in determining the relationship between % change in MALF (mean annual low flow) and mean summer water temperature.

## Density of macroinvertebrate grazers

*Node name:* Grazer density. *Units:* per $m^2$

A number of aquatic macroinvertebrates graze on periphyton, and previous studies (Jacoby 1985, Welch et al. 1992, 2000, Holomuzki et al. 2006) have shown that high densities of macroinvertebrate grazers are capable of reducing the accrual rate of periphyton biomass. In the dataset of Matheson et al. (2015), 85% of samples had periphyton biomass equalling the boundaries between NOF condition bands B-C and C-D when the densities of selected macroinvertebrate grazers were 100 and 708 individuals per $m^2$, respectively. The proportion of samples with periphyton in the higher biomass categories decreased with increasing densities of macroinvertebrate grazers (Table 4).

**Table 4: Frequency of periphyton biomass in NOF categories with different densities of selected macroinvertebrate grazers Matheson et al.'s (2015) dataset.**

| | Chl-a <50 mg/m$^2$ | Chl-a 50-120 mg/m$^2$ | Chl-a 120-200 mg/m$^2$ | Chl-a >200 mg/m$^2$ |
|---|---|---|---|---|
| Grazers <100 m$^{-2}$ | 39% | 33% | 10% | 18% |
| 100 – 708 m$^2$ | 59% | 19% | 14% | 9% |
| >708 m$^{-2}$ | 66% | 21% | 8% | 4% |

Although the density of macroinvertebrate grazers is clearly linked to other macroinvertebrate nodes (Macroinvertebrate Community Index and Trout Prey Index, which is a measure of macroinvertebrate density), I have left Grazer density as a separate node to be entered by the user. The reasons are that a) this is an absolute measure (number per m$^2$), whereas Trout Prey Index is relative to reference, b) MCI and Trout Prey Index are influenced by periphyton, so linking these with Grazer density would create a circularity which is prohibited in Bayesian networks; and c) Grazer density includes only a subset of macroinvertebrate taxa that are known to graze on periphyton. The criteria for selecting which taxa to include in calculations of Grazer density are described in Matheson et al. (2015).

## Days of accrual

*Node name:* Days of accrual. *Units:* days

Clausen and Biggs (1997) and Matheson et al. (2015) determined that the most relevant index of flood frequency is FRE3, the number times per year that river flow equals or exceeds 3x the median flow. Using a dataset consisting of State of Environment monitoring from several regional councils and National River Water Quality Network monitoring by NIWA, Matheson et al. (2015) described a quantile regression relationship between periphyton cover and days of accrual_3 (DA_3). DA_3 is the number of days since an event 3x the median flow, which, averaged over time, is the inverse of FRE3. The value of DA_3 that most clearly separated low from high periphyton biomass was about 14 days. The frequencies of periphyton biomass in the four NOF condition bands above and below DA_3=14 were determined using Matheson et al.'s dataset (Table 5).

**Table 5: Frequency of periphyton biomass in NOF categories with greater or less than 14 days since a flood 3x median flow. Numbers in parentheses represent change relative to DA_3 <14.**

| | Chl-a <50 mg/m$^2$ | Chl-a 50-120mg/m$^2$ | Chl-a 120-200 mg/m$^2$ | Chl-a >200 mg/m$^2$ |
|---|---|---|---|---|
| DA_3 <14 | 85% | 7% | 3% | 4% |
| DA_3 >14 | 70% (-15%) | 15% (+8%) | 5% (+2%) | 7% (+3%) |

Days of accrual is clearly related to the node Flood frequency and magnitude. However, I have kept the two nodes separate for two reasons: a) Flood frequency and magnitude includes magnitude as well as frequency; and b) Days of accrual is an absolute measure whereas Flood frequency is specified in only very general terms.

*Changes in days of accrual with catchment development*

Abstractions that reduce some flood flows from >3x median flow to <3x median flow would reduce Days of accrual. However, a much more significant effect is if development of water resources included a dam on the river mainstem or a large tributary. In this case, the frequency of floods flows would be entirely controlled immediately and for a considerable distance downstream of the dam (depending on inflows from major tributaries). In this case, the user can set a value for Days of accrual if the flushing flow regime of the dam is known.

### 5.1.3 Combining parents of periphyton biomass

Netica is able to learn predictive cause-effect relationships between nodes by applying "Bayesian inference" to a dataset that includes the child and one or more parent nodes. I presented Netica with a dataset from Matheson et al. (2015) that included periphyton biomass, summer water temperature, nutrient sufficiency and macroinvertebrate grazer densities. For the learning process I directed the cause-effect arrows from periphyton to the causative variables to prevent Netica from inferring interactions between the causative variables. Such interactions complicate the Bayesian inference process and were leading to nonsensical results in preliminary trials. I assumed that interactions among the causative variables were minor compared to the main effect of each variable on periphyton. After the initial learning process I then reversed the cause-effect arrows one by one, deleting any cross-links among causative variables formed during the reversal process, so that all arrows were directed from the causative variables to the periphyton node. The probabilistic relationships described by the arrows are symmetrical, therefore reversing them should not alter the effect of each variable on the other (Netica v 4.16 help file).

The dataset linking days of accrual with periphyton biomass was separate to that for the other variables. Therefore, the effect of days of accrual was incorporated by calculating the change in probability of periphyton being in each biomass category with a change in days of accrual from <14 to >14. These changes in probability (Table 5 values in parentheses) were applied to the conditional probability table for periphyton biomass for days of accrual >14.

### 5.1.4 Effects of two development scenarios on periphyton biomass

The possible effects of water resource development on periphyton growth can be seen by comparing the following screen shots of the periphyton part of the BN. The first (Fig. 2), represents a river catchment with little or no development of its water resources. The river has relatively low nutrient concentrations, relatively low summer water temperature since most of the natural river flow remains in the channel. The natural flow regime of the river provides a flood event >3x median flow on average every 2 weeks, and the macroinvertebrate community is healthy, with an average density of grazing taxa. In this situation, periphyton biomass is most likely (70%) to be low (0-50 mg/m$^2$).

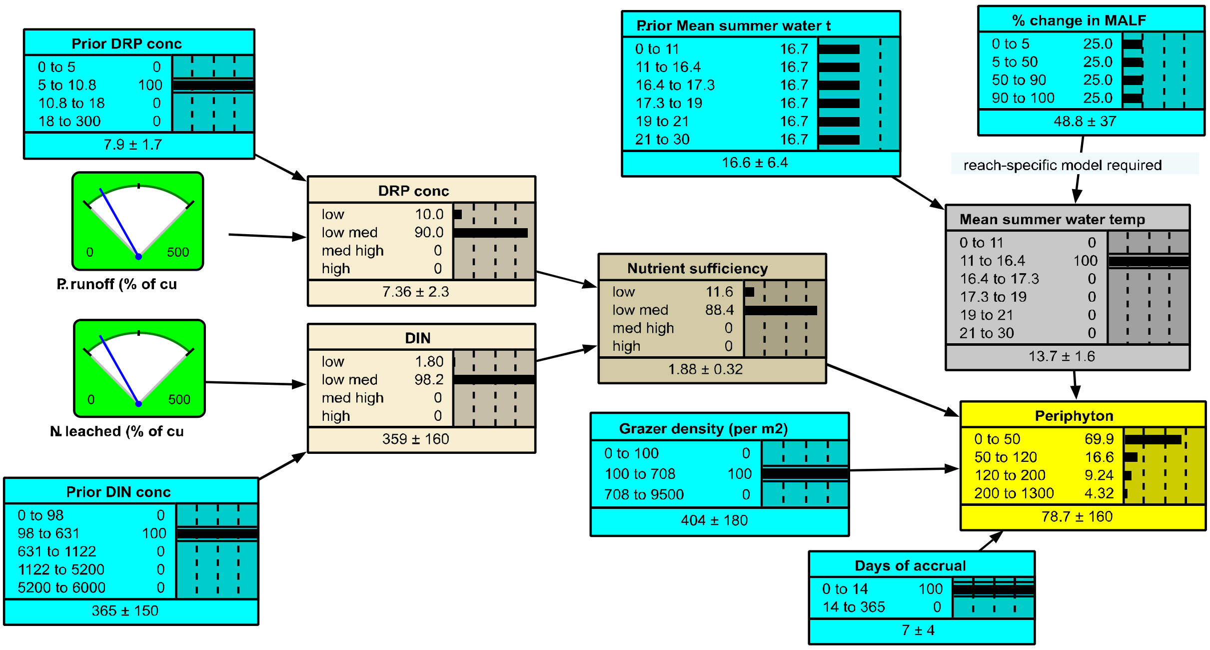


**Figure 2: Predicted probabilities of periphyton biomass in a river catchment with little or no water abstraction and intensive land use.**

The second screen shot (Fig. 3) represents a scenario where a significant proportion of the river's flow has been abstracted for irrigation. As a result the summer water temperature has risen by 3 °C. The water has been used to increase the land area in intensive farming, with a result that dissolved inorganic nitrogen and dissolved reactive phosphorus concentrations in the river have increased by a factor of 4-5x. Associated effects of land use intensification have reduced the densities of macroinvertebrate grazers, releasing periphyton from much of the natural grazing pressure. As a result of these changes, periphyton growth rates have increased, and periphyton biomass has almost equal probabilities of being in low (0-50 mg/m$^2$), medium-high (120-200 mg/m$^2$) or high (>200 mg/m$^2$) categories.

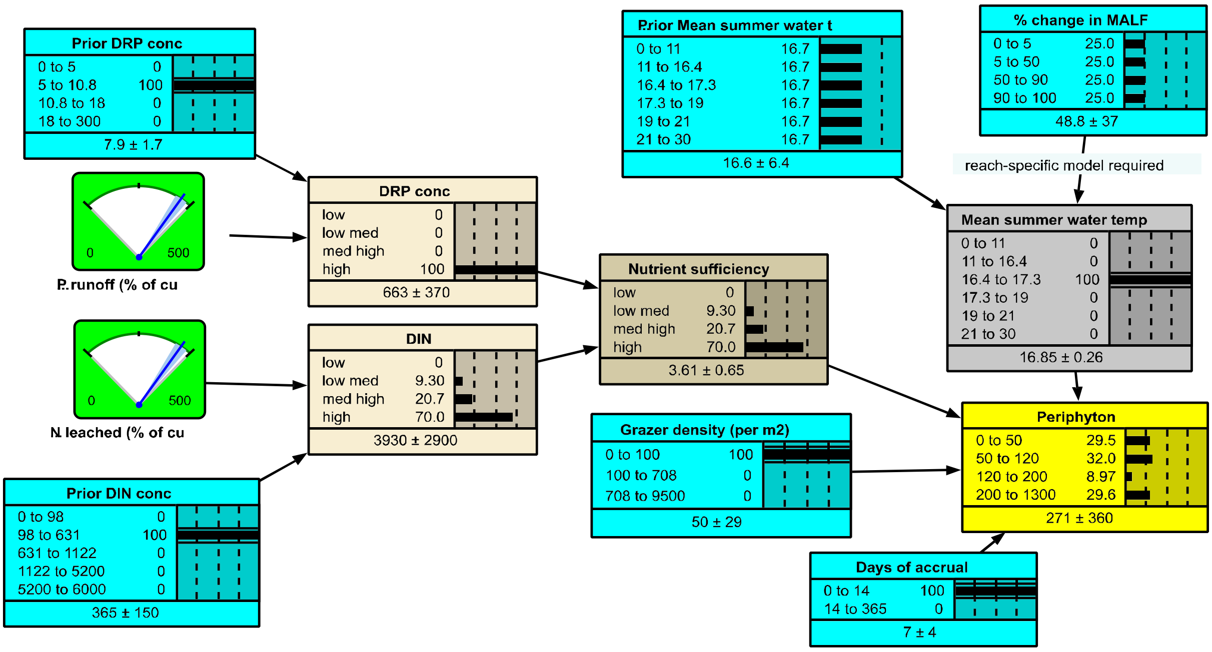


**Figure 3:** **Predicted probabilities of periphyton biomass in a river catchment with significant water abstraction and associated high intensive land use.**

The third screen shot (Fig. 4) represents the same scenario as Fig. 3, except that irrigation water is provided by damming the mainstem of the river. In this scenario, the changes to nutrients, grazing invertebrates and water temperatures are the same, but in addition, days of accrual increase as most floods >3x median flow are prevented by the dam. As a result, periphyton biomass is 15% less likely to be in the low (0-50 mg/m$^2$) category.

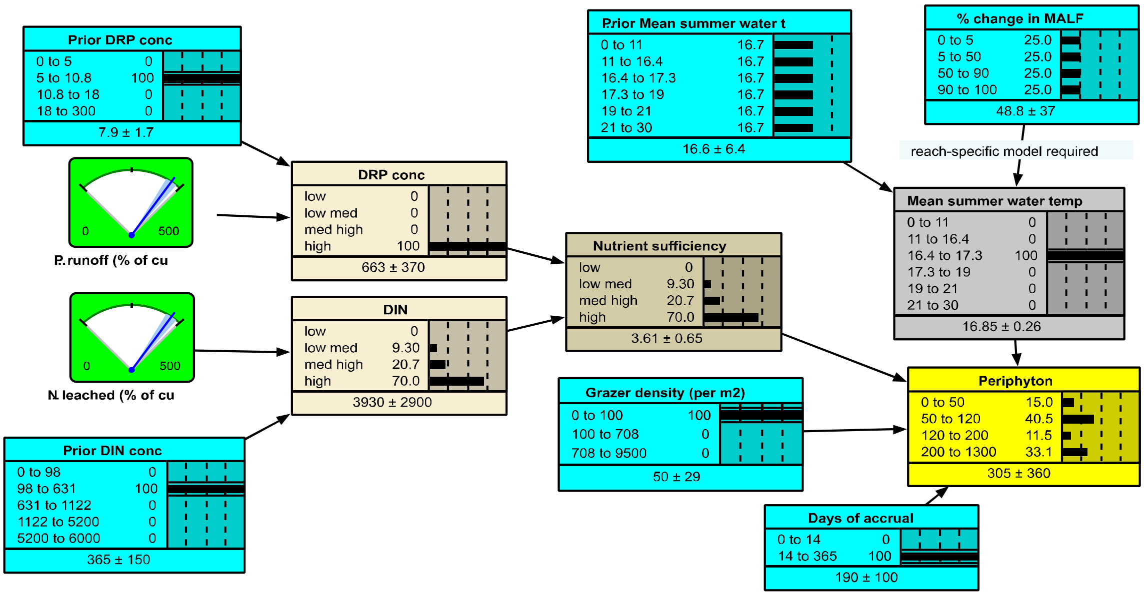


**Figure 4:** **Probabilities of periphyton biomass in a river catchment with significant water abstraction, a dam on the mainstem and high intensive land use.**

## 5.2 Macroinvertebrate Community Index

### 5.2.1 Node description and states

*Node name:* MCI. *Units:* MCI units

The condition of the macroinvertebrate community is one of the main indicators used internationally and in New Zealand to assess the overall ecological health of a stream or river (Boothroyd and Stark 2000). The Macroinvertebrate Community Index (MCI; Stark and Maxted 2007) is one of the main indices used by regional councils across New Zealand to measure the health of the macroinvertebrate community (Davies-Colley et al. 2012).

The states chosen for MCI in this Bayesian network correspond to the condition bands for Excellent, Good, Fair and Poor ecological health defined by Stark and Maxted (2007).

### 5.2.2 Node parents

MCI was originally developed to assess the impacts of organic pollution on rural streams, however, it has been shown to be sensitive to a range of other stressors (e.g., Collier 2008). Because it responds to stressors associated with land use intensification it was considered appropriate to use as a descriptor of macroinvertebrate community condition in this Bayesian network. In addition, because of its widespread use across New Zealand, several studies and datasets were available to determine the key drivers of change in MCI and to quantify the cause-effect relationships between them.

MCI is not as sensitive to changes in flow as the new LIFE index currently under development (Greenwood et al. in press), which is designed to respond to changes in flow. Indeed Greenwood et al. (in press) found MCI was not correlated with water velocity. However, MCI values have been shown to be influenced by large (~90%) flow reductions in pristine rivers, though not at agriculturally impacted sites (Death et al. 2009). In addition, MCI is affected by a variety of factors that are in turn affected by changes in flow regime (e.g., periphyton, which increases with decreasing frequency of floods).

The main factors determining MCI (and/or its quantitative variant QMCI) have been described in several recent publications and reports, e.g., Clapcott et al. (2013); Booker et al. (2015); Death et al. (2015). The causative factors described in these publications informed our choice of the primary factors influencing MCI. However, whereas the first two of these studies aimed to detect correlations among factors across a national dataset, I was focused on the effects of changes to hydrological regime and land use in an individual catchment. Furthermore, national-scale databases lack information for some important variables (e.g., fine sediment cover) that cannot be predicted at small scales. Therefore, the factors I chose as parents of MCI differed somewhat from those identified by these three studies, and were also based on our opinions as freshwater ecologists.

As for periphyton, instead of showing the entire conditional probability table for MCI, I describe the relationship between each parent and MCI, and the method for combining these relationships.

#### Mean summer water temperature

*Node name:* mean summer water temp. *Units:* °C

Clapcott et al. (2013) identified summer temperature as one of the four variables in the FENZ (Freshwater Ecosystems of New Zealand) database that is most strongly correlated with MCI value in a national database of 1033 sites. Using the database of Clapcott et al. (2013), I determined the proportional frequencies (probabilities) of MCI scores in the different condition bands in the categories of mean summer temperature defined by the 25th, 50th and 75th percentiles in the dataset, adjusted slightly to convert air temperature (a FENZ variable) to water temperature and to harmonise these categories with the categories used for predicting periphyton and trout growth. I then calculated the proportional change in probability relative to the lowest temperature category, which was considered to represent an unstressed state and had the highest proportion of high MCI scores (Table 6).

**Table 6: Probabilities of MCI in the four condition bands with different mean summer water temperatures. Probabilities are expressed as proportional change relative to the lowest temperature category**.

| | MCI >119 | MCI 100-119 | MCI 80-99 | MCI <80 |
|---|---|---|---|---|
| Temp <16.4 °C | 0% | 0% | 0% | 0% |
| 16.4 to 17.3 °C | -19% | -1% | -5% | 63% |
| 17.3 to 19 °C | -26% | -19% | 4% | 105% |
| >19°C | -74% | -17% | 41% | 168% |

## Fine sediment cover

*Node name:* Fine sediment % cover. *Units:* % cover

Deposition of fine sediment (silt) is widely recognised as a major impact of changing land use on river ecosystem health (Clapcott et al. 2011). Clapcott et al. (2011) determined a relationship between % cover of fine sediment (assessed visually from bankside) and MCI for 454 sites across New Zealand. They defined a threshold of 20% to separate healthy from unhealthy rivers based on MCI. I used their scatterplot (Fig. 4-21 in Clapcott et al. 2011) to calculate the proportional frequencies (probabilities) of MCI scores in the different condition bands above and below the 20 % threshold, and expressed these as a proportional change in probability relative to the <20% sediment cover state (Table 7). The Fine sediment % cover node has an additional state (10-20%) defined for the sake of trout spawning habitat. Since Clapcott et al. (2011) describe only a single threshold relating to change in MCI, I did not define a change in MCI based on the difference between 0-10% and 10-20% fine sediment cover.

**Table 7: Probabilities of MCI in the four condition bands with different fine sediment % cover based on data in Clapcott et al. (2011). Probabilities are expressed as proportional change relative to the <20% cover state.**

| | MCI >119 | MCI 100-119 | MCI 80-99 | MCI <80 |
|---|---|---|---|---|
| Sediment <20% | 0% | 0% | 0% | 0% |
| Sediment >20% | -64% | -2% | 52% | 123% |

## Change in mean annual low flow

*Node name:* % change in MALF. *Units:* % change

Booker et al. (2015) found that the hydrological variable most strongly influencing MCI among 1075 river sites across New Zealand was specific MALF, i.e., the mean annual low flow divided by catchment area. Since the catchment area of the hypothetical river I consider here will not change with development, a % change in specific MALF should be equal to a % change in MALF, thus I represented Booker et al.'s data as % change in MALF. Only a weak relationship was found, MCI decreasing from an average of 105 at specific MALF of 0.04 to an average of 101.5 at specific MALF of 0.0025. A weak relationship between invertebrate communities and low flow level was also found by Suren and Jowett (2006). I calculated the change in MCI with a % change in MALF (in categories described below). Although Booker et al.'s relationship between specific MALF and MCI was not linear, I assumed a linear relationship, as the error by doing so was considered insignificant. I then converted change in average MCI value to a change in probability of occurrence in each of the four MCI condition bands. Since I did not have access to Booker et al.'s dataset, I used another large macroinvertebrate dataset from Matheson et al. (2015) that likely contains much of the same data. In this dataset I determined the proportional frequencies (probabilities) of MCI scores in the four condition bands, then decreased the MCI scores by the amount corresponding to the % change in MALF and recalculated the probabilities. I then determined the change in each probability as a proportion of the probability at 0-5% change in MALF (Table 8).

**Table 8: Probabilities of MCI in the four condition bands with different % decrease in MALF (mean annual low flow) Probabilities are expressed as proportional change relative to 0-5% decrease in MALF.**

| % MALF decrease | MCI >119 | MCI 100-119 | MCI 80-99 | MCI <80 |
|---|---|---|---|---|
| 0-5% | 0% | 0% | 0% | 0% |
| 5-50% | -3% | 0% | 1% | 1% |
| 50-90% | -4% | 1% | 2% | 2% |
| >90% | -5% | 0% | 3% | 3% |

## Periphyton biomass

*Node name:* Periphyton. *Units:* mg chl *a* per m$^2$.

The main effects of pastoral land use on macroinvertebrates in rural streams (not considering changes to the riparian zone) are via increased inputs of silt, nutrients and organic matter. Nutrient inputs affect macroinvertebrates mainly through their effects on increased periphyton biomass, which can alter the type and quantity of food available to macroinvertebrates and change the physical habitat. A small increase in periphyton biomass can be beneficial to the macroinvertebrate community as it represents an increase in available food. However, the main effect of this subsidy is on macroinvertebrate densities, whereas MCI is based on presence-absence only. Therefore, I expected (and found) that a small increase in periphyton biomass would not change the distribution of MCI scores, and a large increase would reduce proportional frequencies in the higher MCI condition bands. I calculated the proportional frequencies (probabilities) of MCI scores in the different condition bands in the categories of periphyton biomass defined for the National Objectives Framework (NZ Government 2014) using the database of Matheson et al. (2015). I then calculated the proportional change in probability relative to the lowest periphyton biomass category, which had the highest proportion of high MCI scores (Table 9).

**Table 9: Probabilities of MCI in the four condition bands with different amounts of periphyton biomass based on data in Matheson et al. (2015). Probabilities are expressed as proportional change relative to the lowest periphyton biomass category**.

| | MCI >119 | MCI 100-119 | MCI 80-99 | MCI <80 |
|---|---|---|---|---|
| Chl-a <50 mg/m$^2$ | 0% | 0% | 0% | 0% |
| 50 to 120 mg/m$^2$ | -71% | 6% | 88% | 63% |
| 120 to 200 mg/m$^2$ | -90% | 6% | 102% | 138% |
| >200 mg/m$^2$ | -94% | -40% | 198% | 130% |

### 5.2.3 Combining parents of Macroinvertebrate Community Index

The basic distribution of MCI scores among the four condition bands (the “prior distribution” in Bayesian terms) was taken from the dataset of Matheson et al. (2015), which included 1783 sites from Canterbury, Southland, Hawkes Bay, Manawatu, Greater Wellington and the National Rivers Water Quality Network. According to this dataset, 26% of sites were in “excellent” condition (MCI >119), 40% were in “good” condition (MCI 100-119), 28% were classed as “fair” (MCI 80-99) and 6% were classed as “poor” (MCI <80).

For each combination of the four factors described above (summer water temperature, fine sediment cover, change in mean annual flow and periphyton biomass), the prior probabilities were adjusted by multiplying by the values given in Tables 6-9.

### 5.2.4 Effects of a development scenario on Macroinvertebrate Community Index

The possible effects of water resource development on MCI can be seen by comparing the screen shots in Figs. 5 and 6. Fig. 5 represents a river catchment with little or no abstraction of water or intensive land use. The mean annual low flow (MALF) and mean summer water temperatures remain close to natural levels, little silt has settled onto the riverbed, and periphyton biomass remains low. In this situation, MCI is most likely to be in the range 100-120 (good condition), with a 25% chance of being in excellent condition.

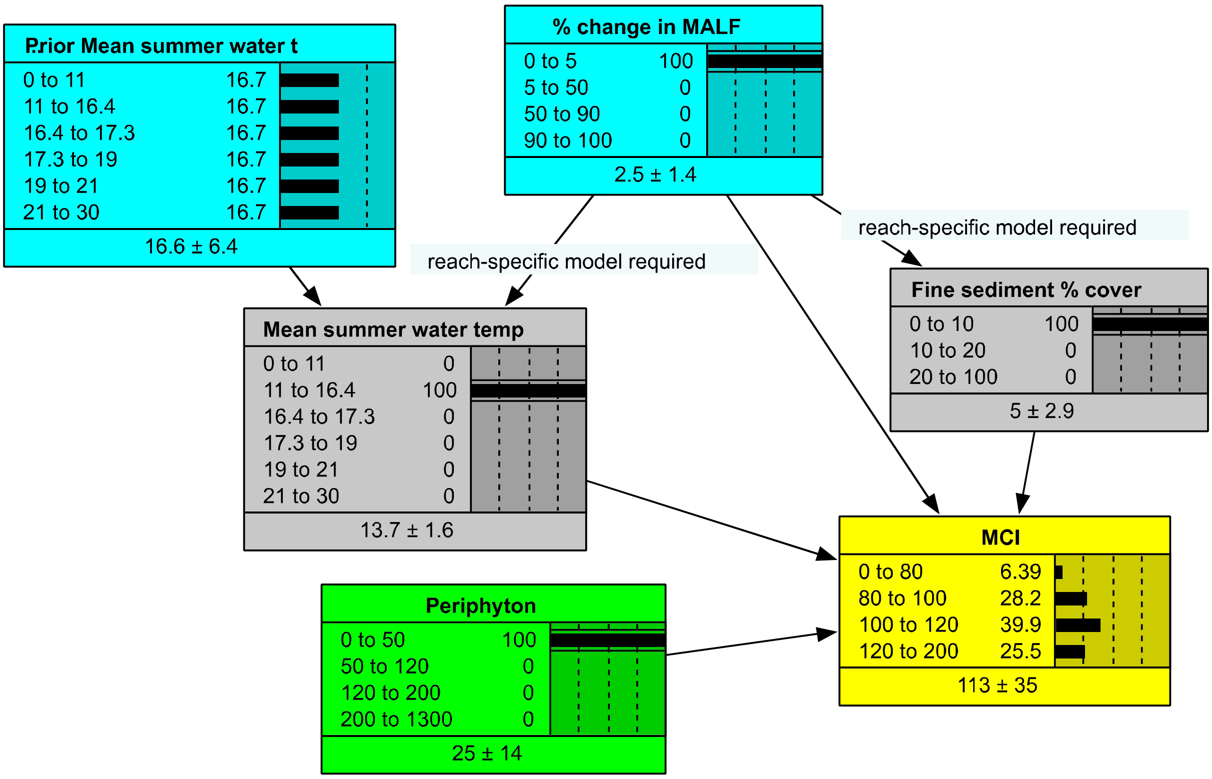


**Figure 5: Probabilities of MCI scores in different condition bands in a river catchment with little or no water abstraction or intensive land use.**

Fig. 6 represents a river catchment in which water abstraction has reduced the mean annual low flow by more than 50%. As a result, summer water temperatures have risen by about 3 °C. Associated with the water abstraction is an intensification of land use in the catchment, resulting in an increase in periphyton biomass (through elevated nutrient concentrations), and an increase in fine sediment deposition due to increased soil erosion. As a result, MCI is most likely to drop to between 80 and 100 (fair condition), with a 32% chance of being less than 80 (poor condition).

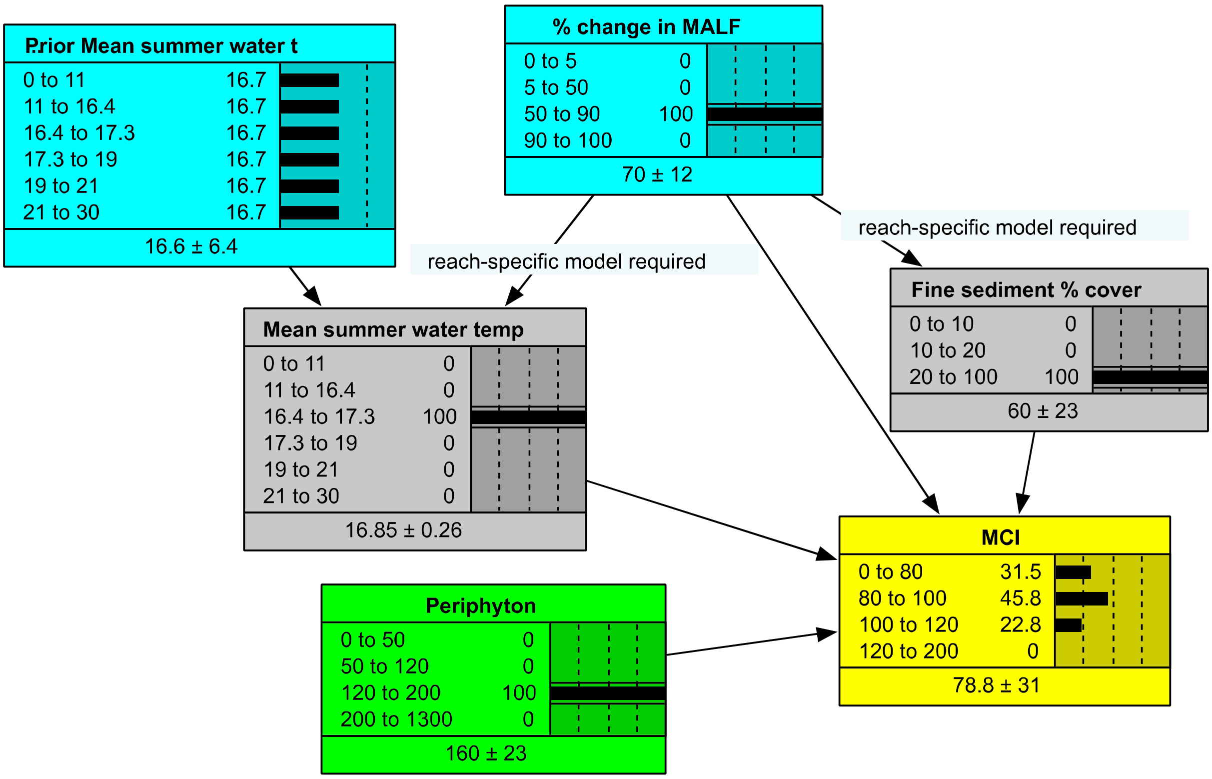


**Figure 6: Probabilities of MCI scores in different condition bands in a river catchment with significant water abstraction and land use intensification.**

## 5.3 Risk of infection from secondary recreational contact

### 5.3.1 Node description and states

*Node name:* Health risk recreation. *Units:* % chance of infection

Infection risk from secondary recreational contact (e.g., wading, boating with only occasional immersion) is included here as it is one of the compulsory attributes specified in the National Objectives Framework. The levels of risk are as specified in the NOF. This node may represent secondary contact recreation if the node for *E. coli* is defined as annual median levels (compulsory), or may represent primary contact recreation (full immersion) if the node for *E. coli* is defined as the 95$^{th}$ percentile of all samples taken (optional for Regional Councils to adopt under the NOF).

### 5.3.2 Node parents

This node has only one parent, the concentration of *E. coli*.

#### *E. coli*

*Node name:* E. coli conc. *Units:* number of colony-forming units per 100 mL.

*E. coli* is an indicator bacteria that may not cause disease itself, but indicates the presence of other animal- or human-derived bacteria that are likely to cause disease. The states of the node *E. coli* conc are as specified for condition bands A-D in the National Objectives Framework.

*E. coli* concentrations are expected to increase with land use intensification. Increases in *E. coli* runoff from land can be determined by the CLUES model. Concentrations of *E. coli* in rivers resulting from land use intensification can be predicted as a product of the current concentration and the % increase.

## 5.4 Suitability for swimming

### 5.4.1 Node description and states

*Node name:* Suitability for swimming. *Units:* categories OK, not OK

Suitability for swimming is considered here to be a function of health risk (risk from waterborne disease, measured as concentration of *E. coli*), physical risk (injury from snags or other underwater features) and aesthetics (Nagels et al. 2001). Physical risk increases mainly when underwater features cannot be seen due to reduced water clarity. The aesthetic quality of a swimming site is affected mainly by excess growths of periphyton and visual clarity, which affect both the look and the feel of a site. One could argue that deposition of fine silt is also a component of aesthetics affecting swimming, but deposited silt was not one of the primary factors identified by the expert panel in Nagels et al. (2001).

### 5.4.2 Node parents

To quantify the effect of clarity on suitability for swimming, I used the relevant suitability curve in Nagels et al. (2001). Suitability for swimming (Table 10) was determined from the curve at the midpoint of each state in the clarity node (0.7 m, 2.6 m, 4.4 m and >5 m).

For the effect of periphyton biomass, I based my estimates on the guideline for contact recreation of "100 mg/m$^2$ as seasonal maximum of exposed surface area" (MfE 1992, cited in Matheson et al. 2015). I assumed in the states <50, 120-200 and >200 mg/m$^2$, suitability for swimming was 100%, 0% and 0% respectively (Table 10). For the state 50 to 120 mg/m$^2$, I assumed that 85 mg/m$^2$ (the midpoint of this category) exceeded one month in 12 was roughly equivalent to an annual maximum sample of 100mg/m$^2$, therefore there is a 50% chance of periphyton biomass in this category exceeding the guideline value.

For the effect of *E. coli*, I assumed equivalence between risk of infection and suitability for swimming being "not OK", except that risk >5% is equivalent to 100% suitability for swimming "not OK", in accordance with National Objectives Framework recommendations. This occurs at >540 CFU/100 mL in the case of full immersion recreation (swimming). Note that for primary contact recreation (swimming), *E. coli* concentrations refer to the 95$^{th}$ percentile of samples, whereas for secondary contact recreation they refer to the median of samples.

To combine the three parents of suitability for swimming I used a "minimum operator" approach as in Nagels et al. (2001). This means that the final probability of suitability for swimming "OK" is the lowest of those predicted by three parents. The rationale for this approach is that any one condition (health risk, physical risk or aesthetics) can reduce the suitability, regardless of how good the other conditions are. Note that the user may regard a decrease in suitability caused by health risk to be more serious than a decrease caused by aesthetics. If so, the user will need to evaluate the seriousness of any decrease in suitability for swimming by determining which factor is causing the decrease.

**Table 10: Probabilities of suitability for swimming being "OK". *E. coli* concentrations are for the 95th percentile of samples**.

| E. coli (CFU per 100mL) | Suitability for swimming OK | Periphyton (mg/m²) | Suitability for swimming OK | Visual clarity (m) | Suitability for swimming OK |
|---|---|---|---|---|---|
| <260 | 99% | Chl-a <50 | 100% | <1.4 | 22% |
| 260 to 540 | 95% | 50 to 120 | 50% | 1.4 to 3.75 | 70% |
| 540 to 1000 | 0% | 120 to 200 | 0% | 3.75 to 5 | 82% |
| >1000 | 0% | >200 | 0% | >5 | 100% |

## 5.5 Trout size and abundance

### 5.5.1 Node description and states

*Node name:* Trout size abund. *Units:* categories good, medium, poor

The trout node attempts to estimate the state of the trout population in a way that relates to its value for angling. Angler surveys show that both trout abundance and trout size are relevant to angling, as rivers supporting large numbers of small trout are not favoured (John Hayes, Cawthron Institute, pers. comm.). Therefore this node represents the product of abundance and size. Jowett (1990) used the product of size and abundance to estimate trout biomass, the measure by which he separated river sites into different classes on the basis of their trout populations. Unlike Jowett (1990) I did not distinguish between brown trout and rainbow trout, in order to keep our model relatively simple.

The state categories of good, medium and poor are defined in terms of trout biomass as >2.0 g/$m^2$, 0.5-2 g/$m^2$ and <0.5 g/$m^2$. These are the thresholds use by Jowett (1990) to distinguish rivers with "high"' "medium" and "low" trout biomass. They separate the upper 15%, the middle 45$^{th}$ to 85$^{th}$ percentile and the lower 45% of the 157 sites surveyed by Jowett (1990), a collection of sites that represents a wide geographic spread across New Zealand and a wide range of catchment and channel conditions (Jowett 1992).

### 5.5.2 Network structure

Trout abundance is influenced by a different set of factors to trout size, therefore they are each shown by a separate branch of the Bayesian network. Trout abundance is mainly a function of habitat area and recruitment from spawning (Jowett 1992, John Hayes pers. comm.) whereas trout size is influenced primarily by temperature, density of prey (drifting invertebrates) and the ability of trout to see their prey. The importance of habitat area and macroinvertebrate prey abundance were highlighted by Jowett (1992) who showed that among 89 river sites across New Zealand, 64.4% of variability in the abundance of large (>200 mm) brown trout was explained by habitat area and benthic macroinvertebrate biomass.

Dissolved oxygen is an over-riding factor that may cause mortality if it reaches very low levels, or retard growth and development at sub-lethal levels (Davies-Colley et al. 2013). The following subsections describe these nodes and associated branches of the network.

## Trout habitat area % protected

*Node name:* Trout habitat area %. *Units:* % of natural extent

Since different trout species and life stages have different habitat requirements (different habitat suitability curves), a clear definition of trout habitat area is needed. For the purpose of this Bayesian network, an appropriate definition for trout habitat area is the area of drift feeding habitat for adult brown trout at MALF (mean annual low flow). This is the definition used by Young and Hayes (1999) for trout bioenergetics modelling and by Jowett (1992) in his model of brown trout abundance.

Using this definition, changes in trout habitat area resulting from changes in flow can be predicted using RHYHABSIM. RHYHABSIM modelling requires data on the morphology of specific river reaches, which involves field surveys.

It is assumed here that a percent change in trout habitat area results in the same percent change in trout abundance. Jowett (1992) found a linear correlation between habitat area and the logarithm of brown trout abundance. Given the broad categories I use in the trout size and abundance node, the difference between raw abundance and log abundance is probably minor.

## Trout spawning

*Node name:* trout spawning. *Units:* categories good, medium, poor

Many New Zealand rivers are recruitment-limited, however in most large river systems spawning occurs mainly in tributaries rather than the mainstem (John Hayes, Cawthron Institute, pers. comm.). Therefore, development scenarios that reduce the spawning potential of the mainstem but not tributaries may not have a large impact on trout abundance in the mainstem. The mainstem may be an important site for spawning if it has very stable flows, e.g., lake- or spring-fed rivers. In this Bayesian network I allow the user to determine whether the mainstem is important for spawning by choosing “yes” or “no” for the node “Important for spawning”. If “no” is selected, the state of the “trout spawning” node will not affect trout size and abundance. The user may select “yes” if a development scenario being considered affects the key spawning tributaries as well as the mainstem.

Jowett (1992) found that brown trout were absent from rivers with poor spawning habitat. Therefore in this Bayesian network I set trout spawning as a “minimum operator”, i.e., trout size and abundance cannot be in a better state than trout spawning (assuming that “yes” is selected for “important for spawning”). Provided other factors are ideal, trout size and abundance will assume the same state as trout spawning.

*Changes in trout spawning with abstraction and land use intensification*

Trout spawning is affected primarily by water temperature, dissolved oxygen and clogging of river beds by silt (Hay et al. 2006). These aspects of water and habitat quality are similar to those affecting adult trout. The differences are that successful spawning requires lower water temperatures, higher dissolved oxygen and “cleaner” gravels than adult trout do, and that these requirements are during the winter spawning period rather than during the summer growth period. Jowett (1992) noted that trout were rare or absent from New Zealand rivers with minimum annual (i.e., winter) water temperatures >10 °C, and attributed this to inability to spawn. He defined three levels of spawning preference as <10 °C, 10-11 °C and >11 °C. I have equated these levels to good, medium and poor trout spawning.

Trout spawning also requires high concentrations of dissolved oxygen within the gravels that eggs are laid in. Maintaining high dissolved oxygen among the gravels requires maintaining even higher oxygen concentrations in the overlying water. Davies-Colley et al. (2013) cite a USEPA study that describes no, slight, moderate and severe impairment of production of early life-stage trout at dissolved oxygen concentrations of 11, 9, 8 and 7 mg/L, respectively. A concentration of 6 mg/L marks the limit to avoid acute mortality. I used these values to define the effect of winter dissolved oxygen on trout spawning (Table 11).

**Table 11: Probabilities of trout spawning being in good, medium or poor state as a function of winter dissolved oxygen concentration in the overlying water.**

| | Trout spawning Good | Trout spawning Medium | Trout spawning Poor |
|---|---|---|---|
| oxygen <5 mg/L | 0% | 0% | 100% |
| 5-7 mg/L | 0% | 0% | 100% |
| 7-8 mg/L | 0% | 20% | 80% |
| 8-11 mg/L | 20% | 80% | 0% |
| >11 mg/L | 100% | 0% | 0% |

Spawning trout appear to be more sensitive to clogging of river beds by fine silt than are macroinvertebrates. According to Crisp & Carling (1989), greater than 20% sediment is generally seen as a threshold for suitable spawning habitat, 10-20% sediment provides adequate to poor spawning habitat (embryo survival will be affected), less than 10% is good and no sediment is optimal. In this Bayesian network, ranges of 0-10%, 10-20% and >20% sediment cover correspond to good, medium and poor trout spawning.

The three factors influencing trout spawning interact as “minimum operators”, i.e., the state of trout spawning corresponds to the lowest state among the three causative factors.

## Trout maximum size

*Node name:* trout max size. *Units:* % of maximum size expected in a reference stream

The maximum size that trout can attain is a direct function of their growth rate, thus I use maximum size and growth rate inter-changeably here. This enables us to use experimental data on growth rates to inform the relationships between nodes, while using a measure that is meaningful to anglers. In addition, because much data on trout size are gathered from fishing spots, but few data on growth rates are collected from field situations, using growth rate and maximum size interchangeably allows us to validate the results of this Bayesian network.

Since experiments by Elliott (1976), it has been commonly recognised that trout growth rates are primarily influenced by temperature, food supply and visual clarity of the water (Hayes et al. 2000). Food supply is defined here as the density of drifting macroinvertebrates. Drift feeding ( c.f. benthic feeding on food items on the river bed) is by far the most energy-efficient form of feeding for trout, and large invertebrates provide a much higher energy return per unit effort than small invertebrates

(Hayes et al. 2000). Therefore trout growth rates depend strongly on the density of drifting large invertebrates.

Water temperature exerts a strong influence on trout energetics (hence growth rate). Efficiency of energy uptake reduces at low water temperatures (Elliott 1976). Meanwhile trout metabolic rate, and hence energy demand, increases exponentially with temperature (Hayes et al. 2000). As a result of these two patterns, brown trout show optimal growth rates at about 13 °C, and steep declines as temperature increases or decreases away from this value. Temperature and food intake interact in complex ways. Growth rates for different combinations of temperature and food intake (expressed as a percentage of full rations) used in this Bayesian network (Table 12) were derived from growth curves in Elliott (1976).

**Table 12: Trout growth rates as percentages of maximum growth rate for different combinations of water temperature and prey abundance (expressed as % of full rations).**

| | 5-11 °C | 11-16.4 °C | 16.4-19 °C | 19-21 °C | >21 °C |
|---|---|---|---|---|---|
| 100% | 44% | 100% | 28% | 0% | 0% |
| 75% | 44% | 91% | 13% | 0% | 0% |
| 50% | 44% | 56% | 9% | 0% | 0% |
| 25% | 37% | 11% | -28% | -44% | -44% |
| 10% | 0% | -28% | -61% | -67% | -67% |

The actual availability of drifting invertebrates as prey depends on the visual clarity of the water as well as invertebrate density. This is because decreases in visual clarity allow invertebrates further from the trout to escape unseen. Hay et al. (2006) state that if clarity is maintained above 1.4 m, the foraging area for small prey should not be substantially reduced. However, large prey can be seen from further away, and these may form a significant part of diet. Therefore, to maintain optimal foraging for large (60 cm) fish on large (30 mm) prey, water clarity must be maintained above 3.75 m.

Despite evidence that visual clarity has a strong effect on trout bioenergetics, very little data are available that quantify the effect of reduced visual clarity on trout growth rates. I scaled growth rates by a factor of 0.8 for visual clarity <1.4 m and 0.95 for visual clarity of 1.4-3.75 m relative to values at clarity >3.75 m. These factors were conservative estimates based on a) the prediction of Hayes et al. (2000) that maximum trout weight would decline by 19 and 44%, respectively, when maximum prey size was reduced from 39 to 12 and 9 mm, and b) the following figure (Fig. 7) taken from Hay et al. (2006):

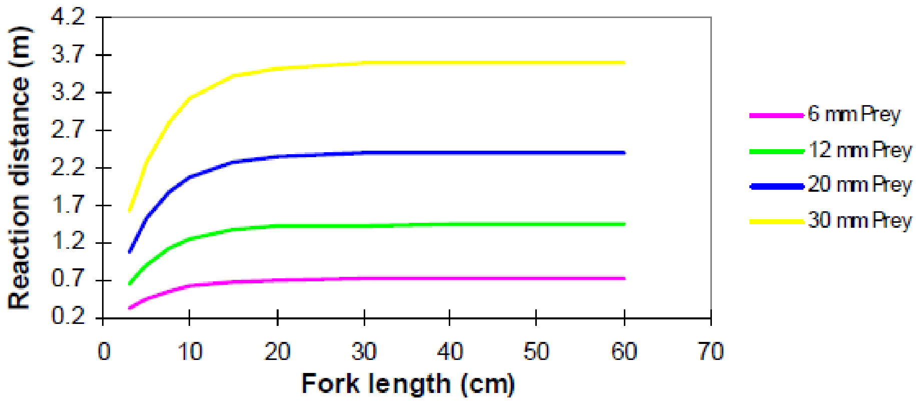


**Figure 7: Reaction distance to drifting invertebrate prey relative to fish size, and on Hughes and Dill's (1990) drift foraging model for a range of sizes of invertebrate prey. Referenced in Hay et al. (2006) Figure 1.**

*Changes in visual clarity with abstraction and land use intensification*

If increases in water abstraction lead to land use intensification in the catchment, then visual clarity is expected to decrease unless compensatory mitigation strategies are employed. Determining the change in clarity is beyond the scope of this Bayesian network, but can be estimated using CLUES (Semadeni-Davies et al. 2011). As with nutrients (Section 5.1.2), CLUES can be used to determine changes in average annual concentrations of suspended sediment by assuming that changes in concentration are proportional to changes in annual average loads. This Bayesian network includes nodes where users can enter prior (current state) clarity and the percent change in suspended sediment predicted by CLUES. Percent change in suspended sediment is converted to percent change in visual clarity using a power relationship derived from data in Davies-Colley and Close (1990).

### Trout prey index

*Node name:* trout prey index. *Units:* % of value expected in a reference site.

Trout prey index (TPI; Hayes et al. unpubl. data) is defined here as the density of large drifting macroinvertebrates. As the TPI is still under development, there are other possible definitions for it, including definitions based on presence/absence rather than density, or based on benthic or cruise feeding. Trout may feed on benthic (bottom-dwelling) as well as drifting macroinvertebrates, but at much greater energy cost, therefore trout growth rate (hence maximum size) strongly depends on the density of drifting invertebrates. This is the reason for our choice of definition.

Since different invertebrate taxa grow to different size, and have different propensities to drift, some taxa contribute much more than others to the energy intake of trout. The trout prey index takes these factors into account, assigning weightings to different taxa according to their potential contribution to trout diet. Because of these weightings, TPI is a somewhat different measure to total invertebrate density or total invertebrate biomass. It also differs from MCI because it is based on abundance rather than presence/absence and has different weightings to MCI. Despite these differences I considered that the main factors that may cause change in TPI with water abstraction and land use intensification would be similar to those causing change in MCI. However relationships may be somewhat different because TPI is a density measure.

Using the taxa weighting scores for drift feeding, Matheson et al. (2015) calculated changes in TPI with different levels of periphyton biomass. TPI shows a "hump-shaped" relationship with periphyton biomass (Table 13), reaching a maximum at biomass of 120-200 mg chl *a* /m$^2$. Up to this level, periphyton represents a food subsidy, whereas above this level it represents a disruption to invertebrate habitat.

**Table 13: Percent changes in TPI (measured as density of large drifting invertebrates) with changes in periphyton biomass. Percent changes are relative to TPI at 120-200 mg chl a / m$^2$.**

| chl-a | % change compared to optimal |
|---|---|
| <50 mg/m$^2$ | -25% |
| 50 to 120 mg/m$^2$ | -16% |
| 120 to 200 mg/m$^2$ | 0% |
| >200 mg/m$^2$ | -32% |

Floods scour macroinvertebrates from riverbeds, and population densities take time to recover following such events. Clausen and Biggs (1997) combined data for FRE3 (number of floods >3x median flow per year) and macroinvertebrate density for 83 river sites across New Zealand. From their dataset I calculated that macroinvertebrate density is about 10% less at sites with <14 days of accrual (FRE3=26) compared to sites with >14 days of accrual.

Generally, taxa providing the bulk of the diet for trout tend to be mayflies, stoneflies and caddisflies, also known as EPT taxa. The proportion of an invertebrate community comprised of EPT taxa (%EPT abundance) is a common measure of stream ecosystem health, and I used %EPT abundance as a surrogate for TPI to determine the effects of fine sediment deposition and water temperature. Clapcott et al. (2011) showed that %EPT abundance declined by 8% and 38% respectively, as cover of deposited fine sediment increased from 0-10% to 10-20% and from 0-10% to >20%, respectively. Data from NIWA's National River Water Quality Network show percent changes in %EPT abundance with water temperature as summarised in Table 14.

**Table 14: %EPT abundance and % change in EPT abundance in different categories of summer water temperature. % change is relative to that at optimal temperature of <11 °C.**

| Summer water temperature | Average % EPT abundance | % change in %EPT abundance relative to optimal (0 to 11 °C) |
|---|---|---|
| 0 to 11 °C | 39.2 | 0% |
| 11 to 16.4 °C | 39.2 | 0% |
| 16.4 to 17.3 °C | 33.4 | -15% |
| 17.3 to 19 °C | 20 | -49% |
| 19 to 21 °C | 5 | -87% |
| >21 °C | 0 | -100% |

A value for Trout prey index for each combination of states in each of the four causative factors was calculated as the product of the % change for the corresponding state in each factor. Values of trout prey index were then discretised into categories of 0-10%, 10-25%, 25-50%, 50-75% and 75-100%.

#### Dissolved oxygen

*Node name:* Dissolved oxygen. *Unit*s: mg/L

Dissolved oxygen is one of the compulsory attributes described in the National Objectives Framework. The states represented in this Bayesian network correspond to the condition bands specified in the NOF. Therefore, dissolved oxygen concentrations used here are defined as 7-day mean minimum values (i.e., the mean value of 7 consecutive daily minimum values) during summer. The narrative attribute states in the NOF state that dissolved oxygen of ≥8.0 mg/L represents no stress on aquatic organisms, 7.0-8.0 mg/L represents occasional minor stress on aquatic organisms including risk of reduced abundance of sensitive fish, 5.0-7.0 mg/L represents moderate stress on aquatic organisms including risk of sensitive fish species being lost, and <5.0 mg/L represents significant persistent stress on a range of aquatic organisms. Studies by the USEPA (cited in Davies-Colley et al. 2013) indicate that low dissolved oxygen can reduce the growth rate of salmonid fish. Using these two sources of information, I considered that at concentrations >8 mg/L dissolved oxygen has no effect on trout size and abundance, at concentrations of 7-8 mg/L trout size and abundance is reduced by a factor of 0.9, at concentrations of 5-7 mg/L trout size and abundance is reduced by a factor of 0.75, and at concentrations <5 mg/L trout size and abundance is reduced to zero.

### 5.5.3 Combining parents of trout size and abundance

A value for "trout size and abundance" was calculated as the product of trout maximum size and habitat area % protected (which is equivalent to trout abundance). This value was then modified by the scaling factors described above for dissolved oxygen. Values >0.85 were designated as "good", values between 0.45 and 0.85 were designated as "medium" and values <0.45 were designated as "poor" according to the percentiles in Jowett's (1990) dataset for high, medium and low biomass. Finally, trout spawning was incorporated by providing an upper limit to the state for trout size and abundance (i.e., if trout spawning was medium, trout size and abundance could only achieve a maximum state of medium, if trout spawning was poor, trout size and abundance could only achieve a maximum state of poor).

### 5.5.4 Effects of development scenarios on trout size and abundance

The possible effects of water resource development on trout size and abundance can be seen by comparing the screen shots in Figs. 8-10 below. Fig. 8 represents a river catchment with little or no development of its water resources. The river flow at MALF (mean annual low flow) is close to its natural level, so that trout habitat area is close to 100% of natural extent. Spawning habitat is healthy due to low water temperatures and high dissolved oxygen levels during the winter spawning season, and little clogging of riverbed sediments with silt. The trout prey index is moderately high due to low water temperatures and little clogging of the riverbed sediments by silt, but not at its peak, as periphyton biomass is low, and floods >3x median flow reduce macroinvertebrate populations relatively frequently. Nevertheless the supply of drifting macroinvertebrate food is adequate to support trout reaching maximum size, and the high water clarity ensures trout are able to see their prey easily, while moderately low temperatures keep their metabolic energy requirements low so that consumed prey can be converted to body mass with maximum efficiency.

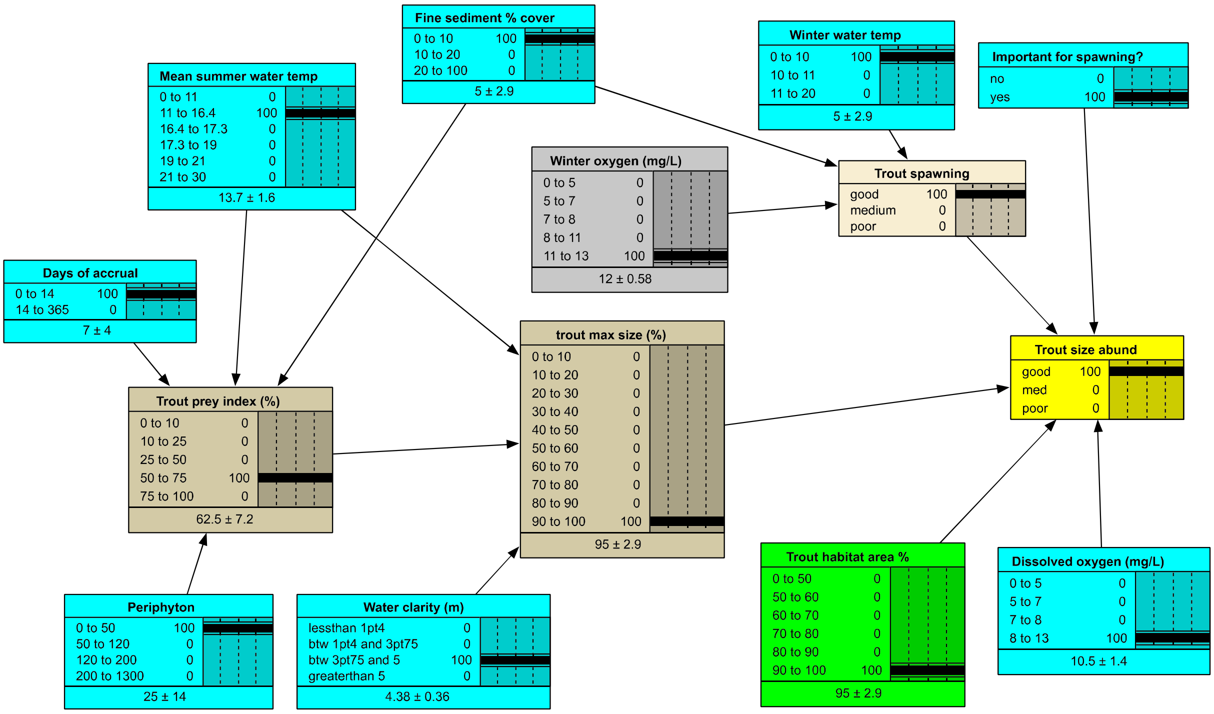


**Figure 8:** **Probabilities of trout size and abundance in good, medium or poor state in a river catchment with little or no water abstraction or intensive land use.**

Fig. 9 describes a scenario where a significant proportion of the river flow is abstracted. However, this has not resulted in significant land use intensification. Therefore, the main effects on trout are via a reduction in habitat area to 70-80% of natural extent (determined by RHYHABSIM modelling) and an increase in mean summer water temperature of about 3 °C. Periphyton cover remains low, as does cover by fine silt. Because water is abstracted mainly during summer, there is little effect on water or habitat conditions during winter, and spawning is maintained in a good state. Despite the minimal changes to food supply, trout maximum size is greatly reduced by the small increase in water temperature. This is because trout metabolic energy requirements increase sharply with a small increase in water temperature, and consumed prey are not converted efficiently into body mass (Elliott 1976). As a result, the trout fishery, as measured by size and abundance, reduces to poor condition.

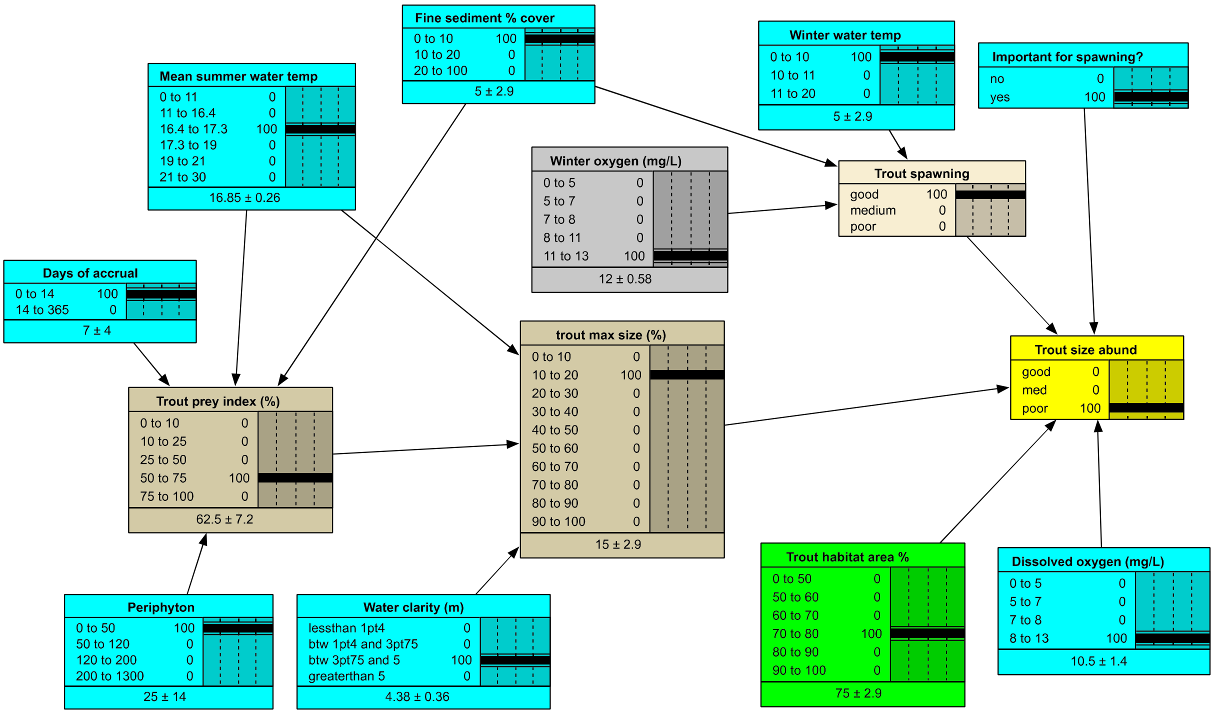


**Figure 9: Probabilities of trout size and abundance in good, medium or poor state in a river catchment with significant water abstraction but minimal effects of intensive land use.**

Fig. 10 shows a scenario where a significant proportion of the river flow is abstracted for irrigation, resulting in land use intensification. A key difference from the previous scenario, however, is that water abstraction does not result in a noticeable increase in summer mean water temperature, because this river is deep and has significant inputs of cool groundwater. Therefore, trout are able to convert energy from food into body mass with maximum efficiency. With land use intensification comes an increase in deposition of fine sediment and increase in periphyton growth, however these two changes have opposite effects on the density of drifting invertebrates, and therefore trout prey index remains moderately high. Nevertheless, trout are less able to see their prey as water clarity is somewhat reduced by the increase in fine sediment coming from the land. Therefore, maximum size of trout is less than optimal. This, combined with the slightly reduced usable habitat area (70-80% of natural) result in a trout fishery in medium condition. The spawning habitat is in poor condition due to clogging by fine sediment, but in this river system there are many alternative sites to spawn in aside from the mainstem.

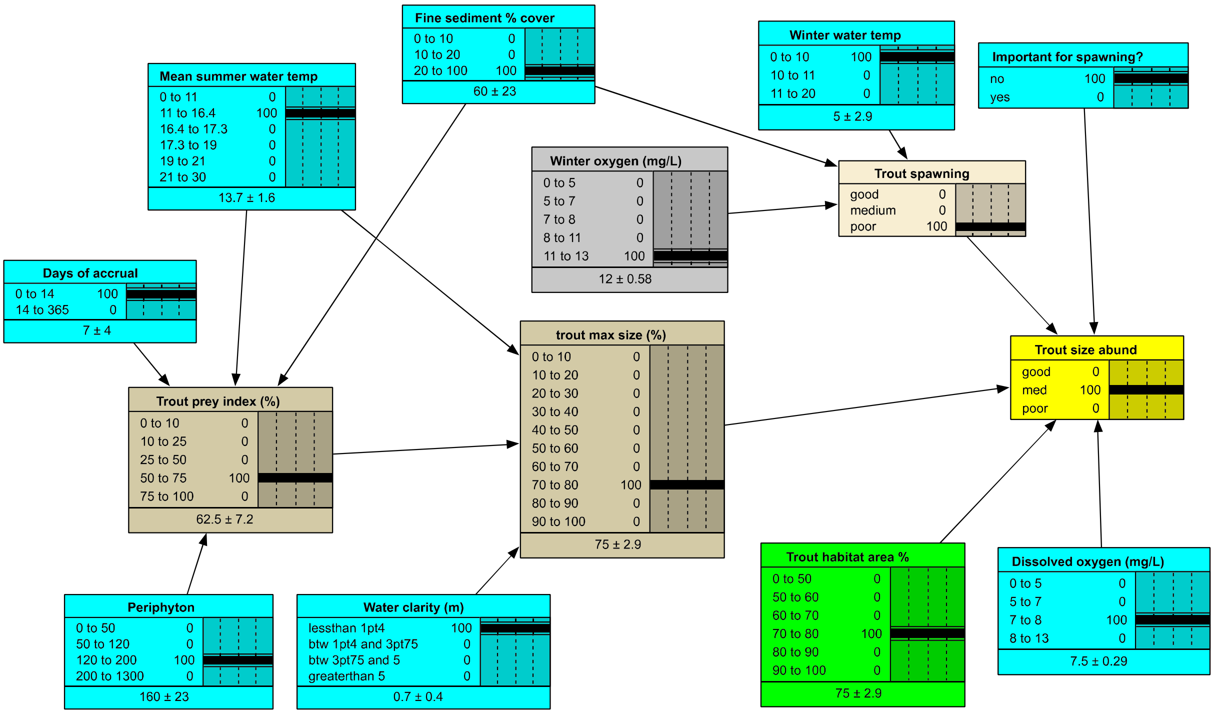


**Figure 10: Probabilities of trout size and abundance in good, medium or poor state in a river catchment with significant water abstraction and associated land use intensification.**

## 5.6 River bird richness and abundance

### 5.6.1 Node description and states

*Node name:* Bird richness abundance. *Units:* categories OK, reduced

Several species of native bird depend largely or entirely on gravel bed rivers for nesting and feeding. The species of greatest conservation concern include black-fronted tern, black-billed gull, banded dotterel and wrybill. Although each of these species has its own preferences regarding nesting and feeding, they have enough in common that a single node of a Bayesian network can be used to summarise the condition of the river bird community.

Richness refers to the number of species present at a river site. For richness to decline, i.e., a species to be extirpated from a site, implies gross habitat degradation or severe pressure from predators. Less extreme environmental changes are likely to cause declines in population sizes. This node seeks to capture the effects of population declines among the river bird species listed above.

Using only two states for this node does not imply that the river bird richness and abundance can exist in only one of these two states. Rather, the two states represent ends of a spectrum, and the probability distribution between them indicates how far along this spectrum the condition of the bird community is. An increase in the probability of "reduced" relative to "OK" should be interpreted as a decline in richness and abundance of the river bird community.

### 5.6.2 Node parents

The network for river bird richness and abundance was informed largely by evidence given by Dr Ken Hughey for the proposed Hurunui and Waiau River Regional Plan, evidence entitled "Assessment of effects of different flow regime scenarios on native riverbed nesting birds of the Hurunui and Waiau rivers", June 2012. In this evidence, Dr Hughey describes threats to two aspects of river bird ecology, nesting success and foods/feeding. In this situation, threats to nesting success were given greater weight than threats to foods/feeding.

#### Nesting success

River birds typically nest on open gravel bars, a habitat that is naturally highly dynamic due to floods that cover gravel bars and shift their locations. Although nesting is naturally at risk from unpredictable flows, nesting success on many gravel bed rivers is also subject to three main human-derived threats. These include predation by introduced mammals, physical disturbance of nests by humans or stock and vegetation encroachment on gravel banks which forces birds to nest at lower levels on gravel banks where they are more frequently disturbed by floods. Although none of these are a direct result of changes to flow regime or land use intensification, flow regime may mitigate or exacerbate the impacts of predators and weeds.

#### Impact of predators

*Node name:* impact of predators. *Units:* categories high, low.

The impact of mammalian predators on river birds is related to two factors. The first is the "braidedness" of a river, i.e., the number and size of gravel islands that are separated from the mainland. This is because mammalian predators visit bird nests on islands much less frequently than nests on connected gravel banks, despite the fact that many predators can swim. Pressure from mammalian predators is also increased where weeds encroach on gravel banks. This is because weeds tend to attract rabbits, which then attract predators, and because the weeds provide cover for predators. Unfortunately these relationships are only poorly understood (Hughey 2012). Therefore the conditional probability table for impact of predators are set more conservatively, i.e., the probabilities are distributed more evenly between "low" and "high" states of predator impact, than would be the case if there were higher certainty regarding the effects of weed encroachment and river braidedness (Table 15).

**Table 15: Conditional probability table relating river braidedness and weed encroachment to impact of predators.**

| | | impact of predators | |
|---|---|---|---|
| river braidedness | weed encroachment | low | high |
| high | low | 80 | 20 |
| high | high | 60 | 40 |
| low | low | 30 | 70 |
| low | high | 20 | 80 |

## River braidedness

*Node name:* river braidedness. *Units:* categories high, low.

River braidedness, i.e., the number and size of gravel bars, and their connection to or isolation from the mainland, is affected by three main factors. First is the supply of gravel from upper reaches. River braiding is a result of large volumes of gravel carried from upper reaches by high energy flows that deposit the gravel when they lose energy (Mosely 2004). The gravels are typically transported along the riverbed rather than in suspension, thus are termed "bedload" in this Bayesian network. Gravel supply (Bedload transport) from upper reaches may be reduced or completely halted by a dam on a major tributary or the mainstem.

The second factor is encroachment of gravel banks and braids by exotic vegetation (the node here called "weed encroachment"). Encroaching vegetation stabilises gravel bars and banks, and as a result, wide, braided rivers tend to become narrow and single-thread (Mosley 2004). The natural way that rivers reduce weed encroachment is by large floods that overtop gravel bars and have the energy to scour vegetation (Hughey 2012). Thus weed encroachment is a function of the frequency and magnitude of floods. Flood frequency and magnitude may be reduced by a dam on the mainstem or a major tributary.

The third factor is the flow at mean annual low flow (MALF). Reductions in MALF by water abstraction may lower water levels so that shallow channels dry up and gravel bars formerly isolated from the mainland become connected (Hughey 2012). Mosley (1983), Hicks et al. (2003) and Duncan (2010) showed that for several braided rivers there is a relationship between increasing flow and the number of braids. Hughey (2012) concludes that "Any reduction in the low to medium flows is thus likely… to increase vulnerability to predation."

The effect of the first and third factors on river braidedness can be determined with some accuracy using models (e.g., Hicks et al. 2003; Duncan 2010), but these require input data specific to an individual river. The effect of vegetation encroachment on river braidedness is inferred from observations (Mosley 2004) but is harder to determine precisely using models. Therefore in a generic Bayesian network, the relationship between these nodes can be described only in general terms. I have described these relationships in the following conditional probability tables (Tables 16 and 17).

**Table 16: Conditional probability table relating flood frequency and magnitude to weed encroachment.**

| | weed encroachment | |
|---|---|---|
| **flood frequency and magnitude** | **low** | **high** |
| lessthan_natural | 0 | 100 |
| natural | 100 | 0 |
| greaterthan_natural | 100 | 0 |

**Table 17: Conditional probability table relating change in mean annual low flow (MALF), bedload transport (sediment supply) and weed encroachment to river braidedness.**

| | | | river braidedness | |
|---|---|---|---|---|
| %change in MALF | bedload transport | weed encroachment | high | low |
| 0 to 5 | natural | low | 1 | 0 |
| 0 to 5 | natural | high | 0.5 | 0.5 |
| 0 to 5 | reduced | low | 0.5 | 0.5 |
| 0 to 5 | reduced | high | 0.25 | 0.75 |
| 5 to 50 | natural | low | 0.75 | 0.25 |
| 5 to 50 | natural | high | 0.375 | 0.625 |
| 5 to 50 | reduced | low | 0.375 | 0.625 |
| 5 to 50 | reduced | high | 0.1875 | 0.8125 |
| 50 to 90 | natural | low | 0.5 | 0.5 |
| 50 to 90 | natural | high | 0.25 | 0.75 |
| 50 to 90 | reduced | low | 0.25 | 0.75 |
| 50 to 90 | reduced | high | 0.125 | 0.875 |
| 90 to 100 | natural | low | 0.25 | 0.75 |
| 90 to 100 | natural | high | 0.125 | 0.875 |
| 90 to 100 | reduced | low | 0.125 | 0.875 |
| 90 to 100 | reduced | high | 0.0625 | 0.9375 |

## Disturbance of bird nesting sites

*Node name:* bird disturbance. *Units:* categories true, false

Hughey (2012) lists several human activities (four-wheel driving, movement across gravel banks for fishing or camping) that may cause disturbance to river bird nests, thus reducing nesting success. In addition, stock allowed to wander freely over gravel banks may crush nests, as can heavy machinery brought to extract gravel or rake beaches for flood control (Stephenson 2011). Such disturbances can be reduced or prevented by policies on the activities causing disturbance, and by reducing access, e.g., by fencing out stock. Although these policies and management actions are not related to changes in water resource development, this node is included because they may be part of a regional plan relating to river management.

### Food and feeding

*Node name:* Feeding OK. *Units:* categories true, false

Among the main river birds of conservation concern (listed above), most feeding is on aquatic invertebrates, especially (at least in South Island rivers) on the mayfly *Deleatidium* sp. (Hughey 2012). Food supply is believed to affect population sizes, as nesting densities of some bird species are highest where habitat conditions provide the greatest feeding opportunities. This is where the amount of river braiding is greatest (Hughey 1998, 2012). Hughey (2012) concludes that "birds, like other animals require sufficient and appropriate food supply to maintain the energy levels required for breeding." Hughey (2012) identifies the main threats to feeding habitat and food supply as:

1. reduction of low flows to the point where minor channels dry up and food supplies are lost – for territorial species this can lead to an increase in energy needed to expand and defend their territories, and

2. factors which reduce food supply (i.e., the density of benthic or drifting invertebrates).

The first point relates to the area of suitable feeding habitat. It is possible to model the areal extent of bird feeding habitat (as weighted usable area) using RHYHABSIM in the same way as for fish habitat. Habitat suitability curves are available for feeding of black fronted terns and wrybills (Duncan and Bind 2009) and possibly other species. Although the water depth and velocity preference are slightly different for each species, the changes in weighted usable area calculated by RHYHABSIM for changes in river flow are not expected to differ greatly, and could be averaged across the range of bird species present in a river. Thus in the Bayesian network, the % change in weighted usable area for bird feeding calculated by RHYHABSIM for an altered flow regime (relative to a flow regime without water abstraction) is represented by the node "Bird feeding habitat %".

The second point relates to the density of invertebrate populations. This is represented by the node Trout prey index. Trout prey index relates primarily to drifting invertebrates. Some bird species (e.g., black fronted terns) feed on drift, while others are benthic feeders (Hughey 2012), however the Trout prey index node is affected by factors driving benthic populations as well as drift density, therefore is considered an appropriate measure of invertebrate food supply for birds.

The node Feeding OK is simply the product of invertebrate food density (measured as Trout prey index) and Bird feeding habitat area. The percent probability of Feeding OK being in the "true" state is the percent of the food supply * feeding area that is retained in a modified flow regime relative to an unmodified one.

## 5.6.3 Combining parents of river bird richness and abundance

The relative importance of predator impact, food/feeding and disturbance of nesting sites is not known. Although John Craig (2013) providing rebuttal evidence on the Tukituki Catchment Proposal stated that predators and weed growth are the primary factors responsible for the decline in a number of river birds, the relative impact of predators is likely to be different in different river systems (Hughey 2012). Therefore, I have not attempted to distinguish the relative importance of predator impact vs. feeding OK, but have given both of these double the weighting of bird disturbance. The final outcome for bird richness and abundance is the product of multiplying the three factors, with the optimal state in each factor being assigned a 1, and the degraded state being assigned 0.25 for predator impact and feeding OK, and 0.5 for bird disturbance. The resulting conditional probability table is shown in Table 18.

**Table 18: Conditional probability table for bird richness and abundance as a function of bird disturbance, predator impact and Feeding OK.**

| | | | river bird richness and abundance | |
|---|---|---|---|---|
| Bird disturbance | Predator impact | Feeding OK | OK | reduced |
| FALSE | low | FALSE | 25% | 75% |
| FALSE | low | TRUE | 100% | 0% |
| FALSE | high | FALSE | 6% | 94% |
| FALSE | high | TRUE | 25% | 75% |
| TRUE | low | FALSE | 13% | 88% |
| TRUE | low | TRUE | 50% | 50% |
| TRUE | high | FALSE | 3% | 97% |
| TRUE | high | TRUE | 13% | 88% |

### 5.6.4 Effects of three development scenarios on river bird richness and abundance

The possible effects of water resource development can be seen by comparing the screen shots in Figs. 11 to 14. Fig. 11 represents a river system with little or no water abstraction or intensive land use. The impact of predators is kept relatively low as river braiding is maintained and weed encroachment is controlled by the natural flood regime and natural supply of sediment from upstream, while at baseflows, gravel islands are separated from the mainland by flows that are not artificially lowered. Nevertheless, there is always some chance of predator impact on birds. Bird feeding is maintained in good condition as the area of shallow habitat (0-10 cm deep along the margins of gravel banks and islands) is maintained, as is the density of macroinvertebrates on the river bed (expected to be proportional to the density of drifting invertebrates, which is measured as Trout prey index). The reason that Feeding OK has a 21% chance of being false is because Netica uses the midpoint rather than the upper boundary of the states in the parent variables. Disturbance of nesting and feeding areas is prevented by fencing, restrictions on access by four-wheel drive vehicles and policies preventing gravel extraction or beach raking while birds are present. As a result, the river bird community is likely (72% chance) to maintain natural levels of richness and abundance.

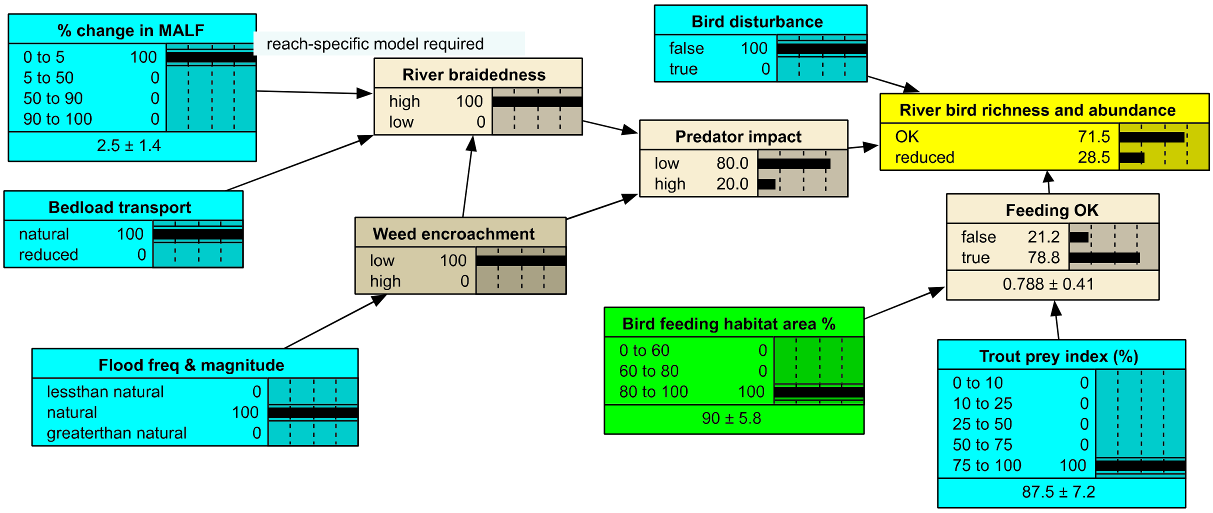


**Figure 11: Condition of river bird community (richness and abundance) with little or no development of water resources, and no habitat disturbance.**

Fig. 12 represents a river catchment where significant volumes of water are abstracted, but there is minimal intensification of land use. Water abstraction results in a decrease in the area of shallow margins for feeding on benthic macroinvertebrates, and in a slight decrease in the density of benthic macroinvertebrates (due to a small increase in water temperatures at summer baseflow). In addition, the reduced summer baseflow leads to some islands connecting with the mainland, resulting in greater access by predators to bird nests. The increased access by predators has a strong effect on river bird richness and abundance, reducing the probability of maintaining these at a natural level to 44%.

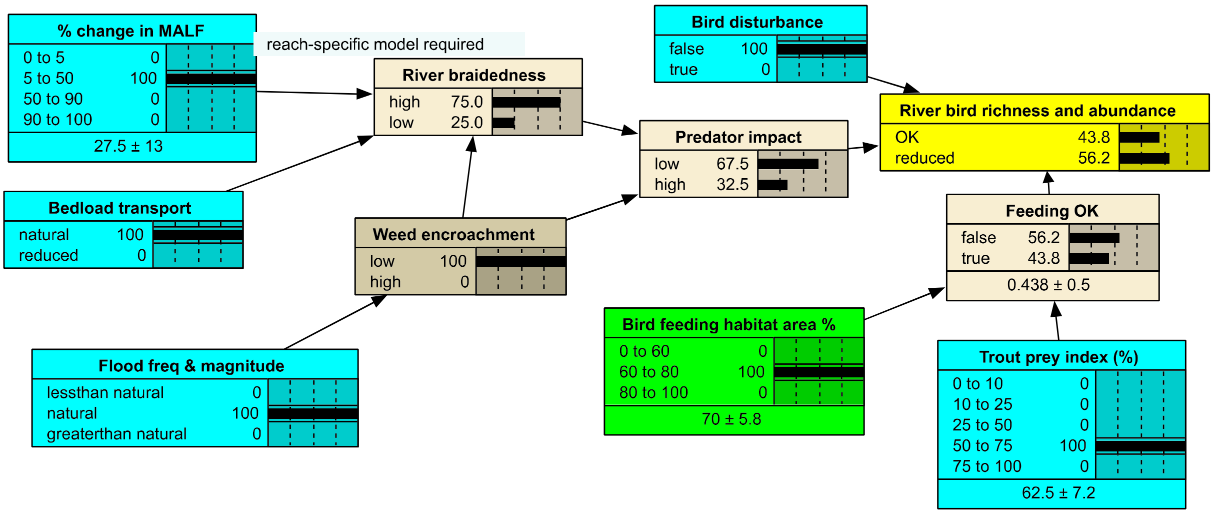


**Figure 12: Probabilities of river bird community condition (richness and abundance) with significant water abstraction but no dam or land use intensification.**

Fig. 13 represents the same scenario as Fig. 12, except that water is abstracted from a dam on the river mainstem. The dam has impacts in addition to the reduction in flows due to abstraction. These impacts include a reduction in sediment supply (bedload transport) and in flood frequency and magnitude, leading to increased encroachment of gravel bars by weeds and a decrease in river braidedness. The result is a significant increase in predator impact on nesting river birds, and a further decrease to 26% probability of maintaining bird richness and abundance.

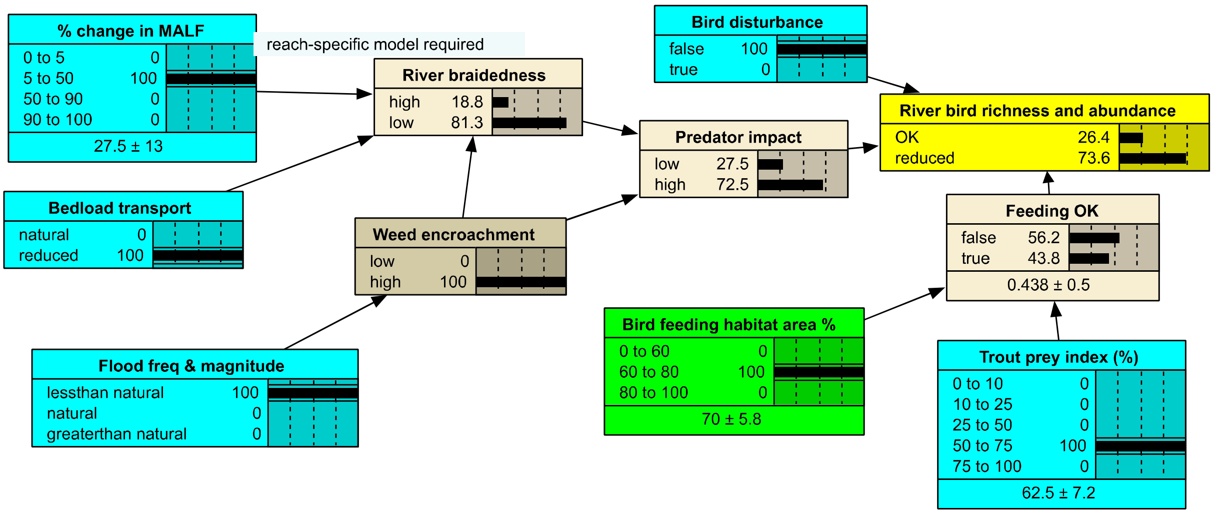


**Figure 13: Condition of river bird community (richness and abundance) with significant water abstraction and a dam on the river mainstem.**

The final scenario (Fig. 14) is the same as in Fig. 13 except that damming and water abstraction are accompanied by land use intensification. This scenario includes all the impacts of the previous one, and in addition includes a decrease in the density of macroinvertebrate food, due to increased siltation of the riverbed. River bird richness and abundance decline further, now having only a 16% chance of being close to natural levels.

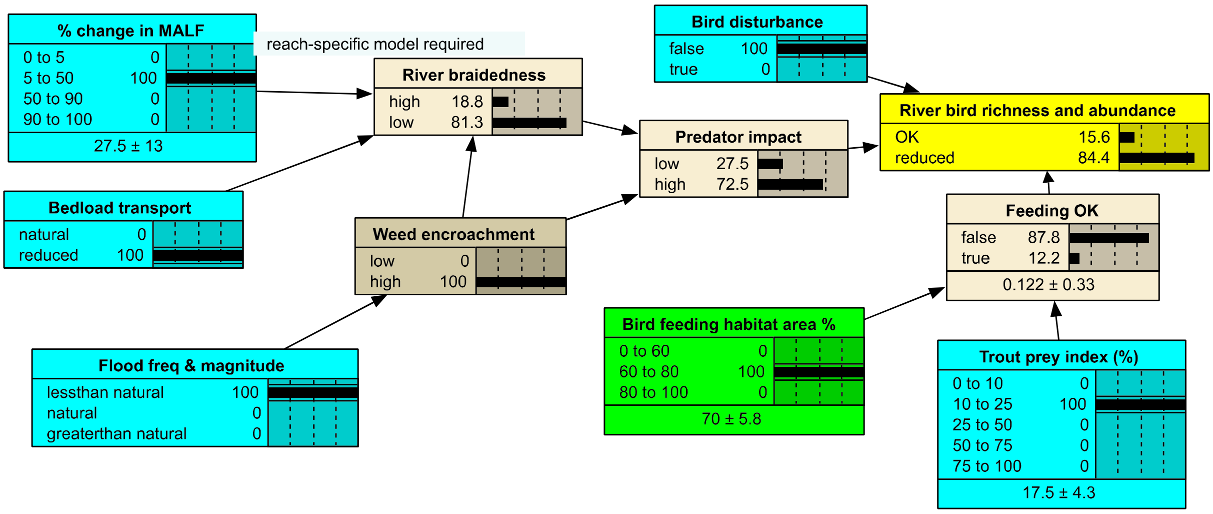


**Figure 14: Condition of river bird community (richness and abundance) with significant water abstraction, a dam on the mainstem and land use intensification.**

# 6 Results: linkage diagram

The linkage diagram (Fig. 15) includes all of the nodes represented in the predictive Bayesian network described above. However, it places these in the wider context of a river system by including outcome nodes for Ecosystem health and mahinga kai, native fish, suitability for swimming, natural character and gross domestic product. For simplicity, the linkage model uses a catch-all "Mitigations" node to indicate that application of appropriate management actions can reduce contaminant losses from land to water. Typical examples of such management actions include exclusion of stock from streams, creation of treatment wetlands, nutrient budgeting, use of nitrification inhibitors, herd shelters and feed pads for cattle, and appropriate land-based application of farm dairy effluent. Models, such as CLUES, can be used to predict the effects of applying various combinations of mitigations (Semadeni-Davies & Elliott 2011, 2012). The following subsections describe each of these outcome nodes and the variables influencing them.

## 6.1 Ecosystem health and mahinga kai

This node is a summary of ecosystem health, combining Macroinvertebrate Community Index, trout size and abundance, river birds and native fish. Mahinga kai is included as its condition is expected to depend strongly on the combination of these other nodes.

## 6.2 Native fish

The main native fish species expected to be resident in the mainstem of a large gravel bed river include torrentfish, shortfin and longfin eels, bullies (especially common bullies) and smelt (David Rowe, NIWA, pers. comm.). Most eels resident in the mainstem will be elvers, as adults prefer smaller tributaries. A variety of galaxiid species (e.g., koaro) will migrate through the river mainstem on their way to smaller tributaries, but are not likely to form significant resident populations in the mainstem.

The major impact on these native fish species expected from increased water abstraction, water impoundment and land use intensification is increased deposition of fine sediment (David Rowe, NIWA, pers. comm.). Siltation will affect eel elvers directly, as they require interstitial spaces between gravels as habitat. Siltation will affect other species primarily through its effect on benthic macroinvertebrates which are the main food of bullies, torrentfish and juvenile eels. Smelt, which are largely pelagic feeders, will therefore not be affected as greatly as bullies by the development impacts considered here.

Other potential water quality impacts on native fish include increased summer water temperatures and reduced dissolved oxygen. In addition, reduction in mean annual low flow will reduce habitat area for native fish (as it does for trout), therefore reducing fish abundances. The species with the highest flow requirements, and therefore the most sensitive to reduction in flow, is torrentfish.

One of the major impacts reducing fish occurrence and abundance across New Zealand is barriers to migration. Most native fish species (including all of those listed above) migrate between rivers and the sea, therefore require access from the sea in order for successive generations to occupy a river reach. Although barriers to fish migration are a separate issue from water abstraction, and the river reach considered here is downstream of a dam, migratory access is included here to recognise that it may be impaired by a dam, and because a policy on barrier removal may be part of a regional plan.

## 6.3 Natural character

Many of New Zealand's large gravel bed rivers are greatly valued for their natural character, and natural character is recognised in the Resource Management Act (Mosley 2004). The definition of natural character in the Resource Management Act is useful for the purpose of this report, that is "the natural or physical qualities or characteristics of an area that contribute to people's appreciation of its pleasantness, aesthetic coherence and cultural and recreational attributes" (Mosley 2004). Braided rivers in particular are valued for the unique "riverscape" they provide.

Various studies have described the most important factors contributing to the natural character of rivers, as perceived by experts and the public (Mosley 2004). Some of these factors are unchangeable features of the landscape while others may change as a result of water resource development. The variables affecting natural character in the linkage diagram (Fig. 15) draw on the results described by Mosley (2004) as well as my own perceptions.

Natural character is presented here as a function of river morphology, appearance of the water, vegetation adjacent to and within the active channel, and (when the observer is close) the visual character of the riverbed. People may notice a change to the natural morphology of the river when flows are reduced and shallower areas dry up, or when complex braided reaches are reduced to single threads. Mosley (2004) identifies the "percentage of the view composed of water surface" as a key factor influencing riverscape quality, while Di Lucas (cited in Mosley 2004) identifies braided channels as one of the scenic elements that may comprise aesthetic quality. Thus "% change in MALF" and "river braidedness" are included as factors influencing natural character in Fig. 15. Another important factor identified by Mosley is the colour class of the river (on a scale from colourless to muddy brown). In Fig. 15 this is represented by the water clarity node. Weeds, which may be trees such as willows, or herbs with bright flowers, are often highly visible features of riverscapes. Whether or not these are perceived as decreasing the natural character of a riverscape may depend on whether the viewer recognises them as weeds, but weed encroachment is included as a factor here. The aspect of the riverbed that the public appear most sensitive to is growth of periphyton, particularly long filaments. Riverbed siltation may also affect perceptions of natural character, but is not included here as it is likely to be less visible than periphyton.

## 6.4 Gross domestic product

Water resource development is usually driven by the goal of economic benefit to local producers, and/or to the region as a whole. Clearly, a primary goal for most water planning is to identify policies or management actions that provide the best outcome for the local economy as well as for ecological and social/cultural values. A Bayesian network to aid this process must therefore include measures of economic benefit.

Experience from previous (Quinn et al. 2013) and current (pers. obs.) water planning processes indicates that key values related to economics can be measured by region-wide gross domestic product (GDP), jobs, production and profit per enterprise, and land value. Water abstraction (with or without storage) increases the availability of irrigation water which can be used to convert land to more profitable uses, or increase stocking rates. It may also increase the security of supply, which results in fewer days of water restrictions and thus greater productivity over a growing season. In most cases, determining quantitative outcomes for these economic indicators will require economic modelling based on local data, as indicated by the green colour for "jobs", "enterprise profit", "production" and "land value" in Fig. 15.

One suitable model for quantifying the effects of water resource development and land use intensification on economic indicators is NZFARM. According to Daigneault et al. (2013), NZFARM is designed to “determine the level of agricultural production that maximizes the net revenue of production across the entire catchment area, subject to land use and land management options, agricultural production costs and output prices, and environmental factors such as soil type, water available for irrigation, and any regulated environmental outputs (e.g., nutrient leaching limits) imposed on the catchment. Catchments can be disaggregated into sub-regions (i.e., NZFARM zones) based on different criteria e.g., land use capability, irrigation schemes etc., such that all land in the same NZFARM zone will yield similar levels of productivity for a given enterprise and land management scheme. Total net revenue in the catchment is the variable optimised by the model.”

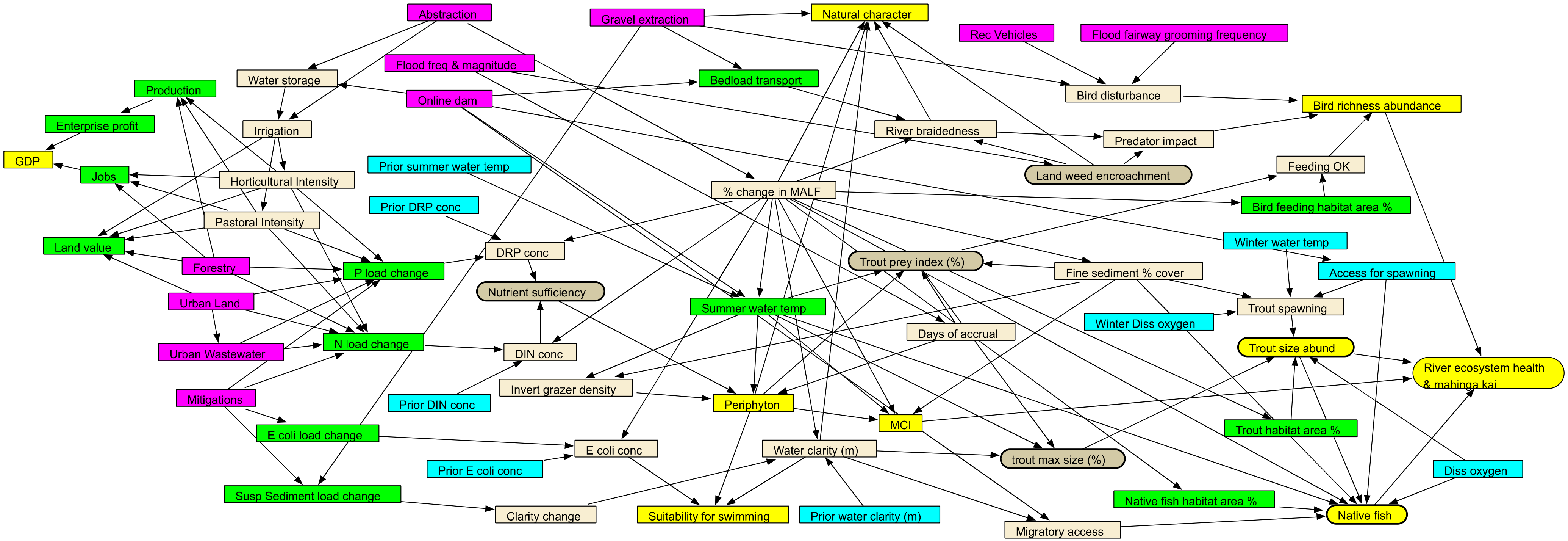


**Figure 15:** Linkage diagram showing the main factors affecting a wide range of values in a large gravel-bed river catchment subject to water abstraction, dam construction and/or land use intensification.

# 7 Conclusions

Water resource planning is inherently complex due to the large number of interacting components of a river system, the multiple values held by various members of the community and the many different management options available to achieve desired outcomes. In addition, the information available for the different components and their interactions comes in different forms, each with its own level of uncertainty. In this context, the challenge for water resource managers to achieve good outcomes across a range of values is significant, but is even greater for a stakeholder group whose members may not have experience in interpreting and synthesising scientific information. Bayesian networks represent a useful tool that can help to overcome these challenges. A linkage diagram or influence diagram, the simplest form of a Bayesian network, is a visual way to represent the components of the river system that are important to people, showing their inter-relations with other components that influence them and the inter-relations between those components and the management tools available to councils. A linkage diagram can assist a stakeholder group by providing a shared understanding of the system such that members can agree on what components of the system should be considered in their decision process and, in broad terms, how they interact. A predictive Bayesian network goes a step further by identifying the states for each component that distinguish between good and poor condition, and quantifying in probabilistic terms how one component affects another. A Bayesian network of this type can be used by a water resource manager or stakeholder group to compare the consequences of a variety of management decisions or policy options for a range of values. The network shows these consequences in a transparent way, and the information and assumptions on which the network linkages are built is also open to query.

The linkage diagram presented in this report indicates how water abstraction and associated land use intensification, with or without construction of an online dam, are related to economic growth and recreational, natural character and ecological values in the catchment and mainstem of a large gravelbed river. The predictive Bayesian network estimates the amount by which these values are likely to change under different development scenarios. The predictive relationships are derived from a number of datasets from national-scale monitoring as well as generalisations of model outputs and expert opinion. Both of these forms of the Bayesian network make it clear that the consequences of water resource development depend strongly on the way that development is done, the mitigation measures employed and the characteristics of the specific river system.

To explore the outcomes for a specific river system in general terms, a user may simply insert values representing the best available knowledge of that river into the blue nodes of the network presented here. However, to gain more accurate results, and to make the network more relevant to local issues and values, the user is invited to adapt the linkage diagram or the predictive Bayesian network through discussions with local stakeholders and using the resources identified in this report.

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# 9 References

Booker, D., Snelder, T., Greenwood, M., Crow, S. (2015) Relationships between invertebrate communities and both hydrological regime and other environmental factors across New Zealand's rivers. *Ecohydrology*, 8(1): 13-32.

Boothroyd, I., Stark, J. (2000) Use of invertebrates in monitoring. In: Collier, K.J., Winterbourn, M.J. eds. *New Zealand stream invertebrates: ecology and implications for management.* Christchurch, Caxton Press: 344-373.

Chapra, S.C., Flynn, K.F., Rutherford, J.C. (2014) Parsimonious Model for Assessing Nutrient Impacts on Periphyton-Dominated Streams. *Journal of Environmental Engineering*, 140(6): 04014014.

Clapcott, J., Goodwin, E., Snelder, T.H. (2013) Predictive models of benthic macroinvertebrate metrics, *Cawthron Report* No. 2301, prepared for Ministry for the Environment: 35.

Clapcott, J.E., Young, R.G., Harding, J.S., Matthaei, C.D., Quinn, J.M., Death, R.G. (2011) Sediment Assessment Methods: Protocols and guidelines for assessing the effects of deposited fine sediment on in-stream values, Cawthron Institute, Nelson, New Zealand: 106.

Clausen, B., Biggs, B. (1997) Relationships between benthic biota and hydrological indices in New Zealand streams. *Freshwater Biology*, 38(2): 327-342.

Collier, K.J. (2008) Average score per metric: an alternative metric aggregation method for assessing wadeable stream health. *New Zealand Journal of Marine and Freshwater Research*, 42(4): 367-378.

Craig, J. (2013) Statement of rebuttal evidence of John Laurence Craig in the matter of the Resource Management Act 1991 and in the matter of the Tukituki Catchment Proposal before the Board of Inquiry: 11.

Crisp, D., Carling, P. (1989) Observations on siting, dimensions and structure of salmonid redds. *Journal of Fish Biology*, 34(1): 119-134.

Daigneault, A., Samarasinghe, O., Lilburne, L. Modelling Economic Impacts of Nutrient Allocation Policies in Canterbury: Hinds Catchment, *Landcare Research report prepared for Ministry for the Environment*: 141.

Davies-Colley, R.J., Close, M.E. (1990) Water colour and clarity of New Zealand rivers under baseflow conditions. New Zealand Journal of Marine and Freshwater Research, 24(3): 357-365.

Davies-Colley, R., Verburg, P., Hughes, A., Storey, R. (2012) Variables for regional water monitoring underpinning national reporting: variables for national freshwater monitoring. *NIWA client report* HAM2012-006. Prepared for Ministry for the Environment: 65.

Davies-Colley, R., Franklin, P., Wilcock, B., Clearwater, S., Hickey, C. (2013) National Objectives Framework - Temperature, Dissolved Oxygen & pH. Proposed thresholds for discussion, *NIWA client report* no. HAM2013-056, prepared for Ministry for the Environment: 83.

Death, R.G., Death, F., Stubbington, R., Joy, M.K., van den Belt, M. (2015) How good are Bayesian belief networks for environmental management? A test with data from an agricultural river catchment. *Freshwater Biology*, 60(11): 2297-2309. 10.1111/fwb.12655

Death, R.G., Dewson, Z.S., James, A.B. (2009) Is structure or function a better measure of the effects of water abstraction on ecosystem integrity? *Freshwater Biology*, 54(10): 2037-2050.

Duncan, M.J. (2010) Statement of evidence to a Special Tribunal in the matter of an RMA 1991 application for a water conservation order pursuant to Section 201 of the Act and the proposed National Water Conservation (Hurunui) Order application.

Duncan, M.J., Bind, J. (2009) Waiau River instream habitat based on 2-D hydrodynamic modelling. *NIWA client report* CHC2008-176, prepared for Environment Canterbury: 72.

Elliott, J. (1976) The energetics of feeding, metabolism and growth of brown trout (*Salmo trutta* L.) in relation to body weight, water temperature and ration size. *The Journal of Animal Ecology*: 923-948.

Greenwood, M.J., Booker, D.J., Smith, B.J., Winterbourn, M.J. (submitted) A hydrologically sensitive invertebrate community index for New Zealand rivers. *Ecological Indicators*.

Hay, J., Hayes, J., Young, R.G. (2006) Water quality guidelines to protect trout fishery values, *Cawthron Repor*t No. 1205, prepared for Horizons Regional Council: 17.

Hayes, J.W., Stark, J.D., Shearer, K.A. (2000) Development and test of a whole-lifetime foraging and bioenergetics growth model for drift-feeding brown trout. *Transactions of the American Fisheries Society*, 129(2): 315-332.

Hicks, D.M., Duncan, M.J., Shankar, U., Wild, M., Walsh, J.R. (2003) Project Aqua: Lower Waitaki River geomorphology and sediment transport. *NIWA Client Report* CHC01/115. Appendix Z to Project Aqua: Assessment of Effects on the Environment. Meridian Energy Limited.

Holomuzki J.R., Lowe R.L., Ress J.A. (2006) Comparing herbivory effects of stream macroinvertebrates on microalgal patch structure and recovery. *New Zealand Journal of Marine and Freshwater Research,* 40: 357-367.

Hughey, K. (2012) Assessment of effects different flow regime scenarios on native riverbed nesting birds of the Hurunui and Waiau rivers, Proposed Hurunui and Waiau River Regional Plan and Proposed Plan Change 3 to the Canterbury Natural Resources Regional Plan Section 42A Report: 36.

Hughey, K.F.D. (1998) Nesting home range sizes of wrybill (*Anarynchus frontalis*) and banded dotterel (*Charadrius bicinctus*) in relation to braided riverbed characteristics. *Notornis,* 45: 103-111.

Jacoby J.M. (1985) Grazing Effects on periphyton by Theodoxus fluviatilis (Gastropoda) in a lowland stream. *Journal of Freshwater Ecology,* 3: 265-274.

Jowett, I.G. (1990) Factors related to the distribution and abundance of brown and rainbow trout in New Zealand clear-water rivers. *New Zealand Journal of Marine and Freshwater Research*, 24(3): 429-440.

Jowett, I.G. (1992) Models of the abundance of large brown trout in New Zealand rivers. *North American journal of fisheries management*, 12(3): 417-432.

Matheson, F., Quinn, J.M., Unwin, M. (2015) Instream plant and nutrient guidelines. Review and development of an extended decision-making framework Phase 3, HAM2015-064: 118.

Ministry for the Environment (1992) Water Quality Guidelines No. 1. Guidelines for the Control of Undesirable Biological Growths in Water: 60p.

Ministry for the Environment (2013) *Freshwater reform 2013 and beyond*: 57.

Ministry for the Environment (2014) *National Policy Statement for Freshwater Management 2014*. NZ Government: 34.

Mosley, M. (1983) Response of braided rivers to changing discharge. *Journal of hydrology. New Zealand*, 22(1): 18-67.

Mosley, M.P. (2004) Rivers and the riverscape. In: J.S. Harding, M.P. Mosley, C.P. Pearson & B.K. Sorrell (Eds). *Freshwaters of New Zealand*. New Zealand Hydrological Society and New Zealand Limnological Society, Christchurch: 8: 1-8:18.

Nagels, J., Davies-Colley, R., Smith, D. (2001) A water quality index for contact recreation in New Zealand. Water Science and Technology, 43(5): 285-292.

New Zealand Government (2014) National Policy Statement for Freshwater Management 2014: 34.

Quinn, J.M., Monaghan, R.M., Bidwell, V.J., Harris, S.R. (2013) A Bayesian Belief Network approach to evaluating complex effects of irrigation-driven agricultural intensification scenarios on future aquatic environmental and economic values in a New Zealand catchment. *Marine and Freshwater Research*, 64(5): 460-474.

Rutherford, K. (2001) Ngaruroro River water temperatures: modelling the effects of low flow. *NIWA Client Report*: HBR01203, prepared for Hawke's Bay Regional Council: 46.

Semadeni-Davies, A., Elliott, S. (2011) Impacts of land use and farm mitigation practices on nutrients. Application of CLUE to the Mataura catchment. *NIWA Client report* HAM2011-018, prepared for Environment Southland: 38.

Semadeni-Davies, A., Elliott, S. (2012) Preliminary study of the potential for farm mitigation practices to improve river water quality. Application of CLUES to the Waikato Region. *NIWA Client report* HAM2012-34, prepared for Waikato Regional Council: 169.

Snelder, T.H., Booker, D.J., Quinn, J.M., Kilroy, C. (2014) Predicting Periphyton Cover Frequency Distributions across New Zealand's Rivers. *JAWRA Journal of the American Water Resources Association*, 50(1): 111-127.

Stark, J., Maxted, J. (2007) A User Guide for the Macroinvertebrate Community Index. *Cawthron Report* No. 1166, prepared for the Ministry for the Environment: 58.

Stephenson, B. (2011) Baseline study and assessment of effects on braided riverbed bird communities. Eco-Vista Photography & Research Ltd. Report prepared for Hawke's Bay Regional Council.

Suren, A.M., Jowett, I.G. (2006) Effects of floods versus low flows on invertebrates in a New Zealand gravel-bed river. *Freshwater Biology*, 51(12): 2207-2227.

Welch E.B., Anderson E.L., Jacoby J.M., Biggs B.J.F., Quinn J.M. (2000) Invertebrate grazing of filamentous green algae in outdoor channels. *Verhandlungen der Internationalen Vereinigung für Theoretische und Angewandte Limnologie,* 27: 2408-2414.

Welch E.B., Quinn J.M., Hickey C.W. (1992) Periphyton biomass related to point-source nutrient enrichment in seven New Zealand streams. *Water Research,* 26: 669-675.

Young, R.G., Hayes, J. (1999) Trout energetics and effects of agricultural land use on the Pomahaka trout fishery. *Cawthron Report* No. 455, prepared for Fish and Game New Zealand: 29.