RIVER RESEARCH AND APPLICATIONS

River Res. Applic. **28**: 283–301 (2012)

Published online 24 September 2010 in Wiley Online Library
(wileyonlinelibrary.com) DOI: 10.1002/rra.1456

BAYESIAN NETWORK MODELS FOR ENVIRONMENTAL FLOW DECISION MAKING IN THE DALY RIVER, NORTHERN TERRITORY, AUSTRALIA

TERENCE U. CHAN,[a]* BARRY T. HART,[a] MARK J. KENNARD,[b,c] BRADLEY J. PUSEY,[b,c] WILL SHENTON,[a] MICHAEL M. DOUGLAS,[c,d] ERIC VALENTINE[c,d] and SANDEEP PATEL[c,d]

[a] *Water Studies Centre, Monash University, Clayton, Australia*
[b] *Australian Rivers Institute, Griffith University, Brisbane, Australia*
[c] *Tropical Rivers and Coastal Knowledge (TRaCK), Commonwealth Environmental Research Facility, Australia*
[d] *School for Environmental Research, Charles Darwin University, Darwin, Australia*

ABSTRACT

This paper reports the development and application of two Bayesian Network models to assist decision making on the environmental flows required to maintain the ecological health of the Daly River (Northern Territory, Australia). Currently, the Daly River is unregulated, with only a small volume of water extracted annually for agriculture. However, there is considerable pressure for further agricultural development in the catchment, particularly with demand for extra water extraction during the dry season (May–November).

The abundances of two fish species—barramundi (*Lates calcarifer*) and sooty grunter (*Hephaestus fuliginosus*)—were chosen as the ecological endpoints for the models, which linked dry season flows to key aspects of the biology of each species. Where available, data were used to define flow–fish habitat relationships, but most of the relationships were defined by expert opinion because of a lack of quantified ecological knowledge.

Recent field data on fish abundances were used to validate the models and gave prediction errors of 20–30%. The barramundi model indicated that the adult sub-population was key to overall fish abundance, with this sub-population particularly impacted by the timing of abstraction (early vs. late dry season). The sooty grunter model indicated that the juvenile sub-population dominated the overall abundance and that this was primarily due to the amount of hydraulically suitable riffle habitat. If current extraction entitlements were fully utilized, the models showed there would be significant impacts on the populations of these two fish species, with the probability of unacceptable abundances increasing to 43% from 25% for sooty grunter and from 36% for barramundi under natural conditions.

KEY WORDS: environmental flows; barramundi; sooty grunter; Bayesian network models; decision support; Daly River

Received 16 July 2010; Accepted 29 July 2010

INTRODUCTION

Numerous methods are now available for assessing the environmental flow regime required to sustain or restore the ecological integrity for a particular river system (Tharme, 2003). These environmental flow methods generally attempt to achieve a flow regime similar to that which would have occurred naturally (Poff *et al*., 1997; Hillman *et al*., 2003). Reviews of the range of available methods by Arthington and Pusey (2003) and Tharme (2003) show clearly that these methods rely heavily on expert opinion, often lack the required hydrological data (simulated daily natural flows), and generally lack transparency on how various flow components are related to ecological outcomes (Hart and Pollino, 2009).

The work reported here, and in a companion paper, Shenton *et al*. (2010), builds on the review by Hart and Pollino (2009) that identified the potential for Bayesian Network (BN) models to assist decision makers in assessing ecological risks of different water management scenarios and establishing optimum environmental flow regimes. BN models are increasingly being used in natural resources management (Ellison, 1996; Batchelor and Cain, 1999; Varis and Kuikka, 1999; Nyberg *et al*., 2006; Castelletti and Soncini-Sessa, 2007; McCann *et al*., 2007; Ticehurst *et al*., 2007; Wang *et al*., 2009) and more recently, specifically for determining environmental flow allocations (Hart and Pollino, 2009; Stewart-Koster *et al*., 2010). BN models have a number of properties that make them particularly useful for ecological and environmental management applications. In particular, they show cause–effect relationships directly through a simple causal graphical structure, but are also easily constructed, extended and modified; they have a natural way to handle missing data; they explicitly incorporate uncertainty in relationships; they are an accessible and intuitive modelling approach; they can show good predictive accuracy even with small sample sizes; they allow the conditional probabilities between variables to be constructed using either observed data, other models, or expert knowledge; they can easily be updated as new data

*Correspondence to: Terence U. Chan, Water Studies Centre, Monash University, Clayton, Australia. E-mail: terry.chan@sci.monash.edu.au

become available, improving system representation and predictive accuracy, which is particularly important in systems with little previous monitoring; they are modular, where models are composed of a set of interacting components; they preserve system knowledge and can be used to educate people unfamiliar with the system; and they can be easily used to aid management decision making. Korb and Nicholson (2004) provide an extensive background to BN modelling, examine methods and techniques, and discuss advantages and disadvantages in detail. BN models can be used to integrate flow information and other biophysical factors to produce measurable ecological outcomes (Pollino *et al*., 2007). Additionally, they can also be used to integrate social and economic drivers and management outcomes (Carpenter, 2002; Castelletti and Soncini-Sessa, 2007; Barton *et al*., 2008; Chan *et al*., 2010).

BN models consist of three elements: (a) a set of nodes representing the management systems key variables; (b) a set of links that represent the cause-effect relationship ('conditional dependence') between the nodes and (c) a set of probabilities representing the belief that a node will be in a certain state given the states of the connecting nodes. BN models can be effectively incorporated in a traditional risk management framework through explicitly displaying the causal web of interacting factors and the probabilities of multiple states of predictor and response variables (Nayak and Kundu, 2001; Korb and Nicholson, 2004; Hart *et al*., 2005; Marcot *et al*., 2006; Barton *et al*., 2008). Hart and Pollino (2009) have reviewed the potential benefits and limitations of BNs for environmental flow applications.

There is increasing pressure to develop the apparently large water resources available in the tropical north of Australia, as the rivers in the south of the country become increasingly over-allocated (Hart, 2004; Blanch, 2008). Concern regarding the added pressure of climate change and the current drought on the already flow-stressed southern rivers, is translating to additional pressure to use water resources in the north (Preston and Jones, 2008). The objective of this paper was to develop Bayesian Network models to inform environmental flow decision-making and water allocation planning in the Daly River catchment in northern Australia. The Daly River is currently unregulated, with only a small volume of groundwater extracted annually for agriculture, but with considerable pressure for further agricultural development and water demand, particularly in the vicinity of Katherine and the Douglas–Daly region (Northern Territory Government, 2009; Figure 1). Existing

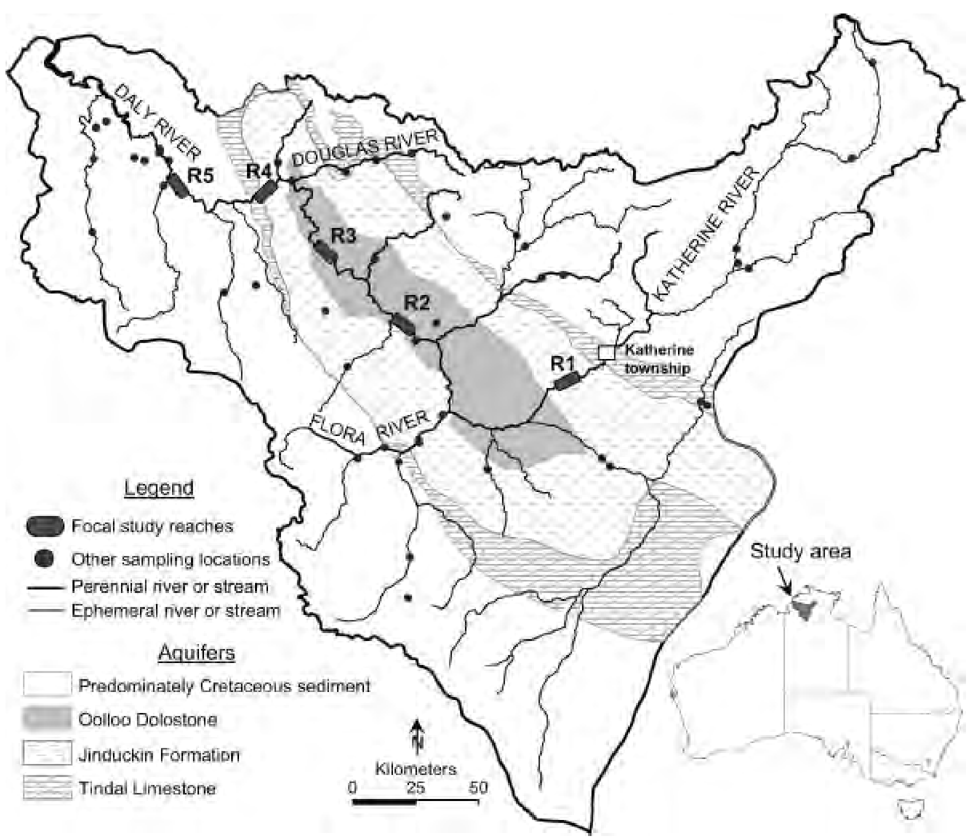

Figure 1. The Daly River catchment, northern Australia. Shown are the locations of major aquifers, the five study reaches along the lower Katherine River and Daly River main channel (R1-R5), and the other 50 field sampling sites.

DOI: 10.1002/rra

agricultural and mining industries, coupled with population growth, already place pressure on the water resources of the catchment (Begg *et al*., 2001) and there is a need to determine how current and future water use will impact the system, and if impacts can be mitigated. Management decisions on the allocation of water resources in the Katherine region are already being made (Northern Territory Government, 2009), despite the currently limited data and understanding of the Daly system (van Dam *et al*., 2008). None-the-less, the Daly River has been identified as being at great risk from multiple threats and pressures (Bartolo *et al*., 2008). The present study seeks to address the environmental water requirements of fish in this system, given the expected changes in river flows from extraction, and attendant ecological impacts.

In this paper, we describe the development and application of BN models that link important flow components with an ecological model to predict the abundance of two important native fish species—barramundi and sooty grunter. These two fish species are important socially, culturally, economically (recreational or commercial fishing) and perform important but different ecological roles in aquatic ecosystems (Pusey *et al*., 2004; Bayliss *et al*., 2008). The BN models used information on modelled changes in dry season flow regimes under various water extraction scenarios, combined with outputs from two-dimensional habitat simulation models of fish species hydraulic habitat requirements and other ecological data and expert knowledge. The models were developed for five key river reaches and validated using fish sampling data collected over a 3-year period. We conclude by arguing that BNs provide an ideal way of combining quantitative data with expert knowledge (where such data are lacking) to inform environmental flow management and planning.

METHODS

Study area and focal study reaches

The Daly River catchment (Figure 1) is 53 000 km^2 in area and comprises mainly tropical savannah woodland. The river and its catchment are considered to be in good condition (Ganf and Rea, 2007), and to be of high ecological value and exhibiting largely intact and diverse vegetation communities in both the catchment and the riparian zones (Faulks, 1998; Gehrke *et al*., 2004; Blanch *et al*., 2005; Douglas *et al*., 2005). The dominant land-uses are low-density cattle grazing (approximately four animals km^{-2}) and conservation (Townsend and Padovan, 2009). In 2008, 5.3% of the catchment had been cleared for more intensive land-uses such as urbanization, pasture and agriculture (van Dam *et al*., 2008; DRMAC, 2009), and substantial areas of native vegetation are at risk because a 6-year moratorium on land clearing in the Daly River region expired in March 2010. Increases in agriculture are of particular interest in the Daly River catchment (Lamontagne *et al*., 2005), as this catchment has the largest area of suitable soil in the region (NTDIPE, 2003). The extensive mineral resources in the region (e.g. gold at Mt Todd) are also a factor in future development (further details are available from DRMAC, 2009). There are no major impoundments on the Daly River system, although there are an unknown number of small farm dams (Begg *et al*., 2001).

Our study focused on a section of the Daly River main stem extending from the lower Katherine River to Mt Nancar (Figure 1). This section was divided into five focal study reaches: R1—the lower Katherine River, R2—Daly River at Dorisvale, R3—Daly River at Ooloo crossing, R4—Daly River at Beeboom Crossing and R5—Daly River at Mt Nancar.

Hydrology and water management

Annual rainfall in the catchment is around 1000 mm, with 90% falling during the wet season months between November and April. As a result, river discharge is highly seasonal, with monsoonal and cyclonic weather producing wet season flows (November–April) of up to 2000 $m^3 s^{-1}$ at Mt Nancar, 75 km upstream of the river's mouth (Figure 2). Rainfall is negligible during the dry season, with flow in the Daly River and its major tributaries supplied predominately from the spatially extensive groundwater inputs from the underlying aquifers associated with the Ooloo, Jinduckin and Tindal formations (Figure 1). The dry season (May–October) constitutes a lengthy period of base flow with infrequent small runoff events (Townsend and Padovan, 2009). Perennial flow distinguishes the Daly River from most other rivers of the wet/dry tropics of northern Australia, which cease to flow for a large proportion of the dry season (Kennard *et al*., 2010).

Water is extracted for consumptive water use directly from the Daly River and its tributaries and from bores that tap into the Ooloo and Tindal aquifers, and occurs mainly in the vicinity of Katherine township and the Douglas–Daly region (Figure 1). Most extractions are from the groundwater for cattle, irrigated agriculture and town and homestead potable water supplies (Townsend and Padovan, 2009). Water for consumptive use has been allocated mainly on a 'first come, first served' basis, but will in the future be allocated according to a plan currently being developed that will take into account the availability of the water resource and requirement for a minimum environmental flow. Widespread water extraction and the spatially extensive nature of the aquifers implies that depletion of groundwater resources could have impacts on river flow over most of the lower Katherine River and main stem of the Daly River (Figure 1).

DOI: 10.1002/rra

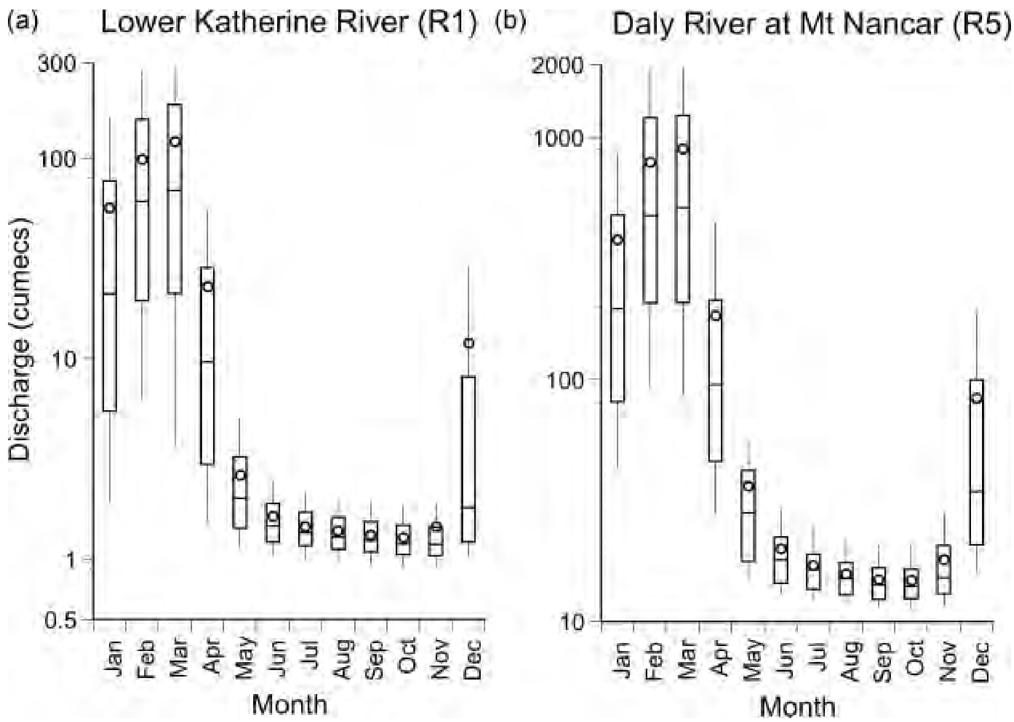

Figure 2. Box plots showing the modelled 'natural' monthly discharge for (a) Katherine River at Galloping Jacks (R1), and (b) Daly River at Mt Nancar (R5). The plots show the range of variation across years (for the 109-year modelling period spanning 1900–2008) in modelled mean daily flow per month. The box represent the 75th percentile, 50th percentile (median) and 25th percentile, the upper and lower bars represent 90th percentile and 10th percentile, and the circle represents the mean.

Water extraction scenarios

Simulated daily discharge data modelled for a 109-year period (spanning 1900–2008) for four water extraction scenarios was available for use in this study. The simulations were provided by the Land and Water Division of the Northern Territory Department of Natural Resources, Environment, the Arts and Sport (NRETAS) and used a finite-element model for subsurface flow and transport, including ground water–surface water interactions (FEFLOW—Trefry and Muffels, 2007). The four water extraction scenarios provided were: 'natural' (i.e. no extraction), 'historic' (i.e. estimated actual levels of water extraction), 'current entitlements (fully utilized)' and 'possible future entitlements', these scenarios are discussed further below. The effects of water extraction on the dry season flow regime are most apparent in the lower Katherine River (R1) and are progressively ameliorated further downstream in the lower Daly River (R5). For example, Figure 3 shows the average annual percentage reduction in mean dry season discharge compared with the 'natural' scenario for two locations (R1 and R5). These data indicate that 'historic' levels of extraction have been relatively minor (reduction in mean dry season discharge for R1 and R5, respectively, from an extraction of 1200 ML $year^{-1}$), but that substantial reductions in dry season discharge would occur if 'current entitlements' were fully utilized (45 and 6% reduction in mean dry season discharge for R1 and R5, respectively, from an extraction volume of 44 700 ML $year^{-1}$). This would be markedly increased under the 'possible future entitlements' scenario (60 and 11% reduction in mean dry season discharge, for R1 and R5 respectively, from an extraction volume of 79 600 ML $year^{-1}$).

Field sampling data

Quantitative sampling of fish communities within the five study reaches in the main Daly–Katherine River channel (Figure 1) was undertaken biannually (early and late dry seasons) over the 3-year study period (2006–2008). This is described more fully in the following paragraph. These data were used to validate the BN models. A further 50 sites situated throughout the Daly River catchment (Figure 1) were sampled once during the study period as part of a broader study on flow and habitat requirements of the fish species in the catchment. These data were used to provide further information on the distribution, abundance and habitat preferences of the resident fish species (see below). Importantly, variation in river flow over the period leading up to and including the field data collection period (2006–2008) was representative of a relatively wide range of flows when compared to the long-term record.

DOI: 10.1002/rra

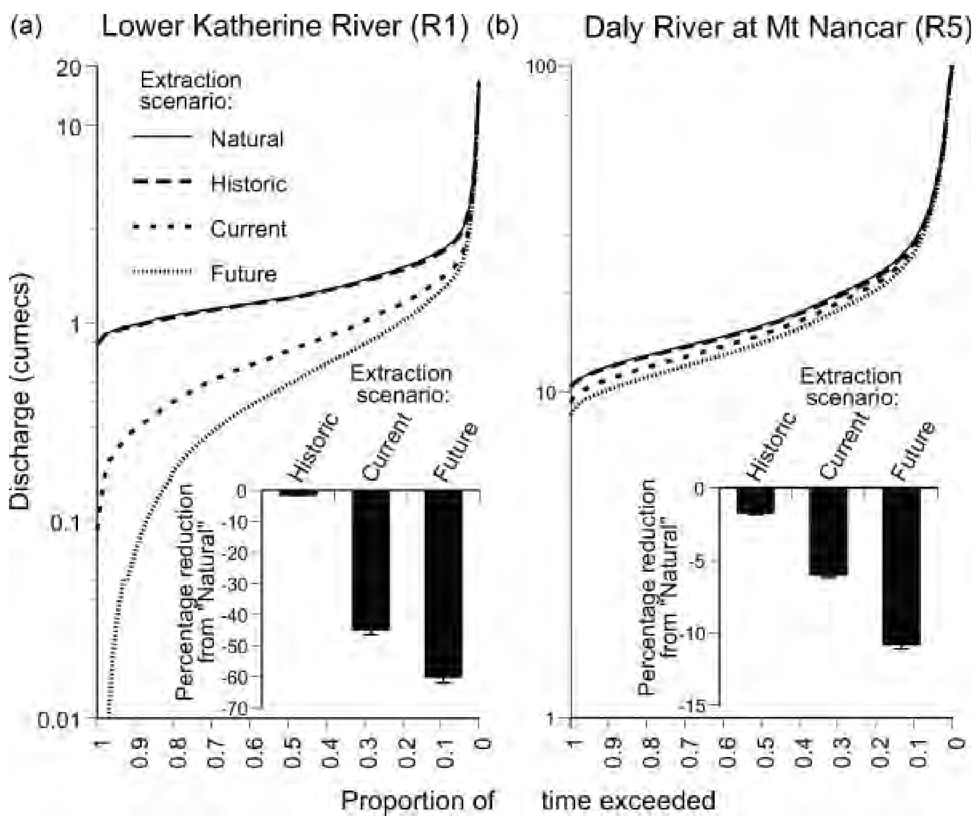

Figure 3. Dry season flow duration curves for (a) Katherine River at Galloping Jacks (R1) and (b) Daly River at Mt Nancar (R5) under four water extraction scenarios. Data are the proportion of time (dry season days) each discharge was exceeded for the 109-year modelling period spanning (1900–2008). Note that the curves for the 'natural' and 'historic' scenarios overlay each other. Inset plots show the average (±SE) annual percentage reduction in discharge for each water extraction scenario compared with the 'natural' scenario.

A detailed description and evaluation of the fish sampling procedures used in this study is provided in Kennard *et al.* (2008) and is briefly summarized here. Within each selected sampling site (500–1000 m in length), fish assemblages were sampled by electrofishing (boat and/or backpack) at multiple discrete locations within each site. These samples are hereafter termed electrofishing 'shots' with each shot fixed at five minutes duration (elapsed time). At least 15 electrofishing 'shots' were usually undertaken at each site as this level of effort provided highly accurate and precise estimates of local fish assemblage attributes (Kennard *et al.*, 2008). Replicate measures of a range of hydraulic and microhabitat parameters were taken for each shot. Fish collected from each electrofishing shot were identified to species level, measured and returned to the approximate point of capture. Estimates of fish abundance (standardized catch-per-unit-effort) and habitat suitability criteria (*sensu* Groshens and Orth, 1994) for each species were determined from these data.

Habitat simulation models

Two-dimensional, depth averaged finite element hydrodynamic models (RMA-10; King, 2001) were developed separately for 9 km of the lower Katherine River (McGarry and Valentine, 2008) and 150 km of the main stem of the Daly River (Patel and Valentine, unpublished data) and were used in this study. The models were calibrated using gauged flows, depth-averaged velocity profiles and measured water levels and were verified successfully for a broad range of dry season flow magnitudes (McGarry and Valentine, 2008; Patel and Valentine, unpublished data). We extracted model outputs of average depth and velocity for polygons situated within each of five focal study reaches under a range of dry season discharge magnitudes. These data were combined with knowledge of fish habitat requirements to estimate changes with discharge in the area of suitable habitat for sooty grunter and barramundi life stages (see Figures 4 and 5).

BAYESIAN NETWORK MODELS

BN models were developed to predict the likelihood of changes in the abundances of two native fish species, barramundi and sooty grunter in response to flow regime changes associated with dry season water extraction.

DOI: 10.1002/rra

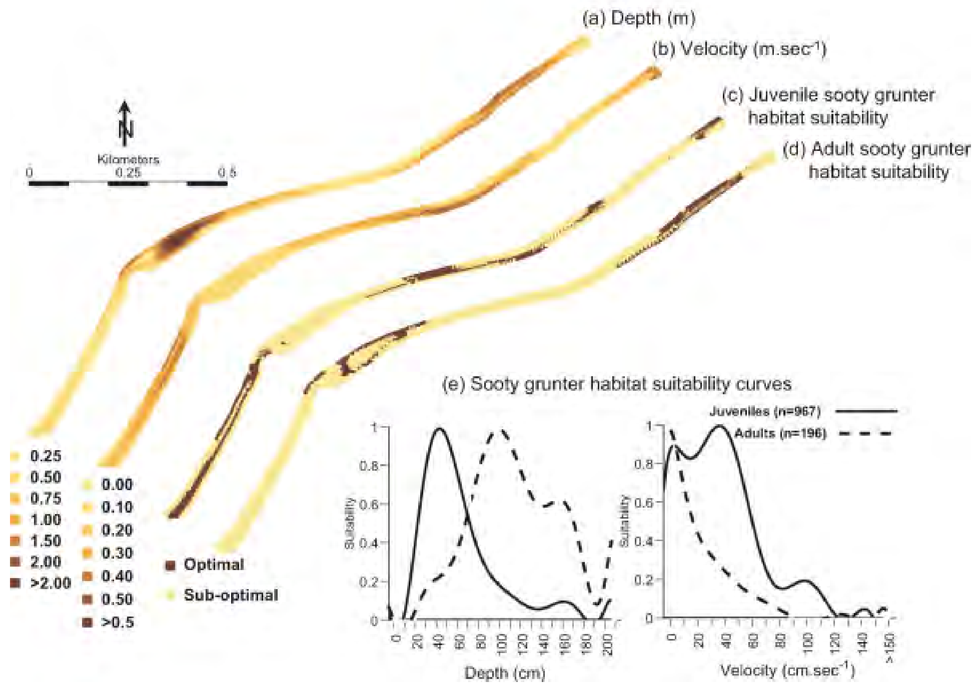

Figure 4. Outputs from the 2-D hydrodynamic model for the lower Katherine River (R1) showing (a) variation in depth and (b) mean water velocity, estimated for a discharge of 1.5 $m^3 s^{-1}$. These data were then used to map habitat suitability for (c) sooty grunter juveniles and (d) sooty grunter adults when combined with habitat suitability curves for depth and water velocity (e). Depths and velocities were considered optimal for suitability >0.7, sub-optimal if suitability was >0 and <0.7 and unsuitable if suitability is 0 (see Groshens and Orth, 1994). This figure is available in colour online at wileyonlinelibrary.com

Changes in fish abundance were modelled at five focal study reaches (Figure 1) under four water extraction scenarios (natural, historic, current entitlement and possible future entitlement).

Ecological endpoints

Two native fish species—barramundi (*Lates calcarifer*) and sooty grunter (*Hephaestus fuliginosus*)—were selected as the endpoints for the environmental flow (eFlow) BN models. Both species are ecologically, economically, recreationally and culturally important. Considerable quantitative data are available for these species. Additionally, they have quite different life histories and ecological requirements, suggesting that different flow components are likely to determine the population dynamics of each species. Fish species are often chosen for environmental flow studies, because their biology is generally related to flow (e.g. Schlosser, 1985; Scheidegger and Bain, 1995) and the presence of healthy fish populations generally reflects a healthy ecosystem (Hocutt, 1981; Whitfield and Elliott, 2002). Additionally, stakeholders and the general community are often more interest in fish than in other potential endpoints (e.g. macroinvertebrates, algae).

Barramundi life history

Barramundi are large (up to 60 kg weight and 1.2 m in length), predatory, long-lived and move extensively between estuarine and freshwater parts of the system. Therefore, they are reliant on the maintenance of connectivity within the riverine system. Adult male barramundi migrate from the freshwater reaches of the river into the estuary and near-shore marine environment to spawn, where the eggs hatch and go through larval stages to become juveniles in their first year. Juveniles then migrate back into the freshwater reaches of the river system, maturing from sub-adults to adults within the third year, by which time they are reproductively mature (Russell and Garrett, 1985; Pusey *et al*., 2004).

Bayliss *et al*. (2008) reported a significant positive relationship between barramundi abundance and wet season flow, as well as a significant lag function relating abundance to the magnitude of the wet season flow 3 years previous as a result of interactions between flow and recruitment dynamics of early life history stages. However, there are still significant unknowns regarding the mechanisms involved. Moreover, much less is known about the relationship between population size and the influence of dry season flows, which may change significantly under proposed extraction scenarios.

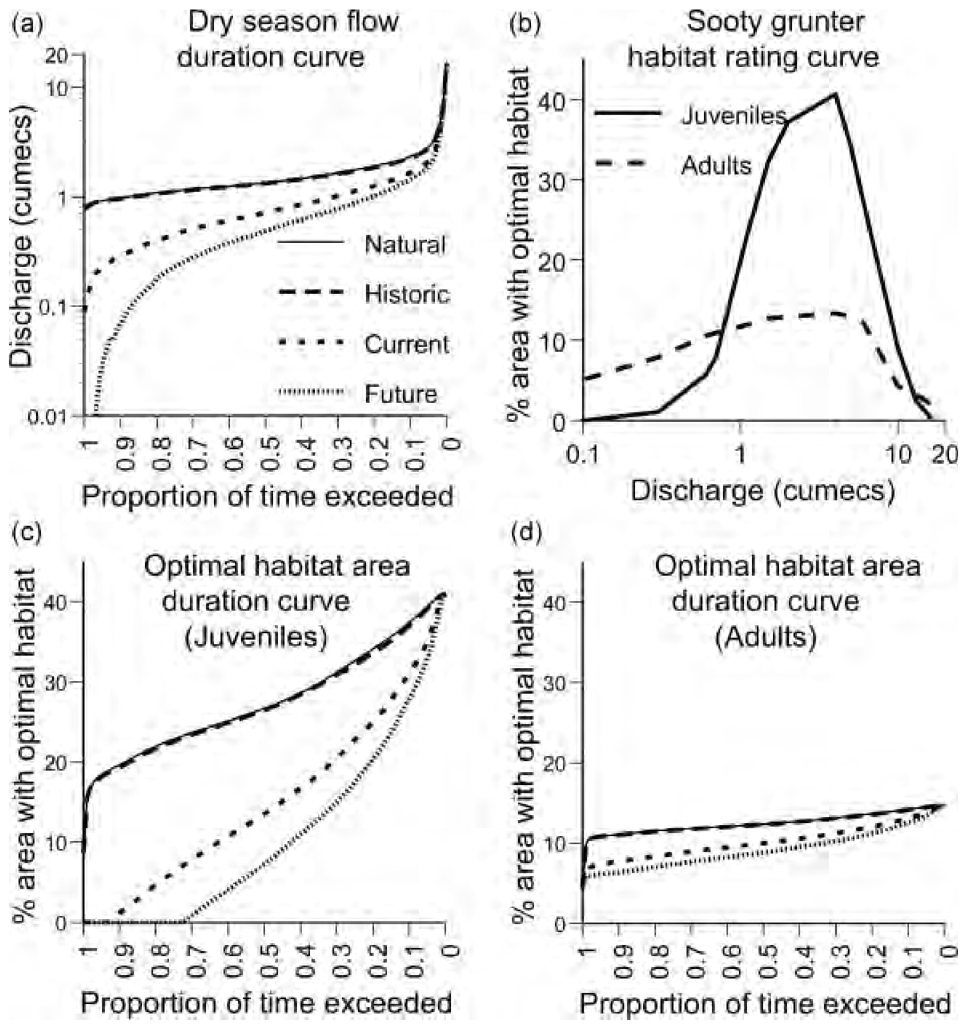

Figure 5. Example of the process used to estimate changes in optimal sooty grunter habitat availability at the lower Katherine River site (R1) under four water extraction scenarios. The dry season flow duration curve (a) was combined with a habitat suitability rating curve (b) (see Figure 4) to estimate optimal habitat duration curves for sooty grunter juveniles (c) and adults (d) under each water extraction scenario. Note that the curves for the 'natural' and 'historic' scenarios overlay each other.

We hypothesize that the survival of barramundi population during the dry season is largely controlled by interactions between availability, quality and longitudinal connectivity of habitat, resource availability (i.e. food and space), and biotic interactions (i.e. predation including cannibalism and competition). In addition, reduced water quality during the dry season (high temperatures and reduced dissolved oxygen concentrations) may result in mortality as well as facilitating the development and transmission of disease (Pusey *et al*., 2004).

Sooty grunter life history

Sooty Grunter is considerably smaller (up to 4 kg and 0.5 m) than barramundi and relatively sedentary (Pusey *et al*., 2004). Very little is known about the environmental and ecological factors responsible for long-term variation in sooty grunter populations, particularly the role of the dry season flow regime. Adults may make limited spawning migrations within freshwater, to access riffle areas. Juveniles primarily inhabit these riffle regions, which provide important feeding (e.g. algae and aquatic insects) and refuge areas (from larger predators). Once they mature into adults, they move into the pools, where habitat structures such as woody debris and bank undercuts (and associated root masses) become important. Connectivity between pools is important as this provides an opportunity to move to more favourable areas if conditions in a particular pool become adverse (Pusey *et al*., 2004).

Conceptual models

Conceptual models relating barramundi and sooty grunter abundance to flow and other physical and biological factors (i.e. habitat, food availability and predation) were initially

DOI: 10.1002/rra

developed during an expert workshop. These initial conceptual models were subsequently refined by further input from fish experts (see Acknowledgements section), with the final versions shown in Figure 6a and b.

The flow component of the conceptual models focused on the dry season (May–October) flows only, as this was the period during which water extraction occurs. We also evaluated if the ecological effects of water extraction

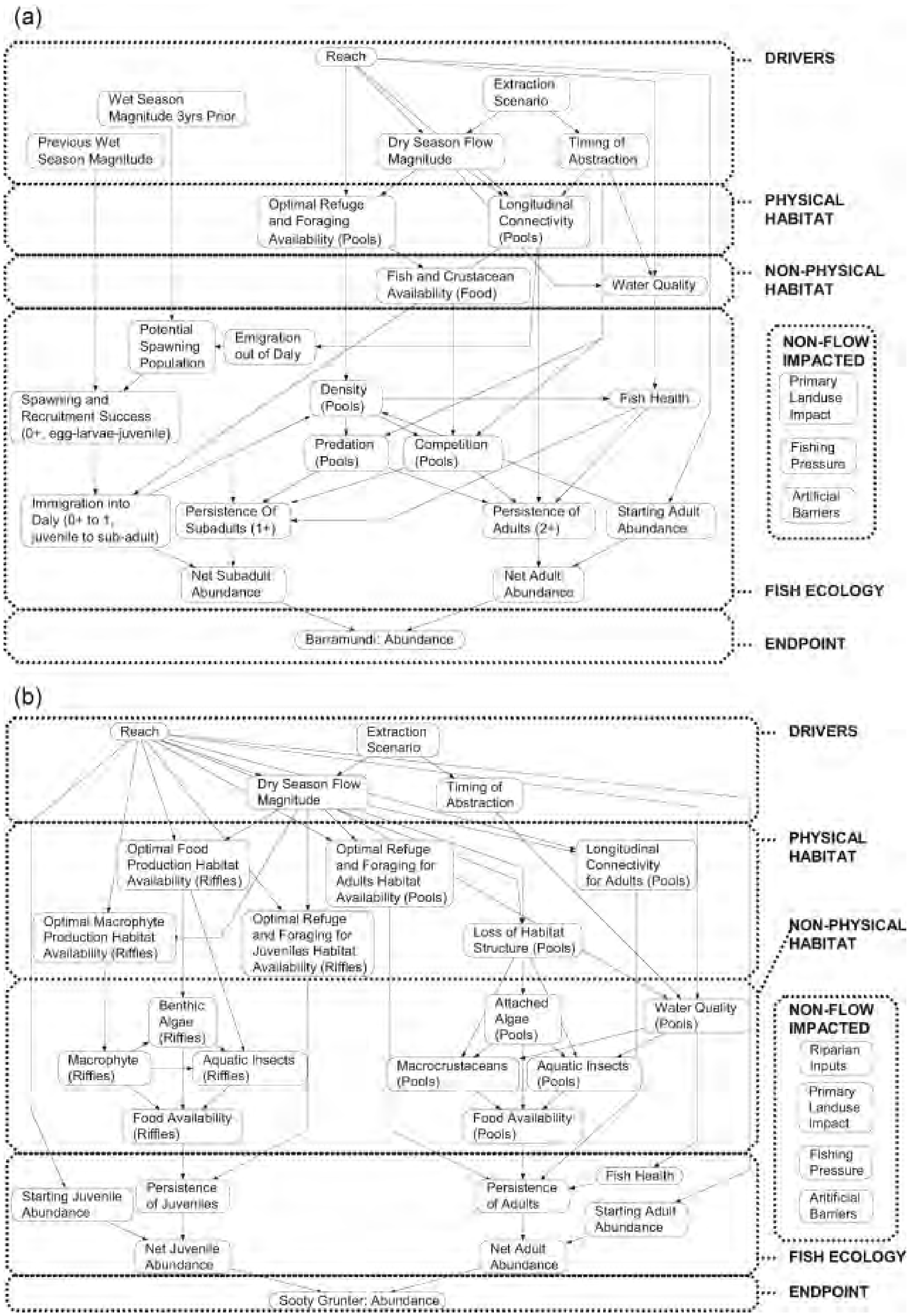

Figure 6. Final conceptual models for (a) barramundi (b) sooty grunter.

DOI: 10.1002/rra

differed depending on the timing of extraction (i.e. early versus late dry season).

The ecological conceptual model was built on the assumption that three ecosystem aspects connected fish abundance to flow, these being:

(a) hydraulic-habitat variables, including habitat suitability (depending on water depth and velocity preferences) and longitudinal connectivity,
(b) variables that are indirectly affected by flow, such as water quality and food production,
(c) species-specific ecological processes such as reproduction, feeding, movement, competition and predation.

These relationships are discussed in more detail in the 'Ecological sub-model' section.

Bayesian models

The two species-specific conceptual models were then developed into Bayesian network models. The *spatial scale* of the models was laterally restricted to the main channel (i.e. floodplain excluded), and longitudinally extended from the Katherine to the Mt Nancar field sites (Figure 1), as this encompasses the region where current water extraction is occurring and may be expected to increase in the future. The BN models were constructed such that the impacts of water extraction scenarios on barramundi and sooty grunter populations could be evaluated for the five key reaches in the Daly–Katherine River system (Figure 1). The *temporal scale* was restricted to one dry season, although the Barramundi eFlow BN model has additional nodes related to the previous wet season and the wet season 3 years previous.

Detailed definitions of each node and the causal linkages from each are listed in Appendix 1 and 2. There are also a number of variables that are likely to be important to the biology of these two fish species, but are not linked into the networks because they are not directly determined by flow. These include: land use impacts, fishing pressure and inputs of riparian vegetation (e.g. fruit and terrestrial arthropods can be important sources of food). Additionally, artificial barriers (e.g. dams and weirs), which affect longitudinal connectivity, would have a significant impact on both species, although currently, there are no such barriers along the main stem of the Daly River.

Model structure

Hydrological sub-model. Similar hydrological sub-models were used for the barramundi and sooty grunter BNs. The dry season flow regime was identified as the primary hydrological element of concern in the Daly River, and this was further identified as comprising of two main elements: (a) the magnitude of the dry season flow, and (b) the timing of any abstractions occurring during the dry season (Figure 6a and b). The dry season flow magnitude affects the fish endpoints by changing habitat suitability and availability (Figures 4 and 5) and thus affecting longitudinal connectivity and fish movement within and between reaches. Changes in dry season flow magnitude were also hypothesized to indirectly affect fish via changes to habitat availability for food production (e.g. aquatic insects, crustaceans and forage fish), and on water quality, which may impact on fish health as it becomes degraded with increased residence time. The timing of any abstraction is assumed to affect the system via changes in water quality for both fish endpoints, and additionally via connectivity and pool size for barramundi. As noted above, the temporal scale of the BN models was restricted to one dry season, although the Barramundi eFlow BN model (Figure 6a) has additional nodes related to the 'previous wet season magnitude' and the 'wet season magnitude 3yrs prior', discussed further below.

The percentile thresholds for the 'natural' scenario daily flow record were used to define states for the 'dry season flow magnitude' node of the BN models, with the upper 10th percentile of the natural record defining the 'high natural' node state (equivalent to more than $1.8\,m^3\,s^{-1}$ at the Katherine River site), the middle 80th percentile as the 'natural' node state (from $0.9–1.8\,m^3\,s^{-1}$) and the lower 10th percentile as the 'low natural' node state (from $0.8–0.9\,m^3\,s^{-1}$) and of course, the minimum modelled 'natural' level for defining the 'below ever recorded' state (less than $0.8\,m^3\,s^{-1}$).

Ecological sub-models. The two BN models are slightly different in the sub-populations influencing the final abundance. For barramundi, the total abundance is made up of a sub-adult population and an adult population (Figure 6a), while for sooty grunter the total abundance is made up of an adult population and a juvenile population (Figure 6b). The size of each sub-population is a function of their respective initial abundance at the start of the dry season and their survival over the dry season.

Barramundi model. Barramundi life history can be represented as a number of stages: (a) emigration of spawning adults from the freshwater reaches of the Daly, (b) spawning and larval development in estuarine areas, (c) juvenile immigration into freshwater reaches and (d) development into sub-adults, which along with the existing adult population (modified by 'persistence') are affected by processes occurring during the dry season and contribute to the final total abundance. Figure 6a shows that much of the BN model structure is focused on the longitudinal connectivity within the Daly River system, and on processes occurring in deeper pool habitats, the latter because expert knowledge and the habitat-use data collected during the study indicated that barramundi rarely utilized shallow riffle-run areas.

Initial sub-adult population abundance (i.e. at the start of the dry season) is determined by reproduction and recruitment success, related to wet season flow in the

DOI: 10.1002/rra

preceding year and the resident adult spawning population, which is determined by the wet season 3 years previously (see Bayliss *et al.*, 2008), and connectivity allowing movement of juveniles into the Daly River system. During the dry season, sub-adult and adult populations are influenced by the availability of suitable habitat for refuge and foraging, food availability and longitudinal connectivity, all of which vary with flow magnitude. During extended periods of low flow when longitudinal movement by barramundi is restricted, local populations densities are expected to increase, food and refuge availability becomes limiting and water quality may deteriorate. We hypothesized that in these circumstances, predation, competition and disease transmission may be particularly important determinants of barramundi survival and population sizes during the dry season.

Sooty Grunter model. As with barramundi, sooty grunter populations are influenced by the availability of suitable habitat for refuge and foraging, food availability and longitudinal connectivity, all of which vary with dry season flow magnitude. The sooty grunter model differed from barramundi in that juvenile and adult sooty grunter occupy different habitats and have distinct ecological requirements and environmental tolerances (Figure 6b). For example, juvenile sooty grunter usually occupy shallow fast flowing riffle areas and feed primarily on aquatic insects and benthic insects, whereas adults occur in deeper slow flowing pools and feed more commonly on larger prey items such as macro-crustaceans. The sooty grunter habitat rating curves for the Katherine River (R1, Figure 5b) clearly show that the availability of suitable juvenile habitat (i.e. shallow fast flowing riffles) is considerably more sensitive to variation in dry season discharge magnitude than is adult habitat (deeper slow flowing pools). Habitat duration curves (Figure 5c) for each water extraction scenario reveal that under the 'natural' and 'historic' scenarios, about 25% of the reach has optimal habitat for 50% of the time whereas under 'current' and 'future' scenarios, the proportional area with optimal habitat exceeded 50% of the time is reduced to only 13 and 7%, respectively. In contrast to juveniles, the effect of increased water extraction on adult habitat availability was much less pronounced (Figure 5d).

Node states. The states for each node in the networks are listed in Appendix 1 and 2, which also contains a brief explanation of the linkages between nodes, plus the sources of information used for knowledge about the nodes, relationships and node state definitions. As a general rule, as few states as possible were used for each node. Where there was significant uncertainty about a node, or about whether quantitative data would ever be available for a node, only two states were used, in keeping with the principle of keeping models (Jakeman *et al.*, 2006). These nodes include 'fish health', 'predation', 'competition' and the food-production nodes. Three states (high, intermediate and low) were defined for the 'physical habitat' nodes and thresholds were based on the upper 10th percentile, lower 10th percentile and minimum habitat area, respectively, derived from the optimal habitat area duration curves generated under the natural flow scenario.

Nodes measuring fish abundance (e.g. 'net juvenile abundance', 'net adult abundance', 'net subadult abundance', 'starting adult abundance') were discretized as percentiles based on a frequency distribution of fish abundance observations derived from the field sampling data (see description of field data collection below) to define the current baseline natural state of the fish population.

Parameterization. All Conditional Probability Tables (CPTs) were defined by expert elicitation, except for the influence of 'extraction scenario' on 'dry season flow magnitude' and 'physical habitat' nodes. In these cases, probabilities were calculated from hydrological modelling of a multi-year daily time series under different scenarios and the optimal habitat area duration curves generated from the 2D hydrodynamic models. The CPTs for both the barramundi and the sooty grunter were defined by the fish experts and are listed in Appendix 1 and 2, where the reasoning behind the probabilities is described and supporting literature is provided. The field data collected on fish abundances at the five focal reaches over the 2006–2008 study period were reserved for validation of the models. As more data are collected, some of it may be used to update and refine the CPTs based on the expert priors, reserving only a portion for validation.

ANALYSIS OF THE BN MODELS

In this section, the behaviour of the parameterized BN models is reported, with an analysis of sensitivity and validation against the collected field data. These tests provide useful information on how the system works, what are the important factors in the system and how well the model predicts fish abundances.

Sensitivity analysis

Sensitivity analysis was performed on the networks to identify which nodes have the greatest influence on the prediction of the endpoints 'barramundi abundance' and 'sooty grunter abundance' nodes (shown as Probability (Barramundi Abundance) = Natural). The sensitivity analysis used here calculates reductions in Shannon's entropy (also known as the 'mutual information') and is described in more detail by Pearl (2000). Ranking the nodes according to entropy reduction identifies those nodes with the most influence on the endpoint in relation to added evidence. This allows further research to focus on the

DOI: 10.1002/rra

priority nodes to reduce uncertainty and maximize confidence in decision making. The results are shown in Figure 7a and b, with the most influential nodes closest to the *X*-axis, with the length of the bars indicating the variation for the endpoint state being in a 'natural' condition (the longer the bar, the greater the influence on the endpoint being in this state).

The sensitivity of the barramundi abundance endpoint to the other network variables is shown in Figure 7a. The order of sensitivity indicates that the adult sub-population is more significant than the sub-adults, as would be expected for a long-lived species. In terms of the drivers, Barramundi appear to be quite sensitive to both the timing of the water abstraction and its magnitude, and this influence occurs mainly via the impact on 'predation' and 'competition' in reduced pool size (even more than natural) at the end of the dry season, resulting in increased 'fish density'. These factors also have an impact on 'fish health' and 'water quality'.

If the width of a bar is less than the bar above, this indicates the variable is less sensitive for this particular state of the endpoint relative to the overall endpoint sensitivity. For example, the 'starting adult abundance' is not very influential on the endpoint being in a 'natural' state, which is expected given that much of the sensitivity for this variable would be in getting an 'extreme high' or 'extreme low' final abundance.

Somewhat surprisingly, the 'previous wet season magnitude' node has little influence on the endpoint. This is in contrast to the findings of Bayliss *et al*. (2008) who found a plausible statistical relationship when focused on wet season populations. Our fish experts believed it reasonable to carry this relationship through to the dry season populations, and

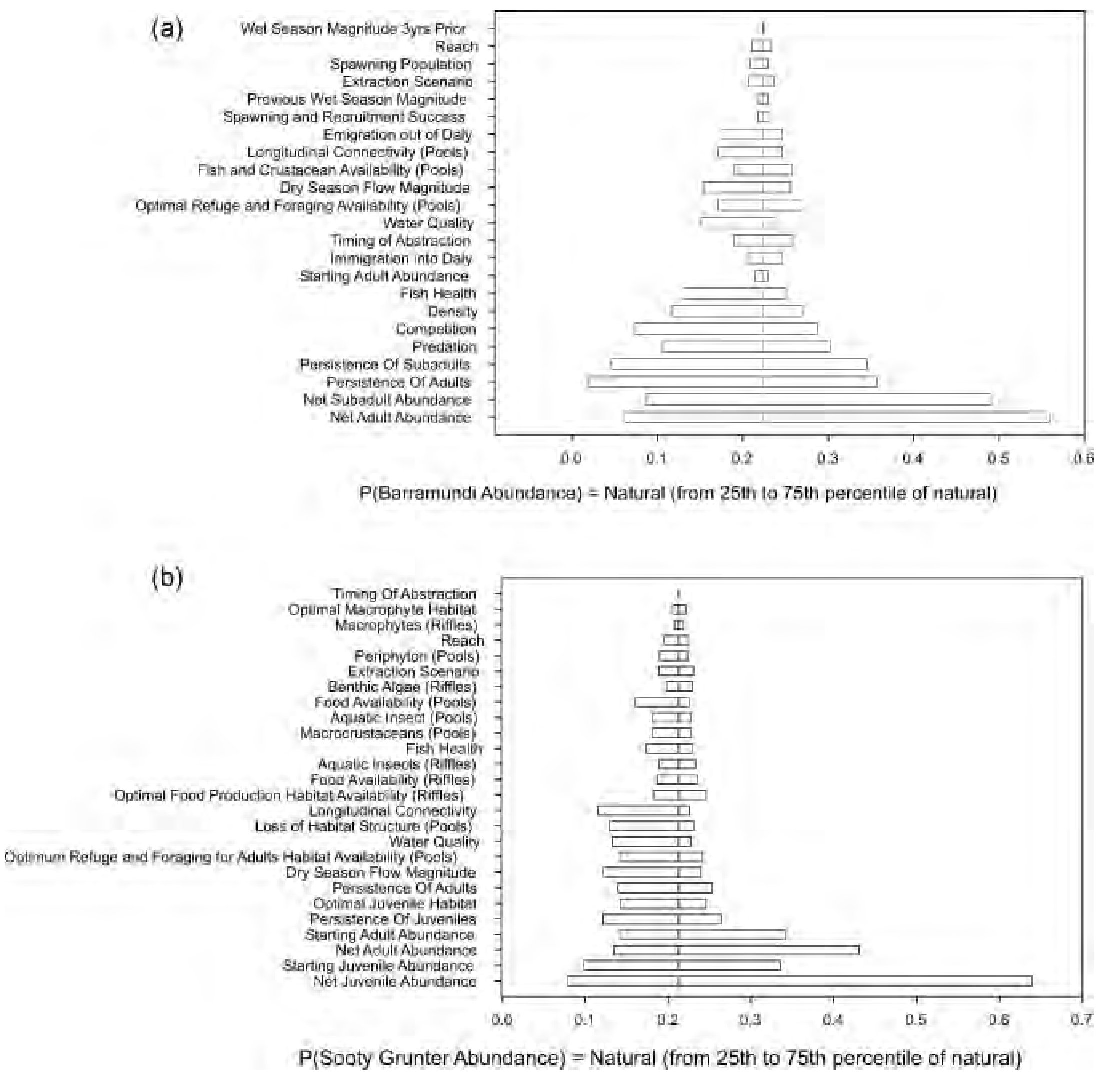

Figure 7. Sensitivity analysis showing the potential influence of network variables on (a) 'barramundi abundance' and (b) 'sooty grunter abundance' being in a 'natural' state.

DOI: 10.1002/rra

this was represented in the BN model. Bayliss *et al*. (2008) also found a strong relationship between wet season abundance and 'wet season magnitude 3 years prior'; however, this is not reflected in the sensitivity analysis shown in Figure 7a. Examination of the BN model by manipulating the state of the 'wet season magnitude 3 years prior' node revealed that although its influence is partly transferred to its child node 'potential spawning population', this effect is then barely passed on to 'spawning and recruitment success', and thus also has minimal impact on the rest of the network. The other parent of the 'spawning and recruitment success' node is 'previous wet season magnitude', which has a much greater influence, in agreement with the Bayliss *et al*. (2008) study. This result can be seen in more detail in a specific sensitivity analysis of the 'spawning and recruitment success' node in Table I, with the influence of 'potential spawning population' and 'wet season magnitude 3 years prior' orders of magnitude less than the influence of the immediate 'previous wet season magnitude'.

The potential effect of changes in each variable on the probability of the 'sooty grunter abundance' being in a 'natural' state is shown in Figure 7b, together with the range (minimum to maximum belief) associated with changes in each node. In contrast to the Barramundi BN model, the sensitivity analysis of the Sooty Grunter BN model indicates that 'juvenile population abundances' in the Daly River have a greater influence on the 'sooty grunter abundance' than 'adult abundances', which matches previous findings regarding the sooty grunter population age structure in tropical rivers (Pusey *et al*., 2004), as well as the field data discussed previously.

In terms of drivers, the 'dry season flow magnitude' is much more important to the 'sooty grunter abundance' than the 'timing of abstraction' (Figure 7b). This influence occurs via changes in the 'optimal refuge and foraging for juveniles habitat availability (riffles)' node. After the 'dry season flow magnitude' node, the next most sensitive nodes are 'optimal refuge and foraging for adults habitat availability (pools)', 'water quality', 'loss of habitat structure (pools)' and 'longitudinal connectivity', all of which are modelled as impacting the adult subpopulation only. After this, variables related to food availability and production become significant.

Table I. Sensitivity analysis for barramundi 'spawning and recruitment success'

Node	Mutual information
Previous wet season	0.93
Migration into Daly	0.69
Potential spawning population	0.011
Wet season 3yrs prior	0.0055
Migration out of Daly	0.00025

Validation

The available fish abundance data were used to test the model predictions. These data consisted of fish abundance at the five sites over a 3-year period. The predicted versus actual results for this test are shown in Tables II and III. Although the amount of field data was limited, it was collected across a wide range of flow conditions. Actual fish abundances also included data points for almost every state (the exception being Barramundi as 'low natural'). The BN models perform surprisingly well, with the error rate for predicted vs. actual 'barramundi abundance' being only 20%, and 27% for the 'sooty grunter abundance'. It is also notable that the erroneous predictions (shaded grey) tended to be for predicted states close (adjacent) to the actual abundances. However, a significant part of this predictive ability is due to the availability of starting abundance data for most data points.

As widely noted in the ecological modelling field, verification and validation of numerical models is inherently partial and can never be complete (Oreskes *et al*., 1994; Bellocchi *et al*., 2010). Model validation can only be considered within the context of the aim of the model (Rykiel, 1996). In this case, a primary reason for using a BN was the sparseness of available data, which necessarily also limits the availability of validation data. Furthermore, validation is necessarily a fluid concept in ecological modelling, involving a range of procedures from traditional quantitative validation originating in physical modelling where larger datasets are available due to historical interest and effort, development of instrumentation, and ease of monitoring, amongst other factors, through to validation of model results and plausibility by independent experts (Jakeman *et al*., 2006), which can be required particularly where there is little data, for example where data are impractical to collect, including remote sites such as the Daly River. The model and model outputs discussed here were presented to Daly River stakeholders, including independent scientists and water resource managers (see Acknowledgements section), at a workshop on 29 June 2009 for additional validation.

Finally, it should be noted that the fish abundance data are short term and recent (2006–2008), and primarily relevant to the 'current' extraction scenario, although the actual 'dry season flow magnitude' data were used in the test, not 'extraction scenario' data. As further water is extracted in the region, any data collected on changes in fish abundance could be used to either update the model by further training the model relationships, or to further validate the model results. Additional data for intermediate nodes are also

Table II. Barramundi BN: predicted vs. actual abundance

Predicted				Actual
Extreme low (<10th percentile)	Low natural (10th–25th percentile)	Natural (25th–75th percentile)	High natural (>75th percentile)	
2	0	0	0	Extreme low
1	1	0	0	Low natural
1	1	5	1	Natural
0	0	0	4	High natural

Correct predictions are aligned along the diagonal from top left to bottom right, e.g. the predicted 2 'extreme low' states in the left column, match the 2 'extreme low' measured actual abundances of the uppermost row. Note incorrect predictions adjacent to the diagonal, e.g. the predicted 'extreme low' in the left column that does not match with the 'low natural' measured actual abundance in the second row down.
Error rate = 20%.

highly desirable to test that model predictions are the result of a correct understanding of the system. Of particular interest would be data on the nodes indicated to be important by the sensitivity analysis, e.g. 'predation', 'competition' and 'density' for barramundi, and 'optimal refuge and foraging for juveniles habitat availability (riffles)', 'optimal refuge and foraging for adults habitat availability (pools)' and 'water quality' for sooty grunter.

APPLICATION OF THE BN MODELS

The eFlow BN models were used to predict barramundi and sooty grunter population abundance for three of the four different water extraction scenarios (given that the 'natural' and 'historic' flow scenarios are very similar we only investigated the 'natural' scenario). For simplicity, only the results for the lower Katherine River site (R1, Figure 1) are discussed here, with the focus being on the probability that the fish abundance endpoint is in an 'extreme low' state, as this is the current concern associated with the extraction regimes. However, additional divisions for the endpoint ('high natural', 'low natural') were included for future use as more data becomes available, and as research and management begins to focus on other scenarios and the natural variability in the system.

'Natural' flow scenario

The 'starting adult abundance' node distribution for the 'natural' flow scenario was set to the distributions defined as 'natural'. For the other flow scenarios, the 'starting adult abundance' nodes were left unknown (uniform probabilities) to represent a stable state under the scenario of interest, as opposed to an intermediate case, showing the progression from a natural regime to the selected flow regime scenario. The results showing the probability of the endpoint abundances being 'extreme low' are compared with the 'natural' case and with each other in Figures 8 and 9. For the barramundi, the 'previous wet season magnitude' and 'wet season magnitude 3 years prior' nodes were similarly set to the 'natural' distribution for the 'natural' flow scenario and uniform distributions were used for the other flow scenarios.

For the 'natural' flow scenario, the Barramundi eFlow BN model predicted a 36% probability that 'barramundi abundance' would be 'extreme low' (Figures 8 and 10), while the Sooty Grunter BN model predicted a 25%

Table III. Sooty Grunter BN: predicted vs. actual abundance

Predicted				Actual
Extreme low (<10th percentile)	Low natural (10th–25th percentile)	Natural (25th–75th percentile)	High natural (>75th percentile)	
1	1	0	0	Extreme low
1	2	1	0	Low natural
0	1	6	0	Natural
0	0	2	2	High natural

Correct predictions are aligned along the diagonal from top left to bottom right, e.g. the predicted 'extreme low' state in the left column, matches 'extreme low' measured actual abundance of the uppermost row. Note incorrect predictions adjacent to the diagonal, e.g. the predicted 'low' in the second column from left that does not match with the 'extreme low' measured actual abundance in the uppermost row.
Error rate = 27%.

DOI: 10.1002/rra

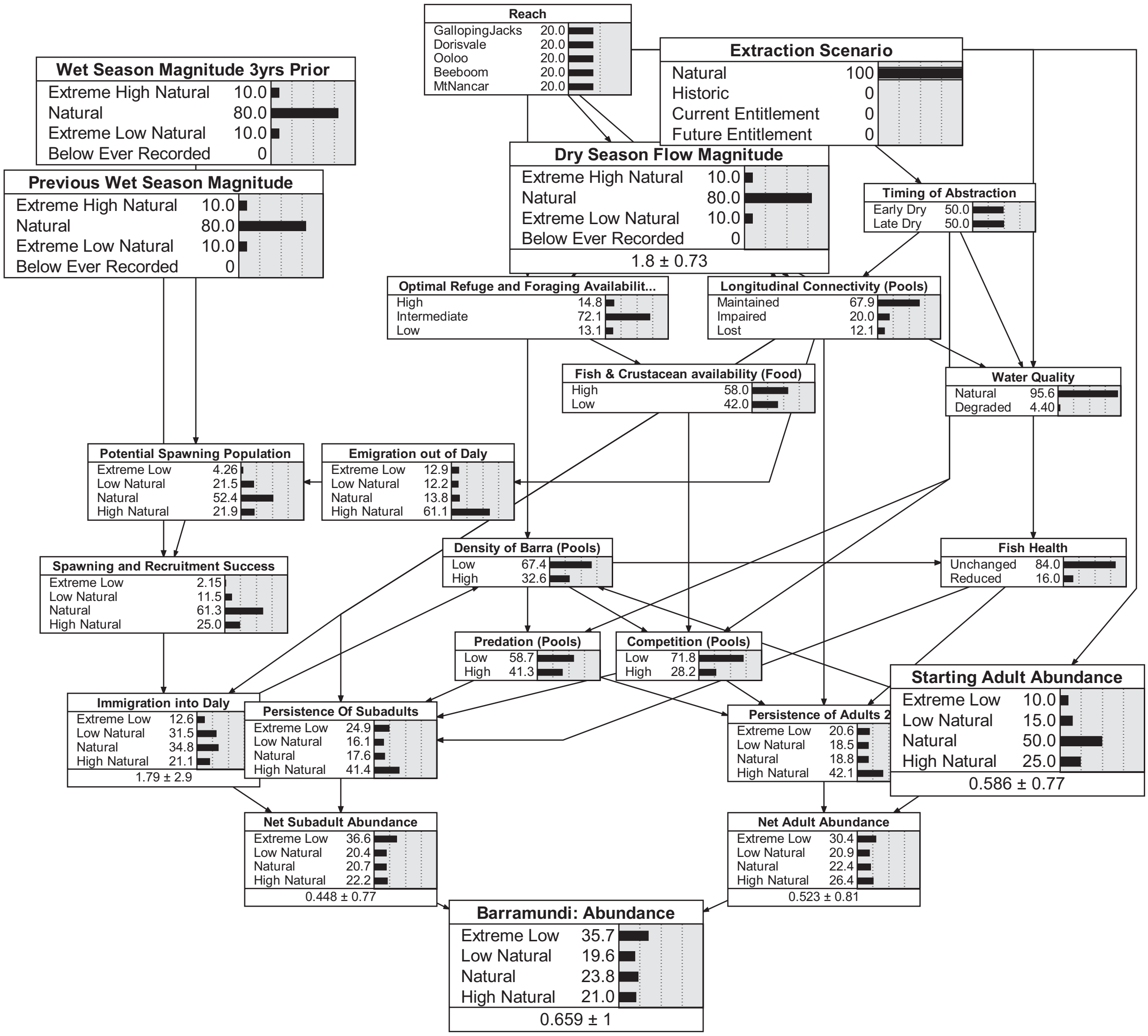

Figure 8. Barramundi eFlow BN model for 'natural' flow conditions. The 'extraction scenario', 'previous wet season magnitude' and 'starting abundance' were set to a 'natural' state or distribution.

probability that 'sooty grunter abundance' would be 'extreme low' (Figures 9 and 10).

'Historic' flow scenario

Because the 'historic' flow scenario is very similar to the 'natural' scenario (Figure 3), the probabilities of 'barramundi abundance' and 'sooty grunter abundance' being in an 'extreme low' state are also very similar (Figure 10).

'Current entitlement' flow scenario

The 'current entitlement' (fully utilized) flow scenario would result in a 45% reduction in mean dry season discharge (Figure 3). Under this flow scenario, there was an increased probability (43%) that 'barramundi abundance' would be 'extreme low' state (Figure 10), and also an increased probability (43%) that 'sooty grunter abundance' would be in the 'extreme low' state (Figure 10).

'Possible future entitlement' flow scenario

Under the 'possible future entitlement' flow scenario (Figure 3), 'dry season flow magnitude' may be reduced by as much as 60% of the natural dry season flow, and as expected the BN models predicted an even higher probability of both 'barramundi abundance' and 'sooty grunter abundance' being in the 'extreme low' state (46 and 46%, respectively—Figure 10). Interestingly, the 'timing of abstraction' under the 'possible future entitlement' flow scenario was predicted to have an impact on 'barramundi

DOI: 10.1002/rra

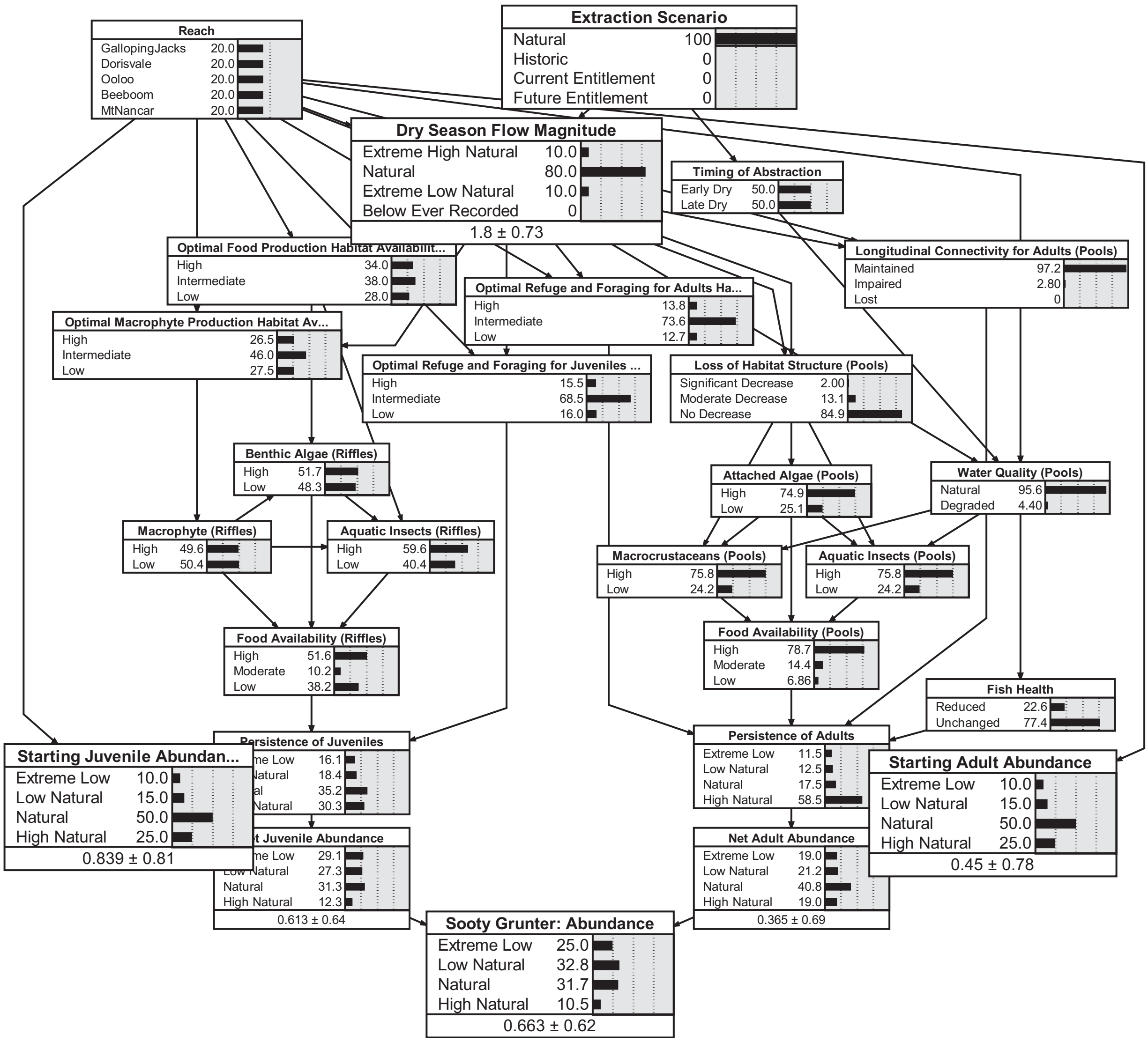

Figure 9. Sooty Grunter eFlow BN model for 'natural' flow conditions. The 'extraction scenario', 'previous wet season magnitude' and 'starting abundance' were set to a 'natural' state or distribution.

abundance'. For example, with extraction occurring in the 'early dry', the probability of 'barramundi abundance' being in the 'extreme low' state was 37%, while if extraction occurred in the 'late dry' this probability increased to 54% (Figure 10). There was no change in the probability of 'sooty grunter abundance' due to the timing of water extraction (Figure 10).

DISCUSSION

eFlow BN models

The BN models reported here appear to be behaving plausibly, however they do require further evaluation and validation. In particular, the sensitivity analysis indicated that more understanding and quantitative data on predation and competition with respect to barramundi in dry season pools would reduce uncertainty. For the sooty grunter, additional information is needed on the impact of water quality on the adult fish. Further seasonal abundance data for both fish species would confirm the validity of the model predictions. Finally, when enough data are collected for any of the nodes, part of this dataset may be used to update the CPTs based on the expert priors.

Under 'natural' conditions, the BN models predict a significant likelihood that the fish abundances would be low (Figures 8–10). This is to be expected given the high inter-annual variation in river flows (Figure 2) and other environmental factors that influence fish populations in the Daly River system.

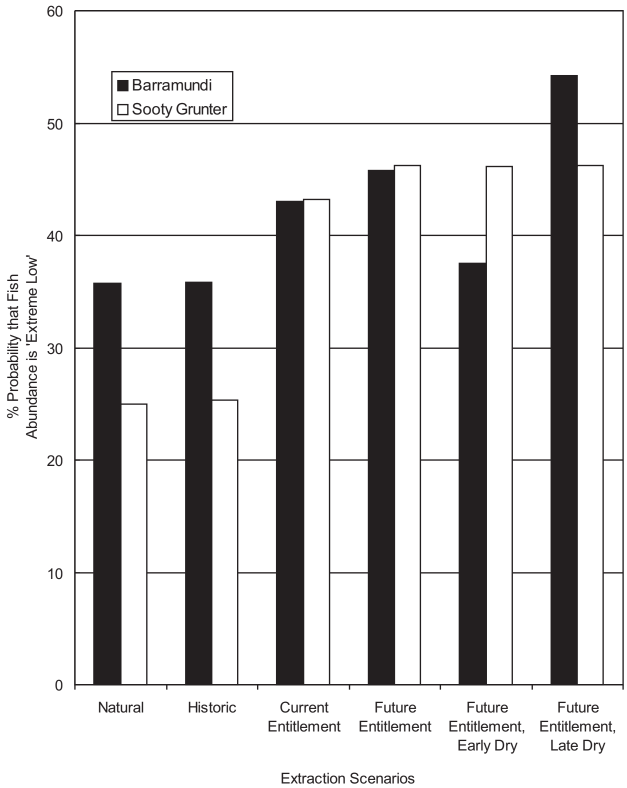

Figure 10. Probability that barramundi and sooty grunter abundances are in an 'extreme low' state under different extraction scenarios.

The greatest demand for water for irrigation purposes will occur during the dry season, and it is of interest to the managers whether there is any difference ecologically if this water is taken early in the dry season or later in the dry. We have modelled this through the 'timing of abstraction' node. For example, the Barramundi eFlow BN model (Figure 6a) allows for the 'timing of abstraction' to influence the endpoint through four ecosystem attributes ('water quality', 'longitudinal connectivity (pools)', 'competition (pools)' and 'predation (pools)'), and in the Sooty Grunter eFlow BN model through the 'water quality' node (Figure 6b). These relationships were defined using expert elicitation rather than quantitative data.

Additionally, the experts defined the BN relationships using the assumption that there would be an immediate impact of (groundwater) abstraction on river flow. Modelled groundwater pumping scenarios indicate the lag in impact on river discharge is on the order of a month or more, depending on distance of the bore from the river. This lag time would have significant implications for managing extraction regimes in the mid-to-late dry season. Overall, more information is needed on the time period elapsing between groundwater extraction and actual impact on river flow.

Management scenarios

Although the risk of negative impacts on barramundi populations due to late dry season water extraction is significant (potentially increasing the likelihood of an 'extreme low' abundance from 36 to 54%), this could be mitigated by ensuring the 'late dry' flow is as natural as possible (which could potentially keep the likelihood of 'extreme low' abundance at 37%). This is a more practical management action than attempting to intervene in intermediate variables such as predation, competition and disease in barramundi. There may also be some scope for improvements in 'water quality' and 'longitudinal connectivity' to mitigate the impact of reduced 'dry season flow magnitude', although our model shows that this would have much less effect on abundance than changes to timing.

For the sooty grunter, mitigation of reduction in 'dry season flow magnitude' is more problematic (potentially increasing the likelihood of an 'extreme low' abundance from 25 to 46%), as timing is not a significant issue. 'Optimal refuge and foraging for juveniles habitat availability' in the riffles may be amenable to manipulation, according to the habitat preferences shown in Figure 4.

Erskine *et al*. (2003) synthesized information from projects that had investigated the flow requirements of five different flora and fauna endpoints in the Daly River, and made recommendations for environmental flows based on this information, although it should be noted fish were not one of the endpoints considered. A direct comparison with our study is not straightforward, because most sites considered by Erskine *et al*. (2003) were in different locations to ours. However, it is interesting that Erskine *et al*. (2003) recommended a cumulative maximum extraction of 20% of the stream flow, decreasing in proportion during lower flows to <8%, with a cut-off for any extraction at a low flow threshold. If current planned entitlements were fully utilized, the river at the lower Katherine site would experience a decrease of 45% in the mean *dry season* discharge. This agrees with our assessment that current full entitlements are too high and would have negative environmental impacts at least on the two fish species investigated here.

The current fish BN models have been used to predict impacts on the two fish species due to changes in dry season flow magnitude, and these data will provide water resource managers with some guidance in how to minimize the ecological effects from water extraction in the future. However, additional ecological and hydrological information on the Daly River system is required to improve the credibility of the fish BN models. In particular, further information on fish abundances, as well as quantitative data on intermediate nodes, will help in validating the models.

In summary, if the full current entitlements for extraction were allowed, we predict there would be a significant

DOI: 10.1002/rra

increase (by 10–20%) in the likelihood of 'extremely low' fish abundances for both barramundi and sooty grunter. The risk of major impact on fish abundances would increase if the potentially further increased future entitlements were to occur. Mitigating the impact of decreased environmental flows may be feasible for barramundi by limiting late dry extraction and focusing extractions on the early dry season. There may be scope to minimize low flow impacts for sooty grunter by manipulating riffle habitat. However, it may be simpler to minimize overall extraction during the dry season, which would be beneficial for both species. Finally, there are some data gaps in the timing of impact from groundwater pumping for extraction, which could also have implications for management.

BNs for environmental flow assessment

This study has shown that Bayesian networks are useful for evaluating the ecological risks of different water extraction scenarios for two important species in the Daly River and that a combination of quantitative information and expert knowledge can be used to achieve this. In this sense, the use of BNs for environmental flow assessment represents a significant step forward from simple approaches based on habitat modelling or expert opinion alone (see Arthington and Pusey, 2003; Tharme, 2003), as recommended by Hart and Pollino (2009). Our applications of BNs utilized quantitative information on the relationships between river discharge, fish habitat suitability and availability, while simultaneously considering the influence of many other ecologically important processes such as migration, feeding, growth, reproduction and survival. BNs are also transparent in that sensitivity analyses clearly identify which sources of information (quantitative vs. expert knowledge) exert most influence on the model outcomes. Knowledge gaps and priorities for further research can therefore be clearly and logically identified.

Quantitative information on the environmental flow requirements of aquatic species is lacking for most tropical northern Australian rivers. This is an important knowledge gap given the increasing focus on the value of the region's water resources for human needs and the potential of global climate change to influence hydrology. Thus, environmental flow management will increasingly require judgments to be made in the absence of extensive quantitative knowledge whilst efforts to provide this quantitative knowledge are undertaken but lag behind in time in their ability to contribute. In this sense, one of the strengths of the BN process is the ability to modify and improve the predictive process by the addition of new information and the replacement of expert opinion by quantitative knowledge when and if such information becomes available. We conclude by arguing that BNs provide an ideal way of combining quantitative data with expert knowledge (where such data are lacking) to inform environmental flow management and planning particularly in areas such as northern Australia where quantitative knowledge is limited.

ACKNOWLEDGEMENTS

Financial support for this project was provided by Land and Water Australia, Melbourne Water, Department of Sustainability and Environment, Queensland Department of Natural Resources and Water, the Goulburn-Broken, East Gippsland and West Gippsland CMAs, and the TRaCK program. TRaCK receives major funding for its research through the Australian Government's Commonwealth Environment Research Facilities initiative; the Australian Government's Raising National Water Standards Program; Land and Water Australia; the Fisheries Research and Development Corporation and the Queensland Government's Smart State Innovation Fund. We thank Peter Bayliss, Neil Collier, Marcus Finn, Sue Jackson, Adam Liedloff, Carmel Pollino, Simon Townsend, Rick van Dam, Danielle Warfe and Chris Wicks for their assistance in developing the conceptual models and participating in the workshops. We thank members of the Wagiman, Wardaman and Jawoyn traditional owner groups for their assistance in the field and generous sharing of knowledge about the Daly River. We thank Des Yinfoo (NRETA) for providing the modelled discharge data. Finally, we are most grateful to Yung En Chee for her excellent review of an earlier draft.

REFERENCES

Arthington AH, Pusey BJ. 2003. Flow restoration and protection in australian rivers. *River Research and Applications* **19**: 377–395.

Bartolo R, Bayliss P, van Dam R. 2008. Semi-quantitative risk assessments. In *Ecological risk assessments for Australia's Northern Tropical Rivers*, Bartolo R, Bayliss P, van Dam R (eds). The Environmental Research Institute of the Supervising: Scientist: Darwin, Australia; 162–270.

Barton DNT, Saloranta T, Moe SJ, Eggestad HO, Kuikka S. 2008. Bayesian belief networks as a meta-modelling tool in integrated river basin management – Pros and cons in evaluating nutrient abatement decisions under uncertainty in a Norwegian river basin. *Ecological Economics* **66**: 91–104.

Batchelor C, Cain J. 1999. Application of belief networks to water management studies. *Agricultural Water Management* **40**: 51–57.

Bayliss P, Bartolo R, van Dam R. 2008. Quantitative ecological risk assessments. In *Ecological Risk Assessment for Australia's Northern Tropical Rivers*. Bartolo R, Bayliss P, van Dam R (eds). Environmental Research Institute of the Supervising: Scientist: Darwin, Australia; 271–415.

Begg GW, van Dam R, Lowry JB, Finlayson CM, Walden DJ. 2001. Inventory and risk assessment of water dependent ecosystems in the Daly basin, Northern Territory, Australia. Supervising Scientist Report 162, Environment Australia: Darwin.

DOI: 10.1002/rra

Bellocchi G, Rivington M, Donatelli M, Matthews K. 2010. Validation of biophysical models: issues and methodologies. A review. *Agronomy for Sustainable Development* **30**: 109–130.

Blanch S. 2008. Steps to a sustainable Northern Australia. *Ecological Management and Restoration* **9**: 110–115.

Blanch S, Rea N, Scott G. 2005. *Aquatic Conservation Values of the Daly River Catchment, Northern Territory, Australia*. WWF-Australia, Charles Darwin University and the Environment Centre NT: Darwin.

Carpenter SR. 2002. Ecological futures: Building an ecology of the long now. *Ecology* **83**: 2069–2083.

Castelletti A, Soncini-Sessa R. 2007. Bayesian Networks and participatory modelling in water resource management. *Environmental Modelling & Software* **22**: 1075–1088.

Chan T, Ross H, Hoverman S, Powell B. 2010. Participatory development of a Bayesian Network model for catchment-based water resource management. *Water Resources Research* **46**: W07544 DOI: 10.1029/2009WR008848

Douglas MM, Bunn SE, Davies PM. 2005. River and wetland food webs in Australia's wet-dry tropics: general principles and implications for management. *Marine and Freshwater Research* **56**: 329–342.

DRMAC (Daly River Management Advisory Committee). 2009. Chapter 27, Daly River catchment: regional perspective. In Northern Australia Land and Water Science Review, Stone, P, (ed.). Final report to the Northern Australia Land and Water Taskforce. CSIRO Publishing. Available at http://www.nalwt.gov.au/science_review.aspx

Ellison AM. 1996. An introduction to Bayesian inference for ecological research and environmental decision-making. *Ecological Applications* **6**: 1036–1046.

Erskine WD, Begg GW, Jolly P, Georges A, O'Grady A, Eamus D, Rea N, Dostine P, Townsend S, Padovan A. 2003. Recommended environmental water requirements for the Daly River, Northern Territory, based on ecological, hydrological and biological principles. *Report 175*, Supervising Scientist: Darwin.

Faulks JJ. 1998. Daly River catchment. Part 1. An assessment of the physical and ecological condition of the Daly River and its major tributaries. Technical Report TR99/10. Northern Territory Department of Lands, Planning and: Environment: Darwin.

Ganf GG, Rea N. 2007. Potential for algal blooms in tropical rivers of the Northern Territory, Australia. *Marine and Freshwater Research* **58**: 315–326.

Gehrke P, Bristow K, Bunn S, Douglas M, Edgar B, Finlayson M, Hamilton S, Loneragan N, Lund M, Pearson R, Prosser I, Robson C. 2004. Sustainable futures for Australia's tropical rivers – a strategy for developing research directions for Australia's tropical river systems. Technical Report No. 17/04, CSIRO Land and Water: Darwin.

Groshens TG, Orth DL. 1994. Transferability of habitat suitability criteria for smallmouth bass. *Micropterus dolomieu. Rivers* **4**: 194–212.

Hart BT. 2004. Environmental risks associated with new irrigation schemes in Northern Australia. *Ecological Management and Restoration* **5**: 107–111.

Hart BT, Pollino CA. 2009. Bayesian for risk-based environmental water allocations. *Waterlines Report Series No 14*. National Water Commission: Canberra. Available at http://www.nwc.gov.au/www/html/1021-bayesian-modelling-report-no-14.asp

Hart B, Burgman M, Webb A, Allison G, Chapman M, Duivenvoorden L, Feehan P, Grace M, Lund M, Pollino C, Carey J, McCrae A. 2005. *Ecological Risk Management Framework for the Irrigation Industry. Report to National Program for Sustainable Irrigation (NPSI)*. Water Studies Centre, Monash University: Clayton, Australia.

Hillman H, Aplin G, Brierly G. 2003. The importance of process in ecosystem management: lessons from the Lachlan Catchment, New South Wales, Australia. *Journal of Environmental Planning and Management* **46**: 219–237.

Hocutt CH. 1981. Fish as indicators of biological integrity. *Fisheries* **6**: 28–31.

Jakeman AJ, Letcher RA, Norton JP. 2006. Ten iterative steps in development and evaluation of environmental models. *Environmental Modelling & Software* **21**: 602–614.

Kennard MJ, Pusey BJ, Olden JD, Mackay SJ, Stein JL, Marsh N. 2010. Classification of natural flow regimes in Australia to support environmental flow management. *Freshwater Biology* **55**: 171–193, DOI: 10.1111/j.1365-2427.2009.02307.x

Kennard MJ, Pusey BJ, Perna C, Allsop Q, Burrows D, Douglas M. 2008. How much sampling is enough? A multi-scaled comparison of fish sampling effort and efficiency in quantifying freshwater fish assemblages. Appendix 2. In Comprehensive Analysis of Freshwater Fish Faunas and Their Key Management Issues Across Northern Australia. Final Report to the Department of Environment and Water: Darwin.

King IP. 2001. *RMA2 – A Two Dimensional Finite Element Model for Flow in Estuaries and Streams—Version 6.6g*. Resource Modelling Associates: Sydney.

Korb KB, Nicholson AE. 2004. *Bayesian Artificial Intelligence*. Chapman and Hall/CRC Press: London.

Lamontagne S, Cook PG, O'Grady A, Eamus D. 2005. Groundwater use by vegetation in a tropical savanna riparian zone (Daly River, Australia). *Journal of Hydrology* **310**: 280–293.

Marcot BG, Steventon JD, Sutherland GD, McCann RK. 2006. Guidelines for developing and updating Bayesian belief networks applied to ecological modelling and conservation. *Canadian Journal of Fisheries and Aquatic Sciences* **36**: 3063–3074.

McCann RK, Marcot BG, Ellis R. 2007. Bayesian belief networks. Applications in ecology and natural resource management. *Canadian Journal of Forest Research* **36**: 3053–3062.

McGarry R, Valentine EM. 2008. A low flow finite element hydrodynamic model of the Katherine River. Proceedings of the 9th National Conference on Hydraulics in Water Engineering, Engineers Australia: Darwin, Australia.

Nayak TK, Kundu S. 2001. Calculating and describing uncertainty in risk assessment: The Bayesian approach. *Human & Ecological Risk Assessment* **7**: 307–328.

Northern Territory Government. 2009. Water allocation plan for the Tindall Limestone Aquifer Katherine – 2009 – 2019. Department of Natural Resources, Environment, the Arts and Sport (NRETAS). Natural Resources Division, Water Resource Branch: Darwin. Available at http://www.nt.gov.au/nreta/water/kwac/pdf/final_wap.pdf

NTDIPE (Department of Infrastructure, Planning and Environment). 2003. *Draft Conservation Plan for the Daly Basin Bioregion*. Northern Territory Government: Darwin.

Nyberg JB, Marcot BG, Sulyma R. 2006. Using Bayesian belief networks in adaptive management. *Canadian Journal of Forest Research* **36**: 3104–3116.

Oreskes N, Shraderfrechette K, Belitz K. 1994. Verification, validation, and confirmation of numerical-models in the earth-sciences. *Science* **263**: 641–646.

Pearl J. 2000. *Causality: Models, Reasoning, and Inference*. Cambridge University Press: New York, USA.

Poff NL, Allan JD, Bain MB, Karr JR, Prestegaard KL, Richter BD, Sparks RE, Stromberg JC. 1997. The natural flow regime: a paradigm for river conservation and restoration. *Bioscience* **47**: 769–784.

Pollino CA, Woodberry O, Nicholson A, Korb K, Hart BT. 2007. Parameterisation and evaluation of a Bayesian network for use in an ecological risk assessment. *Environmental Modelling & Software* **22**: 1140–1152.

Preston BL, Jones RN. 2008. Evaluating sources of uncertainty in Australian runoff projections. *Advances in Water Resources* **31**: 758–775.

DOI: 10.1002/rra

Pusey BJ, Kennard MJ, Arthington AH. 2004. *Freshwater Fishes of North-eastern Australia*. CSIRO Publishing: Collingwood, Victoria.

Russell DJ, Garrett RN. 1985. Early life-history of barramundi, *Lates calcarifer* (Bloch), in Northeastern Queensland. *Australian Journal of Marine and Freshwater Research* **36**: 191–201.

Rykiel EJ. 1996. Testing ecological models: the meaning of validation. *Ecological Modelling* **90**: 229–244.

Scheidegger KJ, Bain MB. 1995. Larval fish distribution and microhabitat use in free-flowing and regulated rivers. *Copeia* **1**: 125–135.

Schlosser IJ. 1985. Flow regime, juvenile abundance, and the assemblage structure of stream fishes. *Ecology* **66**: 1484–1490.

Shenton W, Hart BT, Chan T. 2010. Bayesian network models for environmental flow decision-making: 1. Latrobe River Australia. *River Research and Applications* **26**, DOI: 10.1002/rra.1348

Stewart-Koster B, Bunn SE, MacKay SJ, Poff NL, Naiman RJ, Lake PS. 2010. The use of Bayesian networks to guide investments in flow and catchment restoration for impaired river ecosystems. *Freshwater Biology* **55**: 243–260.

Tharme RE. 2003. A global perspective on environmental flow assessment: emerging trends in the development and application of environmental flow methodologies for rivers. *River Research and Applications* **19**: 397–441.

Ticehurst JL, Newham LTH, Rissik D, Letcher RA, Jakeman AJ. 2007. A Bayesian network approach for assessing the sustainability of coastal lakes in New South Wales, Australia. *Environmental Modelling & Software* **22**: 1129–1139.

Townsend SA, Padovan AV. 2009. A model to predict the response of the benthic macroalga *Spirogyra* to reduced base flow in tropical Australia. *River Research and Application* **25**: 1193–1203.

Trefry MG, Muffels C. 2007. FEFLOW: a finite-element ground water flow and transport modeling tool. *Ground Water* **45**: 525–528. DOI: 10.1111/j.1745-6584.2007.00358.x

van Dam R, Bartolo R, Bayliss P. 2008. Identification of ecological assets, pressures and threats. In *Ecological Risk Assessment for Australia's Northern Tropical Rivers*, Bartolo R, Bayliss P, van Dam R (eds). Environmental Research Institute of the Supervising Scientist: Darwin, Australia; 14–161.

Varis O, Kuikka S. 1999. Learning Bayesian decision analysis by doing: lessons from environmental and natural resources management. *Ecological Modelling* **119**: 177–195.

Wang QJ, Robertson DE, Haines CL. 2009. A Bayesian network approach to knowledge integration and representation of farm irrigation: 1. Model development. *Water Resources Research* **45**: 18.

Whitfield AK, Elliott M. 2002. Fishes as indicators of environmental and ecological changes within estuaries: a review of progress and some suggestions for the future. *Journal of Fish Biology* **61**: 229–250.