Integrating biophysical and economic systems in a Bayesian Network Hydro-economic framework

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Abstract

Management of water resources needs to jointly consider the multiple, interdependent, uses of water. Decision support tools that aim to assist efficient integrated water resources management should integrate the environmental and socio-economic systems affected by changes in resource allocation. There exist, however, few models that assess the trade-offs between environmental and economic impacts of water management changes in an integrated framework. This paper presents a hydro-economic model that integrates hydrological and ecological systems with economic costs and nonmarket benefits in a Bayesian Network modelling framework. A suite of different modelling tools were used to assess the biophysical and economic impacts of catchment management scenarios, for a case study in Tasmania, Australia. The Bayesian Network provides a flexible modelling approach to incorporate different types of data and had the advantage of explicitly accounting for accumulated uncertainty in information.

Keywords: Integrated Modelling; Uncertainty; Nonmarket Valuation; Choice Experiments; Integrated Water Resource Management; Hydrological Modelling; Hydro-ecological modelling

JEL Classifications: C69; Q20; Q51; Q57

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1. Introduction

Sustainable management of water resources faces the difficult task of coordinating multiple, often competing, uses of water, in a way that balances environmental and socio-economic demands. An approach to Integrated Water Resources Management (IWRM) implies that water, land and related resources, are managed in ways that aim to "maximise economic and social welfare in an equitable manner, without compromising the sustainability of vital ecosystems" (Global Water Partnership, 2000). If management is targeted at maximising economic and social welfare, we need information about the socio-economic values affected by different uses of water resources, including those associated with agricultural production, biodiversity conservation, recreation and other economic activities.

One of the tools that is often used to support IWRM is hydro-economic modelling. Hydro-economic models aim to integrate hydrology, water management, environmental conditions, and socio-economics aspects of water resources management in a coherent modelling framework (Harou et al., 2009). Hydro-economic models bring economics to the core of water resources management, by evaluating the economic values of alternative water management options (Pulido-Velazquez et al., 2008; Harou et al., 2009). In these models, optimal water use is thus driven by the economic values generated by the different uses of water resources.

Hydro-economic models have been used for decades by engineers and economists to support policy decisions about water management (see, for example, Andreu et al., 1996; Rosegrant et al., 2000; Ahrends et al., 2008; Volk et al., 2008; Medellín-Azuara et al., 2009; Cools et al., 2011; George et al., 2011). Many hydro-economic models focus on optimal water extraction to maximise (agricultural) production values, or on minimising the costs of reduced water supply or pollution. In a typical hydro-economic model, water extraction infrastructure (such as canals, reservoirs, pumping stations) is linked to hydrologic features of the system (such as river flows, precipitation, evaporation, groundwater recharge) in a node-link network, where economic costs and benefits are generated by water use and production activities (see Rosegrant et al., 2000; Harou et al., 2009). Excellent reviews of hydro-economic modelling applications include Ward et al. (2006) and Heinz et al (2007) on catchment-scale models; and Harou et al. (2009) on water allocation models. McKinney et al (1999), Heinz et al. (2007), Brouwer and Hofkes (2008), and Cai (2008) further present methodological papers about linked hydrologic-economic models.

Notwithstanding the emphasis of IWRM on maximising economic and social welfare while preserving ecosystems, there are relatively few models that integrate hydrological changes, ecosystem impacts, and economic costs and benefits in a robust framework to guide water management decisions (Brouwer and Hofkes, 2008; Kragt et al., 2010; Grové, 2011). Methodological and operational challenges to linking hydrological and economic systems (see McKinney et al., 1999; Brouwer and Hofkes, 2008) have resulted in limited ability of existing models to account for the inherent uncertainty in environmental systems, and a lack of models that incorporate the effects of water management changes on ecology and nonmarket (intangible) values provided by ecosystems.

Risk and uncertainty play a major role in IWRM. The ways in which uncertainty manifests itself will vary between economic, hydrologic, and ecological systems in terms of the underlying sources, characteristics and size. A key challenge to integrated modelling is how to account for the fundamentally different types of uncertainty in input data, model structure, parameter values, and model results in a consistent way (Brouwer and Hofkes, 2008). In modelling, uncertainty is often accounted for through stochastic programming or sensitivity analysis. For example, Cai et al. (2003) describe how they vary four important model input parameters over a specified range to assess the effects of parameter uncertainties on aggregate estimates. In integrated models, traditional stochastic programming with uncertain parameters can meet serious computational difficulty if uncertainties are correlated across items (Cai, 2008). An alternative approach to accounting for uncertainty is described by Barton et al. (2008), where a Bayesian Network modelling approach is use to define joint uncertainties in the model input, relationships between variables, and output. The researchers used empirical data and expert opinion to express uncertainties as conditional probability distributions, which allows a quantification of uncertainty in the model components as a risk.

Water managers need information about the impacts of policy changes on ecosystem conditions. Hydro-ecological models are aimed at assessing such changes by explicitly considering hydrological and ecological processes, and the interactions between them. Hydro-ecological models may focus on the ecological impacts of eutrophication and acidification of surface water (Fujita et al., 2007); protection and restoration of healthy natural wetland habitats (Zhou et al., 2008); or flood control policies (Brouwer and van Ek, 2004). Although most of these models include the direct financial costs of undertaking management actions, few assess the economic costs and benefits that result from a change in ecosystem conditions. An example of a hydro-ecological model that incorporates economics is the NELUP model (Moxey et al., 1995; Rushton et al., 1995). NELUP assesses how rural land use changes in the River Tyne catchment, UK, affect surface water and groundwater flows. An economic module, based on a linear programming, predicts how agricultural land use varies with different policy conditions and market prices. Subsequently, hydrological and ecological modelling components predict changes in water flows, plant community, and species composition in response to the agricultural land use changes. Another economic-ecological model of water quality is described in Volk et al. (2008), who develop a spatial decision support tool called FLUMAGIS. FLUMAGIS integrates water quality models with ecological and socio-economic information for a catchment in Germany. The ecological assessments were based on macro-zoobenthic community, macrophytes, and a typological classification of watercourses. The economic assessments were limited to analysing the cost-effectiveness of different management options (i.e. benefits were not directly calculated in the model).

The direct financial costs of changes in water resources management may be relatively easy to estimate. However, water resources also support a range of intangible (use and non-use) values, many of which are not captured by market production costs and benefits (Hanley and Barbier, 2009). There are relatively few hydro-economic models that include estimates of such nonmarket values, even though there are obvious benefits of water resources for recreation, biodiversity, landscape aesthetics, etc. Although not entirely uncontroversial (Shabman and Stephenson, 2000), economic valuation techniques can be used to estimate nonmarket values so that they can be incorporated into decision making. An example of a hydro-economic model that integrates nonmarket values is the ChREAM project (Bateman et al., 2006a, 2006b), which examines the agricultural costs and nonmarket benefits associated with implementing the European Water Framework Directive (WFD).

The project involved multiple disciplines, who developed a suite of models to integrate hydrological, economic, agronomic, and geographical elements to analyse effects on water quality, farm revenues and the less tangible, but potentially important, nonmarket values (Bateman et al., 2006b). Hydrological models and spatially explicit models of land use change provided input into ecological modelling of water quality conditions. The researchers used choice experiments (Bateman et al., 2009) and travel cost surveys (Bateman et al., 2010) to estimate the nonmarket value impacts of water quality changes. These values were integrated into a hydro-economic model that showed that the nonmarket benefits of implementing the WFD might be substantial, although the model does not compare these benefits against the cost of interventions.

The study presented in this paper advances hydro-economic modelling by integrating hydrology, ecology and nonmarket valuation in one framework that directly incorporates system uncertainty. Several modelling tools used to predict the impacts of catchment management changes. Water quality changes, predicted by a detailed, spatially explicit, hydrological model, are linked to changes in ecological conditions and nonmarket values. A choice experiment survey was used to link indicators of ecosystem change to nonmarket economic values. Management activities, hydrological changes, ecosystem conditions and nonmarket values are integrated in a Bayesian Network model (BN) that accounts explicitly for uncertainty.

The next section of this paper presents the case study area, choice of environmental indicators, and management scenarios considered. The different modelling methods that are used in this research are described in Section 3. Section 4 details the water quality, ecology and economics model components. How these models were linked into one integrated hydro-economic model is described in Section 5. The final section summarises the main findings, and discusses some of the lessons learned from this study.

2. Case study area

The study described in this paper developed an integrated hydro-economic model for catchment management changes in Tasmania. The George catchment was selected as a study area, through a collaborative process involving biophysical modellers and economists.

The George catchment covers an area of approximately 56,000 hectares in the North-East of Tasmania (Figure 1). The total length of rivers and streams is about 113km. The George Rivers flow into the Georges Bay estuary (2,200ha), which is an important aquaculture area for oyster production (DHHS, 2008). About 2,500 people live in the catchment, with the main town being St Helens (ABS, 2012).

Land use is dominated by native forests, forestry plantations, and agriculture. Most conservation areas and plantations are located in the hilly upper catchment, while the lower catchment contains mostly agricultural and residential areas. The national parks in the catchment are a popular holiday destination, and Georges Bay is intensively used for recreational activities including boating, swimming, sailing and recreational fishing.

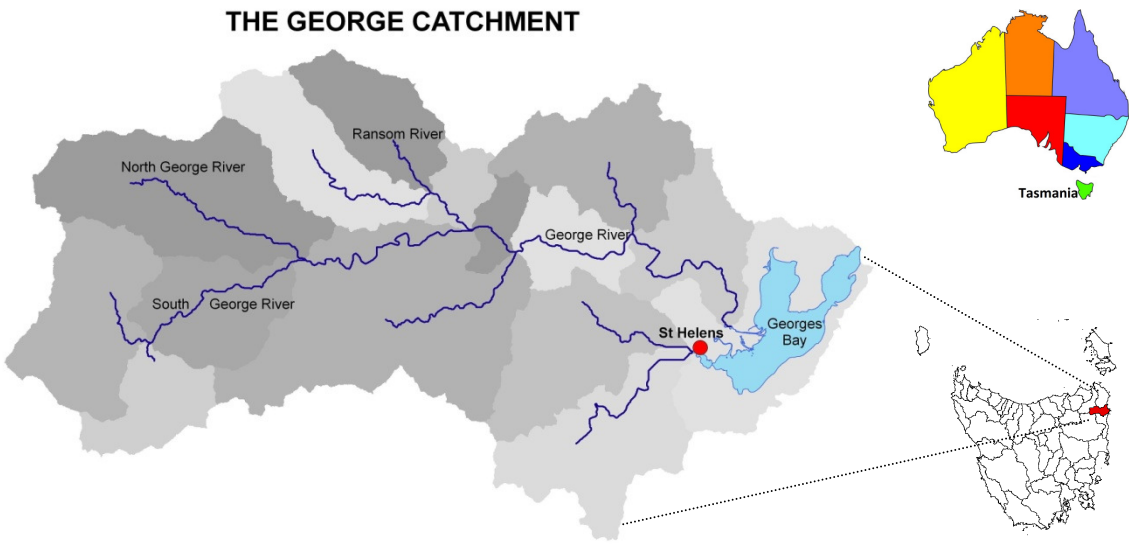


Figure 1 Location and principal rivers of the George catchment study area, Tasmania, Australia

The main threats to environmental conditions in the George catchment are runoff from dairy farms, erosion and pollution from forestry operations, and urban pollution (NRM North, 2008b; NRM North, 2008a). Local management aims to prevent environmental degradation in the George catchment. Management activities include limiting stock access to rivers, removing weeds along river banks, developing riparian buffer zones, recovery of dairy effluent, and improving wastewater treatment. The management scenarios that were specified in the hydro-economic model are: (i) Stream-bank engineering works; (ii) Riparian zone management through revegetating buffer zones; (iii) Changed catchment land use; and (iv) Vegetation management through weed removal. Including actions that were already implemented in the George catchment on a small scale was expected to increase the plausibility of the scenarios in the nonmarket valuation survey.

A conceptual influence diagram was developed to define the scale and scope of the system that would be modelled. The multi-disciplinary modelling process (described in more detail in Kragt et al., 2010) involved three workshops with Tasmanian scientists and 31 structured interviews with experts on river health, threatened species, bird ecology, forestry management, riparian vegetation, estuary ecology, and local natural resource managers. This extensive process ensured that the considered variables, and links between variables, matched the scientific and policy context of the system. The geographical scale of the model was based on the contours of the George catchment, delineated using digital elevation models. A projection of changes in the next twenty years was considered an appropriate time frame from both a biophysical and socio-economic modelling perspective. The model development was an iterative process, aimed at identifying a parsimonious model that would represent the interactions between catchment management actions and environmental variables that impact human welfare (Kragt and Bennett, 2011). Three main ecosystem indicators were used as attributes in the CE survey and as output nodes in the integrated Bayesian Network: native riparian vegetation, number of rare native species and the area of seagrass in the estuary. The conceptual integrated model is included as Appendix I.

3. Modelling techniques

Several modelling methods were used to construct the integrated hydro-economic model. The water quality modelling, Bayesian Network modelling, and the nonmarket valuation techniques are described in this section.

3.1 Water quality modelling

A physically based, semi-distributed model was developed to predict river flows and the delivery of nutrients and sediments in the George Rivers. The water quality model was based on the Catchment Scale Management of Diffuse Sources framework (CatchMODS - Newham et al., 2004). CatchMODS is based on a node-link structure where loadings from upstream sub-catchments provide inputs to the downstream reaches. Data input requirements include climate and associated hydrologic factors, catchment topography, land use, and nutrient concentrations in the various soil types of the George catchment. Physically-based sub-models simulate the hydrological processes, and sediment and nutrient export. These are linked with additional models of pollutant trapping and decay to predict average annual flows, sediment delivery, and nutrient loads into receiving waters (Figure 2).

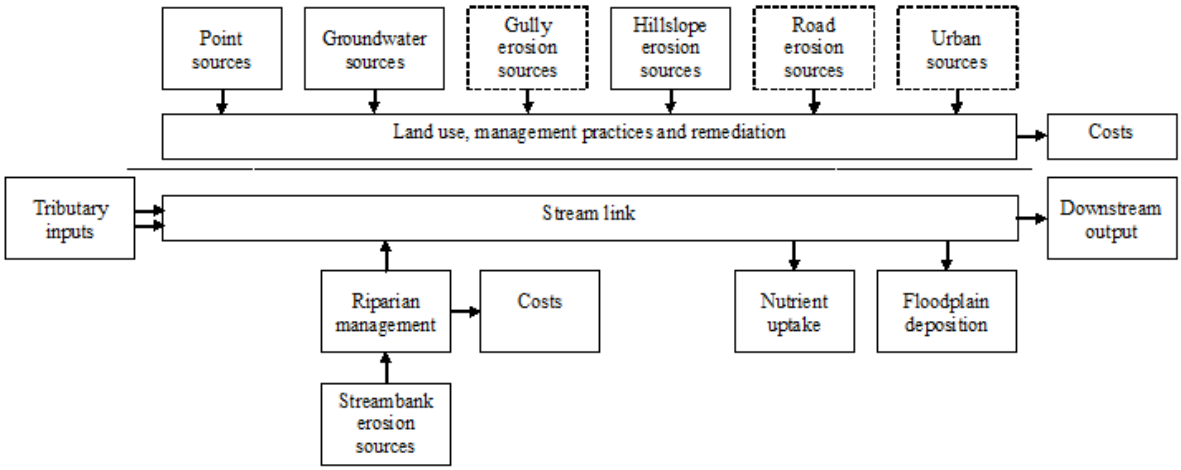


Figure 2 CatchMODS reach structure

CatchMODS requires a relatively small number of parameters and the framework has successfully been tested in other parts of Australia (Newham et al., 2002; Newham et al., 2004; Drewry et al., 2005; Vigiak et al., 2009). Stream reach and catchment data input are coded in ArcGIS software to provide spatially explicit predictions of pollutant loads. The George catchment boundaries were constructed using the 25m digital elevation model (DEM) for Tasmania (3rd ed.), with the Georges Bay defined as the catchment endpoint. Sub-catchment boundaries were delineated using an area threshold of 30km^2, resulting in fifteen sub-catchments for the George catchment.

Climate data was available for the period 1957-2006. Daily maximum temperatures were obtained from the Bureau of Meteorology for gauges at the St Helens Post Office and St Helens Aerodrome. Daily rainfall data was available from seven rain-gauges in the George catchment (BOM, 2012). These observations provide average annual rainfall for each sub-catchment. Land use data for the George catchment were sourced from digital mapping by the Bureau of Rural Science (BRS, 2003). These data were validated to aerial photography and TasVeg mapping (DPIW, 2005b), which lead to the selection of six land use categories in the catchment (native forest for conservation, native forest for production, forestry plantations, agricultural cropping land, agricultural grazing

lands, urban areas). The Revised Universal Soil Loss Equation (Renard et al., 1997) was used to estimate hill-slope erosion, while stream-bank erosion was modelled as a function of empirical parameters, stream bank height and the length of actively eroding sites. The nutrient delivery estimates are based on observed relationships between suspended sediment, nutrient concentrations and stream characteristics. The model was calibrated using literature values, river water quality and flow data for the Ransom River and the George River (Kragt and Newham, 2009). Results of George CatchMODS are described in Section 4.1.

3.2 Bayesian Network Modelling

This study aimed to combine information about multiple biophysical and socio-economic systems into a logically consistent framework. This data was collected from a range of sources, which needed to be combined in one integrated model. A further challenge was the representation of knowledge uncertainty and environmental uncertainty, and the interactions between them. In this study, Bayesian Network modelling was used to link the information about the various catchment systems in a single modelling framework.

Bayesian Networks (BNs - sometimes called belief network or Bayesian Decision Networks) are increasingly used to model natural systems, due to their advantages to incorporate uncertain information and the relative ease in which they can accommodate missing or qualitative data (Castelletti and Soncini-Sessa, 2007; Beresniak et al., 2012). BNs are probabilistic graphical models, consisting of a directed acyclic graph of variables (called 'nodes'). The values that each variable ('node') can assume are ordered into discrete, mutually exclusive, 'states'. These states can be expressed quantitatively (e.g. <50, 50-150, 150-300, >300mg/L) or as qualitative categories (e.g. 'decrease', 'no change', 'decrease'). BNs can integrate data from different sources, including expert opinion when observational data is not available (Pearl, 1988; Jensen, 1996). Their causal graphical structure means that networks can reasonably easily be understood by non-technical users and stakeholders.

BNs differ from most integration approaches by using probabilistic, rather than deterministic, expressions to describe the relationships between variables (Borsuk et al., 2004). In BNs, the links between variables are described as conditional probability distributions. The models rely on Bayes' theorem of probability to propagate information between nodes through these probability definitions. These probabilities account for uncertainty in states and relationships between variables. A basic BN modelling example is shown in Figure 3. In this figure, there are two management actions, three intermediate system nodes, and two outcome nodes. The state names and probabilities shown are purely chosen for illustrative purposes. For example, the economic system node can assume five different states. The probability that each of these states occurs is captured by the discrete probability distribution, which is conditional on the values of the management variables. The values of the outcome variables will be determined by the combined states and probabilities of the three system nodes.

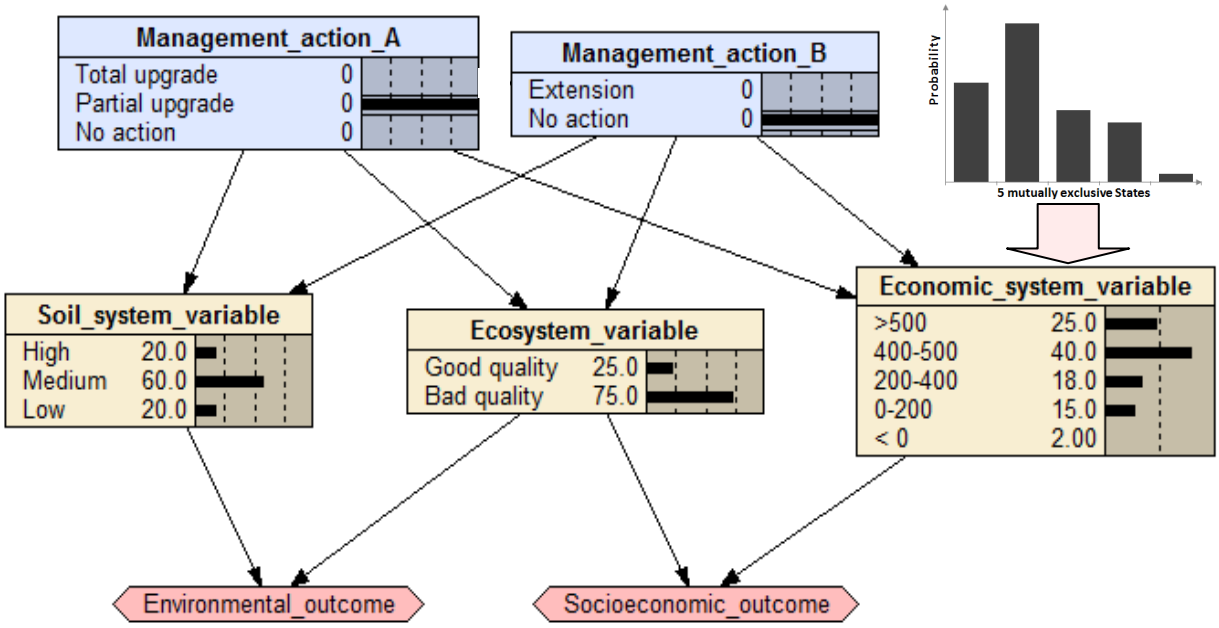


Figure 3 Bayesian Network modelling basics

BNs are widely used for knowledge representation and reasoning under uncertainty in environmental systems. Applications of BNs include, for example, simulation models of water quality and fish populations (Borsuk et al., 2004; Pollino et al., 2007); decision support tools for coastal lake management (Ticehurst, 2008; Ticehurst et al., 2008) or stream protection measures (Bryan and Garrod, 2006); groundwater management (Henriksen and Barlebo, 2008; Carmona et al., 2011; Molina et al., 2013); and catchment-wide assessments of agricultural non-point-source pollution (Dorner et al., 2007).

3.3 Nonmarket valuation

There are relatively few BN applications that assess the economic value impacts of environmental changes. Although most recent decision support tools will include income or management costs (e.g. Carmona et al., 2011; Molina et al., 2013), BN models that incorporate nonmarket costs and benefits of management changes are rare. Exceptions are described by Ticehurst et al. (2007) and Barton et al. (2008), who developed BN models to link information about lake water quality to the economic benefits provided by water. Ticehurst et al. (2007) used literature review and benefit transfer, rather than targeted case studies, to estimate recreational and amenity values associated with coastal lakes in New South Wales, Australia. In Barton et al. (2008), nonmarket values were sourced from an existing 1994 contingent valuation (CV) study in the same catchment. In the CV study, local households were asked for their willingness-to-pay for improving lake quality from currently "poorly suited", to "well suited" for bathing, boating, fishing and drinking water. A drawback of using CV is that the value estimate for water suitability is a binary variable (i.e. only two BN states are assessed for one node). An estimation of marginal value changes (i.e. multiple states) for several environmental attributes (i.e. multiple nodes) is not possible with the CV approach.

In this study, the values of changed catchment management were estimated using a choice experiment (CE) nonmarket valuation survey. In a CE survey, respondents are shown a series of choice questions that each describes the outcomes of hypothetical policy scenarios as impacts on several non-marketed attributes and a cost attribute. Respondents are asked to choose their

preferred scenario in each choice question. It is assumed that, in making their choice, respondents make a trade-off between the attribute levels. The CE technique can be used to estimate willingness to pay for environmental assets, including aquatic systems (Birol et al., 2009; Brouwer et al., 2010; Beville et al., 2012). A CE was considered the most appropriate valuation technique for this study, as it enables estimates of marginal values for separate environmental attributes. These marginal values can then be linked to the output nodes in the BN model, with node states directly corresponding to the CE attribute level ranges

The CE survey was developed using a combination of literature review, interviews with science experts and regional natural resource managers, biophysical modelling and feedback from various focus group discussions (Kragt and Bennett, 2011). Specific care was taken to align the CE attributes and the indicators in the integrated model. The final attributes and their levels are shown in Table 2.

4. Modelling results

Sub-models were developed for the separate systems that would be linked in the integrated hydro-economic model. In this section, the results of each modelling effort are briefly presented.

4.1 Catchment hydrology and water quality

The process-based George-CatchMODS water quality model was used to predict the impacts of the management scenarios (Section 2) on steady state average mean annual river flow (MAF in '000 ML/year) and steady state average loadings of total suspended sediment (TSS in tonnes/year), total phosphorus (TP in tonnes/year) and total nitrogen (TN in tonnes/year) to the George catchment streams and estuary. The model further allows predictions of dissolved and particular N and P, as well as additional management changes (such as removing point sources of pollution), but those results are not presented here.

In this section, only selected results for the base case scenario are provided. This is a scenario with current land use and no change in catchment management.[1] George-CatchMODS was run over a 49-year period to identify the most important sources of sediment and nutrients. The predicted flows, sediment loads, and total nutrient loadings for the George River sub-catchment are shown in Table 1. Estimates of flow, TN, and TP loadings from the George River into the Georges Bay are in line with the observed nutrient concentrations at the George River stream-gauge, providing confidence in the model.

Table 1 Water quality model results for sub-catchment outlet of George River into Georges Bay

Sub-catchment outlet	Variable	Observed	Predicted
George River at St Helens	Mean annual flow (*ML/yr*)	195,530	191,209
	Suspended sediment (*t/yr*)	na	7,356
	Total Nitrogen (*t/yr*)	101	108
	Total Phosphorus (*t/yr*)	5.76	5.50

Results for each sub-catchment are presented as a map that shows which sub-catchments contribute most to sediment and nutrient loadings. The sub-catchments in the hilly West and middle

[1] Results of additional scenarios and detailed descriptions of CatchMODS are available in Kragt and Newham (2009).

regions of the catchment have high base flow volumes, which means less sediment deposition and a larger volume of sediment (and associated pollutants) loading downstream reaches. The North, South, Upper and Mid George sub-catchments were identified as critical sources of nutrients, and are thus priority areas for remediation works. These sub-catchments have a relatively high proportion of agriculture, which is likely to contribute to the nutrient runoff. The model was also used to assess how management interventions in different sub-catchments would reduce pollutant loads (see Kragt and Newham, 2009 for more detail).

4.2 Ecosystem changes

The ecological modelling in this study focused on predicting the states of the three environmental output nodes (native riparian vegetation, number of rare native species and the area of seagrass in the estuary) in 20 years' time, under the different catchment management scenarios.

A specific challenge in the ecological modelling was the virtual absence of quantitative scientific studies and limited long-term monitoring data about ecosystem changes. Because of the imperfect knowledge and limited information about ecosystem functioning in the George catchment, no deterministic simulation models could be developed. Instead, probabilistic BN models were developed for each ecological sub-system to predict changes in riparian vegetation, native species and seagrass area (see Figure 5 for an example).

Data input for the riparian vegetation network was derived from digital vegetation mapping (DPIW, 2005b), river health modelling (DPIW, 2005a), interviews with local NRM officers, agricultural and forestry practitioners and natural scientists. Native riparian vegetation was defined as percentage of total riparian zone in the George catchment with intact vegetation, consisting of at least 70 per cent native vegetation (Table 2). The current percentage of riparian zone with native vegetation is approximately 65%, or 74km, of the total river length in the George catchment.

Table 2 Definition and predicted states of the ecosystem variables in the BN model and choice experiment (CE) survey for the George catchment (source: Kragt et al., 2011)

Attribute	Description	States in Bayesian Network	Matching CE attribute levels
Seagrass area	The area in hectares of dense seagrass (*Heterozostera tasmanica* and *Zostera muelleri*) beds mapped in the estuary	< 490 ha 490 – 620 ha 620 – 760 ha > 760 ha	420 ha 560 ha **690 ha** 815 ha
Rare native animal and plant species	The number of different native Tasmanian flora and fauna species listed as vulnerable, endangered or critically endangered listed under Tasmania's *Threatened Species Protection Act*, with more than one observation in the Natural Values Atlas (DPIW, 2008)	< 40 sp. 40 - 60 sp. 60 - 70 sp. > 70 sp.	35 sp. 50 sp. 65 sp. **80 sp.**
Native riparian vegetation	The length of total riparian zone in the George catchment with intact vegetation, of which at least 70 per cent is native vegetation	< 40 km 40 - 60 km 60 - 70 km > 70 km	35% / 40km 50% / 56km **65% / 74km** 75% / 84km

[a] Currently observed levels in bold. In the CE survey, the levels of native riverside vegetation were presented as the % of total river length as well as the km absolute length

Information about rare species was obtained from the Natural Values Atlas (DPIW, 2008) and through consultation with flora and fauna experts at the DPIW Threatened Species Unit. In total, 68 rare flora species and 34 rare fauna species have been listed in the George catchment. Not counting migratory species, about 80 rare species could be directly affected by local environmental management such as land use change, riparian revegetation or weed management.

The area of healthy seagrass beds in Georges Bay was assessed using monitoring data and digital mapping of the Bay (Mount et al., 2005). The current area of healthy seagrass beds is approximately 690ha, or 31% of the total estuary area. Changes were predicted using estuary models developed for mainland Australia (Baird et al., 2002)and expert opinion. Estuary scientists stressed that accurately predicting seagrass' ecological responses to water quality changes will require further modelling of estuary hydrodynamics and detailed mapping of changes in seagrass beds in the Georges Bay.

The likelihood that different management scenarios result in a change in the ecological variables is captured by a conditional probability distribution for each node. These conditional probability distributions were specified using a combination of empirical observations, biophysical modelling, and semi-structured interviews with with natural scientists and local NRM officers. (Merritt et al., 2006) and (Ticehurst et al., 2008) used similar expert knowledge elicitation processes. Experts would first review the BN sub-model for the ecological asset under consideration. In the interviews, experts answered a questionnaire in which several sets of management scenarios and catchment conditions were specified. They were asked to indicate the likelihood that a certain state of the environment would be observed for each scenario. The interviews would start with probabilities for 'worst case' and 'best case' scenarios, and progressively work through more complex scenarios. Although such an elicitation process may not yield detailed probability distributions, it provided a reasonable approach to populate the integrated BN model with limited existing data and within the time-frame of the study.

4.3 Economic costs and benefits

The CE survey was administered in Hobart, Launceston and St Helens between November 2008 and March 2009. The econometric analysis of the data has previously been published in, for example, Kragt and Bennett (2012) and Kragt (2013).

The status quo scenario that was presented in the CE survey was the predicted 'worst case' scenarios for each of the attributes (for example, "if we don't take any management actions, native riparian vegetation will degrade to 40km in 20 years' time"). The nonmarket values for changes in each of the ecosystem indicators are estimated against this status quo point of reference, using mixed logit models. CE results indicated that Tasmanian households were willing to pay about $3.91 for every km increase in native riparian vegetation, $0.11 for every hectare increase in seagrass area; and $8.62 per number of threatened species protected (Kragt and Bennett, 2011).

5. Bayesian Network Integration

The main objective of this study was to explore a method for integrating hydrological and ecological modelling with economic data in a single, comprehensive framework. The study goes beyond simply linking the outputs from multiple single-disciplinary models. The water quality, ecological and economic models were 'translated' into Bayesian networks, resulting in one integrated hydro-

economic model. In this section, the techniques used to define the linkages between the various model components are described. The integrated model will be demonstrated for a sub-section of the model: the impacts of management actions on water quality, native riparian vegetation, and nonmarket values.

5.1 The water quality network

The complex, process-based George-CatchMODS model was converted into a BN sub-model. Figure 4 shows the simple structure of the water quality BN (without intermediate nodes). Changes in management actions (engineering, revegetation or land use changes) affect flow and water quality through stream-bank erosion, the length of riparian buffer that serves as a nutrient and sediment trap, and runoff from various land uses. The BN structure does not represent all the hydrological processes modelled in CatchMODS, but focuses only on the management actions and their impacts on water quality. This BN is a simpler representation of the complex water quality model, while still incorporating all the detailed system processes that are modelled in CatchMODS.

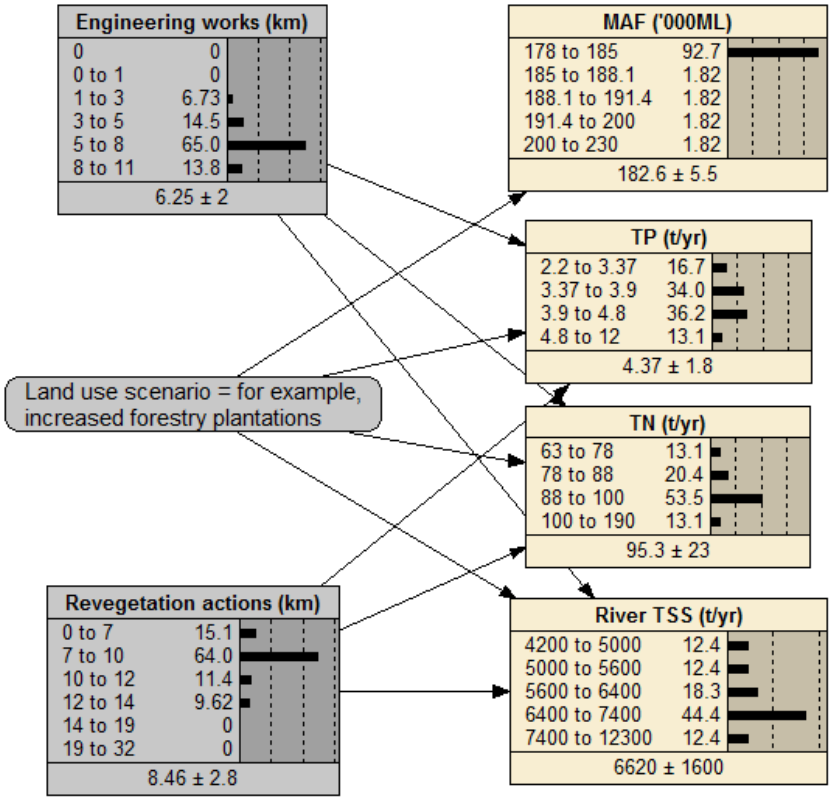


Figure 4 Bayesian Network structure of the water quality sub-model in the George catchment framework

The conditional probabilities that describe the links between catchment management actions and water quality parameters were generated using Monte Carlo simulations of management scenarios in George-CatchMODS. In total, 576 scenarios were run that combined land use changes in individual sub-catchments, with varying lengths of stream-bank engineering works and riparian revegetation in different sub-catchments. The results from these Monte Carlo simulations provided input to define the conditional probability distributions for the water quality variables. Uncertainties in the predictions were specified by varying 19 model parameters along the possible range of values (Kragt

and Newham, 2009). Figure 4 shows an example output of the model. In this example, the land use scenario consists of an increase in forestry plantations, while revegetation and engineering actions are most likely to be between 7-10km and at between 5-8km respectively. Note that this example shows an uncertainty around the length of revegetation and engineering actions taken. The model shows the *most likely* states and probabilities for mean annual flow (between 178,000-185,000ML), total phosphorus (between 3.9-4.8 t/yr), total nitrogen (between 88-100 t/yr) and river total suspended sediment (between 6,400-7,400 t/yr).

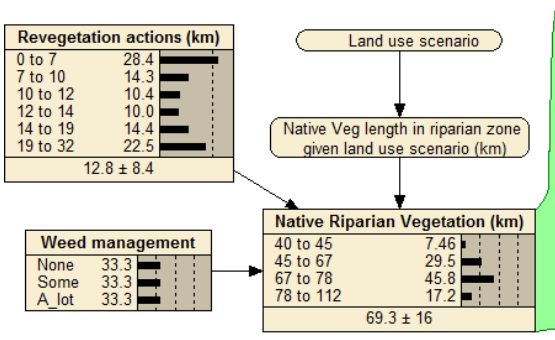
5.2 The ecological network

Based on the ecological modelling process described in Section 4.2, BN sub-models were created for each output node in the Netica Bayesian Network modelling software (Norsys, 2005). The riparian vegetation sub-network is shown in Figure 5. The output node 'Length of Native Riparian Vegetation' measures the total length of rivers in the George catchment with healthy native vegetation along both sides of the river. The management input that affects riparian vegetation is: land use, establishing riparian buffer zones and weed management.

An intermediate node ('Native Veg in riparian zone given different land uses') was included to measure the length of native vegetation in the riparian zone under different land use scenarios. The proportion of riparian zone that is likely to have native vegetation under each land use scenario was determined using existing Tasmanian digital vegetation mapping (DPIW, 2005a; DPIW, 2005b), site visits, and expert review.

Revegetation actions in the riparian zones will directly contribute to the length of riparian vegetation. The 'nativeness' of the riparian zone was modelled as a function of weed management as follows: (i) Low weed management = 15% of healthy native vegetation; (ii) Medium weed management = 50% of healthy native vegetation; and (iii) High weed management = 85% of healthy native vegetation in the established riparian buffer zones.

Because of uncertainty in natural system responses, land use changes, revegetation actions and weed management will not always lead to the same proportion of native vegetation in the riparian buffer zone. The conditional probability distributions for the 'Native Riparian Vegetation' output node were created by simulating over 1,200 scenario combinations of land use, revegetation and weed management. These simulations were used as a 'case file' in the Netica software (Norsys, 2005) to create a conditional probability table (CPT) for native riparian vegetation length. Part of this CPT is shown in Figure 5.



If state of management activities is:			% Probability of Native Riparian Vegetation (km)			
Reveg actions	**Weed mgt**	**Nat Veg given LU**	**40 to 45**	**45 to 67**	**67 to 78**	**78 to 112**
0 to 7	Some	69 to 74	3.70	3.70	88.89	3.70
0 to 7	A_lot	48 to 60	3.57	89.29	3.57	3.57
0 to 7	A_lot	64 to 67	4.35	82.61	8.70	4.35
0 to 7	A_lot	67 to 69	6.25	37.50	50.00	6.25
....	....	....	....	....	....	....
7 to 10	None	67 to 69	3.45	3.45	89.66	3.45
7 to 10	None	69 to 74	4.17	4.17	87.50	4.17
7 to 10	Some	48 to 60	14.29	57.14	14.29	14.29
7 to 10	Some	64 to 67	6.67	60.00	26.67	6.67

Figure 5 The Bayesian Network sub-model for changes in riparian vegetation length, depicting part of the CPT for the 'Native Riparian Vegetation' node.

5.3 The economics network

The direct costs associated with implementing and maintaining management actions were based on literature values (NLWRA, 2000; Lliff, 2002; Freeman and Dumsday, 2003; Sprod, 2003; ABS, 2006; FPA, 2007; ABARE, 2009). Land use costs were represented as the opportunity costs of changing from one land use into another. The costs of establishing riparian buffer zones and stream-bank engineering works were calculated as the present value of the summed one-off implementation costs and discounted maintenance costs over a twenty year period, using a discount rate of three per cent. Uncertainties in management and land use costs arise from, for example, variable returns to land use, the variety in types of materials used, and uncertainty about the labour time involved in implementation and maintenance. These uncertainties are represented in the model by using a range of uniformly costs, rather than a single value (Table 3). Given the limited number of data-sources and the high levels of uncertainty in knowledge, these predicted costs should be seen as an illustration rather than accurate estimates.

Table 3 Range of land use costs and management costs used as input in the George BN model[a]

Land use ($/ha)	Min.	Max.	**Management costs ($/km)**	Min.	Max.
Native vegetation non-production[b]	0	0	**Stream-bank engineering**	13,167	37,882
Native production forest	156	260	**Riparian buffer zone[c]**		
Forestry plantations	612	1,740	- with low weeding	3,332	13,897
Grazing pastures	-23	220	- with medium weeding	17,927	28,491
Irrigated agriculture	491	546	- with high weeding	61,709	79,571

[a] Values are presented as present values calculated over a twenty year period with a three per cent discount rate; [b] No direct returns from native forests were included in the calculations. However, given that the George catchment is visited by >150,000 individuals each year (Tourism Tasmania, 2008) and the positive forest recreational values found in other studies (e.g. Dyack et al., 2007), the returns from native forest may be considerable. No returns were calculated for urban land use, which covers a very small area of the George catchment; [c] Assuming that creating riparian buffers incurs a one-off establishment costs for fencing, willow removal and provision of alternative watering points, with maintenance costs based on the level of continuing weed management in the riparian buffer zone.

The ecosystems BNs predict the state of the environmental indicators: seagrass, native species, and riverside vegetation—relative to the status quo scenario of no management actions (Section 4.3). The CE study was subsequently used to estimate nonmarket values for each ecosystem attribute, expressed as household willingness-to-pay (WTP) for attribute changes relative to the status quo. Uncertainty in nonmarket values were based on the confidence intervals around the WTP estimates. In the Netica software, the CPT was created by specifying a normal distribution based on mean WTP values, and their estimated standard deviations.

Willingness-to-pay estimates per individual household need to be aggregated to calculate the total nonmarket values for changes in George catchment environmental conditions. A variable was included in the integrated BN, to capture different assumptions about the total number of households that hold a positive WTP for the George catchment environment. These assumptions were based on the number of respondents (832), the sampled locations, and the survey response rate (64 per cent). Three possibilities were accounted for in the model: (i) Only the survey respondents hold positive values; (ii) 64 per cent of all households at the sample locations hold positive values; and (iii) 64 per cent of all Tasmanian households hold positive values. This is shown for the native species example in Figure 6.

The impacts of catchment management on nonmarket values are a joint function of: the likelihood that the ecosystem attributes (output nodes) are in a certain state; the distribution of the

CE estimates; and the aggregation assumptions. Using this approach, the defined uncertainty in the economic estimates (riskiness) is combined with the uncertainty of observing a certain ecosystem condition to give a more complete representation of system uncertainties.

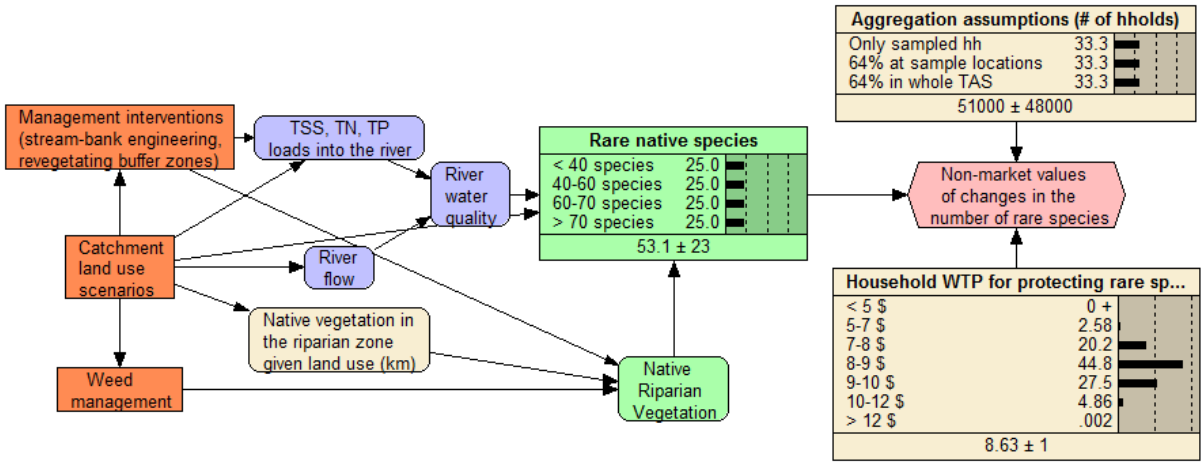


Figure 6 Example sub-BN for non-market values of native species (no management data entered)

5.4 Integrating multiple systems in a hydro-economic modelling framework

The BN modelling approach allows us to integrate the output from different modelling tools and data sources in one framework, and has the advantage of directly accounting for uncertainty. The conceptual integrated model (Appendix I) shows the relationships between natural resource management and management costs, water quality parameters, ecosystem attributes, and nonmarket environmental values. For simplicity, the integration process is illustrated here for a sub-section of the final model, focussing on river water quality and native riparian vegetation changes. Similar analyses can be carried out for the seagrass or native species attributes.

The integrated hydro-economic BN model of management actions, water quality, native riparian vegetation, and nonmarket values is shown in Figure 7. Each of the network variables is described in detail in Appendix II. The cost nodes capture the costs of implementing new management actions. Stream-bank engineering and revegetation are measured in kilometres of actions taken. In the absence of quantitative information, weed management in the riparian buffer zone is defined qualitatively as ‘low’, ‘medium’, ‘high’. Eighteen scenarios were specified for land use. Predefining scenarios reduced the number of possible management combinations, and simplified the model considerably. The environmental conditions include water quality parameters, and native riparian vegetation predictions. The benefits of management interventions are the nonmarket values of native riparian vegetation.

Figure 7 shows the outcomes of an example management scenario. This scenario demonstrates how hydrological modelling predictions, ecological changes and economics are integrated in the model. This demonstrated integration approach can provide valuable information to support decision making, provided that sufficient input data were available. In scenario depicted in Figure 7, no stream-bank engineering works are undertaken. The ‘catchment land use’ node simulates an expansion of production forestry (an important policy concern in Tasmania). Between zero and six kilometres of the riparian zone is revegetated, with high weed maintenance management.

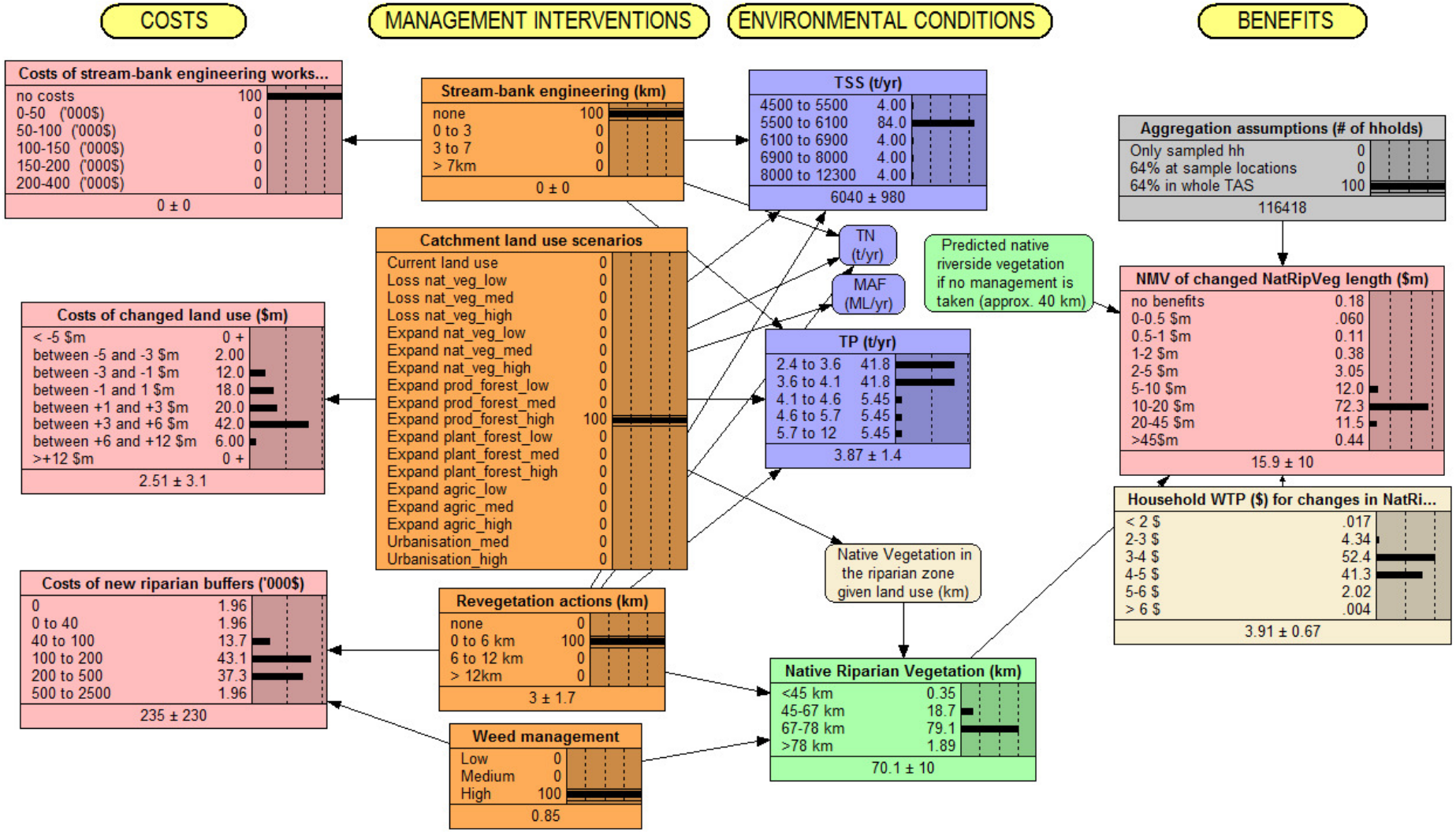


Figure 7 The integrated hydro-economic model for management impacts on water quality, riparian vegetation, costs and nonmarket values (example scenario of no stream-bank engineering, expansion of production forest, 0-6km revegetation, and high weed management)

The blue nodes show the predicted water quality conditions under a 'high expansion of production forest' scenario and no stream-bank engineering. For example, the model predicts that total suspended sediment loads (TSS) are between 5,500 and 6,100 tonnes per year at 84.0 per cent probability. Total phosphorus loads (TP) have a 83.6 per cent probability of being between 2.4 and 4.1 tonnes per year. The low probabilities for the other states of TSS and TP represent the uncertainty in the model's predictions arising from uncertainties in the CatchMODS model parameters (Section 5.1).

Expanding production forests, and establishing between zero and six kilometres of riparian buffer with high weed management, is predicted to result in 70.1 km of native riparian vegetation (79.1% probability that native riparian vegetation is in the 67-78km state). Note that uncertainty in the model leads to an 18.7 per cent probability that the length of native riparian vegetation will be lower: between 45-67 kilometres.

Costs and benefits are shown on, respectively, the left and right sides of the model. The costs of establishing new riparian buffers are approximately $235,000. Uncertainty in the predicted costs is represented in the model by showing a 43.1 per cent probability that cost are between $100,00 and $200,000, and 37.3 per cent probability that costs will be between $200,000 and $500,000. Figure 7 clearly shows the large uncertainties in predicting land use costs. There is essentially a 68 per cent change that land use costs are somewhere between one and twelve million. These large uncertainties were noted in Section 5.3, and reflect the limited information about returns to land use in the George catchment.

The benefits of native riparian vegetation are captured by the node 'NMV of changed NatRipVeg length'. In the scenario demonstrated here, it is assumed that 64 per cent of all Tasmanian households have a positive WTP for riparian vegetation changes. Household WTP is approximately $3.91 per km of native riparian vegetation. The output node shows that, in this scenario, there is a 72.3 per cent probability that the total nonmarket value of the change in native riparian vegetation is between 10 and 20 million dollars. However, there are uncertainties in the predicted length of native riparian vegetation and in household WTP. This result in cumulative uncertainties of total nonmarket values: a predicted probability of 12.0 per cent that the total nonmarket values are between five and ten million dollars, and even a very slight chance that there is no change in nonmarket values. Overall, however, it is likely that the depicted management scenario will result in nonmarket benefits that outweigh the management costs.

6. Summary and conclusion

Integrated water resources management (IWRM) that aims to maximise human welfare needs to be based on sound scientific knowledge and on information about socio-economic systems. The study described in this paper developed an integrated hydro-economic model that allows an assessment of hydrological, ecological and economic values. Acknowledging the interdisciplinary needs of IWRM, this study engaged multiple academic disciplines in the model development prote model was developed in a multi-disciplinary process. Biophysical models were developed to predict the impacts of management actions in the George catchment, Tasmania, on water quality and ecosystem attributes. Choice experiments (CEs) were used to assess the economic values Tasmanians hold for the different environmental attributes in the George catchment. The integrated model was developed using Bayesian Network (BN) modelling techniques. While the integrated model was

developed for a case study area in Tasmania, the techniques are straightforward enough to be applied in other hydro-economic modelling studies.

There are several advantages of using BN modelling techniques for integrated modelling. The visual nature of the BN makes the model a useful tool to communicate the trade-offs between environmental conditions and economic costs and benefits with stakeholders. Using the graphical representation of the system in a BN, experts can relatively easy comment on the included variables and system processes.

The BN can accommodate different data sources in a single framework, thus linking models' predictions, literature values, expert opinion and model assumptions. A BN also provides a straightforward approach to consider knowledge uncertainties about environmental system processes and human-environment interactions, allowing an analysis of the risks associated with management changes. This quantification of uncertain relationships between variables as probability distributions, and the graphical representation of these probabilities, provide a more explicit depiction of system uncertainty than is usually the case in integrated models. Furthermore, the integrated BN model provides an approach to directly capture accumulated model uncertainties.

The CE study conducted as part of this research was developed based on the predictions from the biophysical models. The formalisation of ecosystem processes in BN models provided a better scientific foundation for the CE attribute selection than the approaches that are typically used in environmental valuation studies (Kragt, 2013, in press). An advantage of using CE was the possibility to value changes in multiple attributes on a stepwise scale, making it suitable to directly link values to the BN attribute states.

Notwithstanding the benefits of the BN technique to integrated hydro-economic modelling, several challenges were encountered while developing the model. For example, it was not easy to define a parsimonious conceptual model that captured the variables relevant to all stakeholders, including scientists, economists, decision makers and CE survey respondents. The variables and the definition of their levels needed to be based on sound scientific predictions, while also meeting the requirements of the CE valuation exercise. It took several workshops to reach agreement between the various disciplines. For example, scientists preferred to define environmental processes in detailed ways that were not always comprehensible to respondents. Getting scientists to express environmental conditions in a limited number of attributes posed a considerable challenge.

Another challenge was encountered in defining the conditional probability tables in the ecological networks. Due to a lack in monitoring observations, the changes in ecosystem indicators were predicted through expert interviews. The experts that participated in the interviews found it difficult to express their opinions as a probability. Using a structured questionnaire that guided experts through different scenarios provided a useful tool to defining probabilities.

Finally, the accumulated uncertainty in the input information results in considerable uncertainty in the states of the output nodes. Improved quantity and quality of environmental data is needed to improve the model accuracy. In particular, monitoring data and advanced modelling of ecological systems is required to achieve a more sophisticated representation of the relationships between natural systems changes and impacts on socio-economic systems. Nevertheless, the integrated hydro-economic model demonstrates a promising approach to be used for the development of future decision support tools in integrated water resources management.

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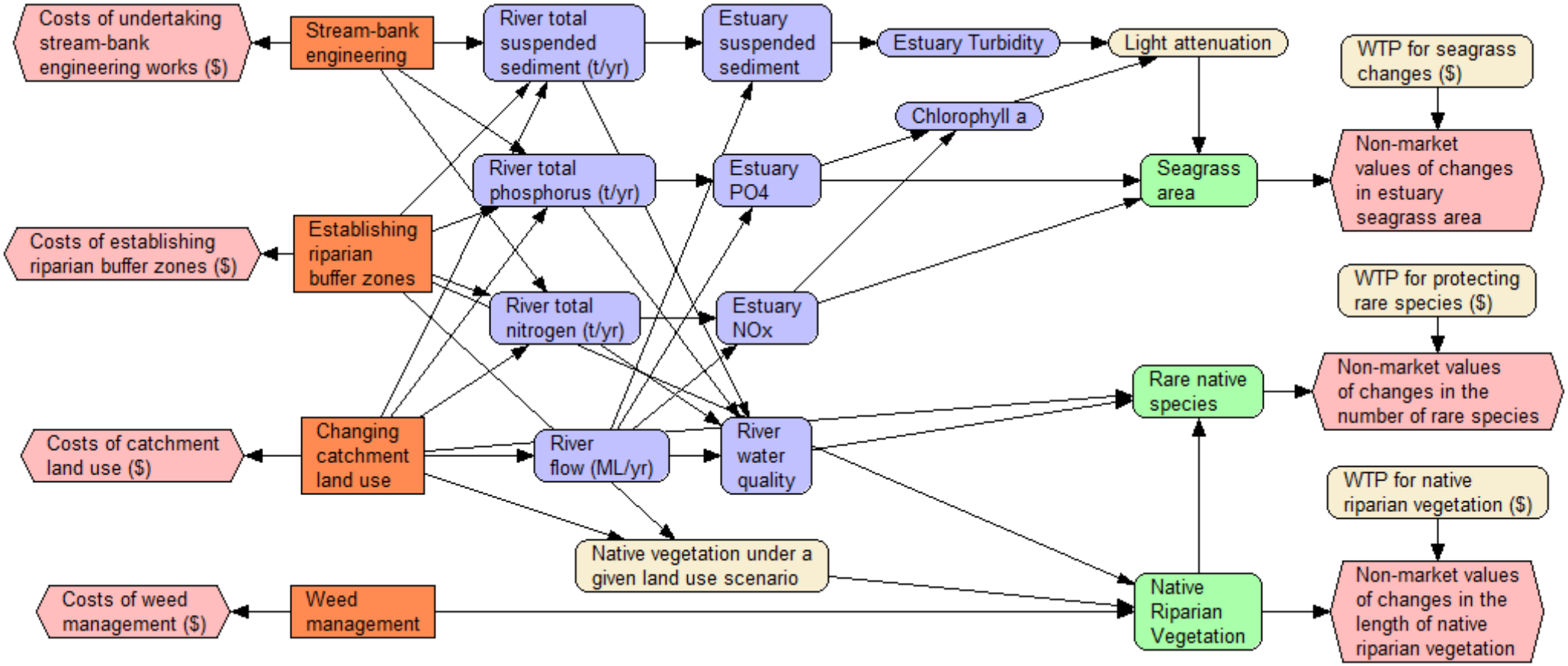
Appendix I – Conceptual integrated BN model for the George catchment



Appendix II – Variables in the integrated BN: native riparian vegetation sub-model

Variable	Description	States	Variable type	Data/information sources
Costs of stream-bank engineering works	Present value of the one-off implementation costs of stream-bank engineering works plus the discounted maintenance costs^	0, 0-50, 50-100, 100-150, 150-200, 200-400 ('000$)	Utility, continuous	Literature values (see Section 5.3)
Costs of changed land use	Total change in gross margins representing value of land use changes in the George catchment	<-5, -5to-3, -3to-1, -1to+1, 1-3, 3-6, 6-12, >12 ($m)	Utility, continuous	Literature values (see Section 5.3)
Costs of new riparian buffers	Present value of the one-off implementation costs of revegetating the riparian buffer zone plus the discounted maintenance costs associated with continuing weed management in the riparian zone^	0, 0-40, 40-100, 100-200, 200-500, 500-2,500 ('000$)	Utility, continuous	Literature values (see Section 5.3)
Stream-bank engineering	Length of stream-bank engineering works undertaken in the George catchment to reduce stream-bank erosion	none, 0-3, 3-7, >7 (km)	Management action, continuous	Observed length of actively eroding sites from George Rivercare Plans (Lliff, 2002; Sprod, 2003).
Catchment land use scenarios	Changes in the total catchment area under alternative land uses (native vegetation non-production, native production forest, forestry plantations, grazing pastures, irrigated agriculture, urban area)	Current land use, loss native vegetation, expanding native vegetation, expanding production forest, expanding plantation forest, expanding agriculture, urbanisation (low, medium, high)	Management action, discrete	Modelling assumptions
Revegetation actions	Length of revegetation of riverside zones to establish buffers on agricultural and urban lands. These buffer zones reduce stream-bank erosion and trap sediment runoff from hill-slope erosion	none, 0-6, 6-12, >12 (km)	Management action, continuous	Modelling assumptions
Weed management	Weed control measures to conserve native vegetation and improve the naturalness of the riparian zone	low, medium, high	Management action, discrete	Australian National Resource Atlas (NLWRA, 2000)
TSS (River total suspended sediment)	Total Suspended Sediments loads into the Georges Bay at St. Helens	4500-5500, 5500-6100, 6100-6900, 6900-8000, 8000-12300 (tonnes/year)	Nature, continuous	Modelled in CatchMODS water quality model
TN (River total nitrogen)	Total Nitrogen loads into the Georges Bay at St. Helens	66-80, 80-90, 90-100, 100-120, 120-220 (tonnes/year)	Nature, continuous	Modelled in CatchMODS water quality model
MAF (Mean annual river flow)	Mean annual total flow from river into the Georges Bay at St. Helens	178-183, 183-188, 188-191, 191-203, 203-230 ('000 ML/year)	Nature, continuous	Modelled in CatchMODS water quality model

Variable	Description	States	Variable type	Data/information sources
TP (River total phosphorus)	Total Phosphorus loads into the Georges Bay at St. Helens	2.4-3.6, 3.6-4.1, 4.1-4.6, 4.6-5.7, 5.7-12 (tonnes/year)	Nature, continuous	Modelled in CatchMODS water quality model
Native veg in the riparian zone given land use	The total length of native vegetation in the riparian zone under alternative land use scenarios	<60, 60-65, 65-70, >70 (km)	Nature, continuous	Calculated in the model, based on assumptions of native vegetation
Native Riparian Vegetation	The total length of native riparian vegetation given land use changes, creation of riparian buffers and weed management	<45, 45-67, 67-78, >78 (km) (equivalent to <40%, 40-60%, 60-70%, >70% of total catchment stream length)	Nature, continuous	Calculated in the model, based on assumptions of native vegetation from expert consultation
Aggregation assumptions	Assumptions on the total number of households in Tasmania with a positive marginal willingness-to-pay	Only sampled households (832 hholds), 64% at sample locations (35,799 hholds), 64% in whole TAS (116,418 hholds)	Nature, discreet	Modelling assumptions based on choice experiment response rate and total number of households in Tasmania
Predicted native riverside vegetation	The total length of native riparian vegetation under a 'status quo' management scenario (degrading land use, no creation of riparian buffers, and no weed management)	Predicted at approximately 40km	Nature, discreet	Modelling assumptions and literature values (see Section 4.3)
Household WTP for change in native riparian vegetation	Household marginal willingness-to-pay for every additional km of native riparian vegetation, *compared to the status quo scenario (= 40km of native riparian vegetation left in the catchment)*	<2, 2 to 3, 3 to 4, 4 to 5, 5 to 6, >6 ($)	Nature, continuous	Equation from choice experiment survey results generating a normal distribution of WTP using estimated mean and standard deviation
Total nonmarket values of changes in native riparian vegetation	The total nonmarket value of changed length in native riparian vegetation in the George catchment, *compared to the status quo scenario*	loss in values, no benefits, 0-0.5, 0.5-1, 1-2, 2-5, 5-10, 10-20, 20-45, >45 (m$)	Utility, continuous	Equation combining parent nodes 'WTP', 'Aggregation assumptions' and 'Native Riparian Vegetation'

^ Discounted at three per cent over a twenty year period