Chapter 4

Quantitative ecological risk assessments for the Daly River

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Contents

4.1 Introduction and methods

Key guiding principles of ecological risk assessment are introduced in Chapter 1. The risk assessment process is now increasingly applied to catchments and their aquatic ecosystems because it is transparent, consistent and reliable (Diamond & Serveiss 2001, Serveiss 2002, Billington 2005, Hart 2004, Hart et al 2005). Our overall approach is to first develop a conceptual model with stakeholders that capture the multiple threats and their pathways to multiple assets, and to then prioritise or rank them all based on qualitative and/or semi-quantitative risk analysis (Chapters 2 & 3). The most important step in ranking multiple risks occur between the conceptual model and qualitative assessments where lesser or even trivial risks are filtered out in order to focus quantitative effort on more significant risks. Where data allow, quantitative ecological risk assessment (QERA) is used in preference to qualitative risk rankings, as the cause-effect mechanisms behind the ratings are made explicit.

The scope of the TRIAP program, however, does not allow detailed quantitative ecological risk assessments (QERA) to be undertaken for all four focus catchments. Hence, the Daly River catchment in the Northern Territory (NT) was chosen to test the utility of various QERA approaches that could be applied to the other focus catchments to assess threats to natural assets. The key ecosystem assets of the Daly River catchment (see Faulks 1998a & b; DIPE 2003), and potential threats to those assets, were chosen for examination *a priori* from previous stakeholder consultations and community-based preliminary risk assessments (DRCRG 2004). Those choices included also *a priori* choices of associated ecological and measurement endpoints (Table 4.1). Our choice of conceptual model, assets, threats and endpoints were re-affirmed by further consultation with NT stakeholders (mainly NT DPIFM & NRETA, WWF), and cross-referenced to our assets and threats analysis reported here (Chapter 3). The chosen assets and, hence, ecological assessment endpoints are 'at risk' from multiple regional stressors and so are ideal candidates also to assess the utility of the Relative Risk Model approach (Chapter 3) used to prioritise and select threats for further detailed QERA.

The following generic approach to QERA was adopted (& see Figure 4.6): (i) construct a working conceptual model that identifies hypothesised cause-effect links and interactions between assets and threats; (ii) where adequate empirical data exist use frequentist statistics to characterise risk at a minimum and, if possible, develop more informative and predictive ecological models; (iii) where there is combined reliance on empirical data and expert opinion/knowledge, and/or where decisions need to be made in the face of uncertainty, use Bayesian Belief Networks (BBN, or BN hereafter) to capture all uncertainties; (iv) where possible and desirable, undertake the QERA spatially with respect to assets and threats in order to provide a better basis for on-ground management; and (v) make all uncertainties explicit and examine their influence on assessment outcomes using Monte Carlo simulation and sensitivity analysis. The overall approach is similar to the risk assessment process suggested by Deere and Davidson (2005) for water management in Australia, which encompasses three critical steps: conceptual analysis of the pathways by which risk arise; which generally leads to a process of semi-quantitative ranking for prioritisation of risks; and, if warranted, undertake quantitative prediction of specific risks (see Bevan 2000). The risk assessment approaches used here are consistent with the most recent national and international guidelines with respect to robustness, transparency, coherency and reliability (eg see US EPA 1998, 2003, Cain 2001, AS/NZS 2004a & b, Burgman 2005).

Table 4.1 Summary of key ecosystem assets and threats, and ecological and measurement endpoints, used for quantitative ecological risk assessment of aquatic ecosystems in the Day River catchment. CPUE = Catch Per Unit Effort; DIN = dissolved inorganic nitrogen. Note: fishing effort is not considered a threat to barramundi catch or stock levels (see text).

Ecosystem asset	Key threat	Ecological assessment endpoint	Ecological measurement endpoint
FLOODPLAIN	Water extraction	Magpie goose nesting success	Magpie geese nest density (numbers/km^2)
	Wetland weeds	Condition of magpie goose nesting habitat	% or area (ha) of nesting habitat displaced by wetland weeds
		Condition of magpie goose dry season habitat	% or area (ha) of refuge habitat displaced by wetland weeds
		Plant biodiversity	Risk probability (=exposure x effect) or % of plant species lost in infected area
IN-STREAM	Water extraction	Barramundi catch success	Recreational & commercial barramundi catch
	Water extraction	Sustainable barramundi stock	Barramundi CPUE population index
	Land clearing for land use	Surface water quality	Modelled total sediment & phosphorus exports (t/y) & modelled DIN concentration (μM)

4.1.1 Aquatic ecosystem dynamics

Flood Pulse Concept

Poff et al (1997) proposed a paradigm for river conservation and restoration based on the premise that the ecological integrity of river ecosystems depends on their natural dynamic character, highlighting the critical role that flow variability plays in driving aquatic ecosystems. Hence, a dominant theme in aquatic ecology is that flow variability underpins riverine ecosystem processes, particularly those with floodplains. The Flood Pulse Concept (Junk et al 1989) is based on large tropical rivers and asserts that regular pulses of river discharge are a key factor in the dynamics of river-floodplain systems. However, because the concept focuses on pulses that overflow the bank (hence 'flood' pulses), it emphasises the significance of variability and predictability of the duration, amplitude, frequency, timing and rate of rise and fall of the 'flood' pulse. Puckeridge et al (1998) stated that ecological processes in large rivers are controlled by their flow variability, and suggested that the Flood Pulse Concept can be expanded to encompass hydrological variability and differences among groups of rivers from different climates.

Bunn and Arthington (2002) also argued that flow regime is the key driver of river and floodplain wetland ecosystems, and highlighted the following four relationships that link hydrology and aquatic biodiversity to illustrate potential impacts of altered flow regimes: (i) flow is a major determinant of physical habitat and, hence, biotic composition of aquatic ecosystems; (ii) aquatic species have evolved life history strategies primarily in direct response to natural flow regimes; (iii) maintenance of natural patterns of longitudinal and lateral connectivity is essential to the viability of populations of many riverine species; and (iv) the successful colonisation of introduced aquatic species is generally facilitated by altered flow regimes. They state also that, despite general recognition of these relationships, aquatic ecologists still struggle to predict and quantify biotic responses to altered flow regimes, particularly when direct effects of water developments (eg extractions) are confounded with indirect effects of land use change (eg via increased sediment & nutrient exports). In contrast to many tropical river systems worldwide, and most eastern Australian rivers, the tropical rivers of northern Australia have largely unmodified flow regimes and are relatively free from intensive land use impacts (Douglas et al 2005, Gehrke 2005, Hamilton & Gehrke 2005). Douglas et al (2005) outlined five general principles that characterise ecosystem processes of tropical rivers in Australia, and foremost of these is that seasonal hydrology is a key driver of ecosystem processes and food-web structure.

However, the effects of variable freshwater flow on aquatic ecosystem dynamics are most likely complex and, hence, uncertain, in that they may reflect physical effects, trophic linkages or both. Kimmerer (2002) developed a simplified conceptual food web model for the San Francisco estuary to illustrate possible causal pathways for mechanisms of freshwater flow effects on the abundance of estuarine organisms, whereby hydrodynamic forcing influences the physical environment, which in turn affects the biotic environment (Figure 4.1).

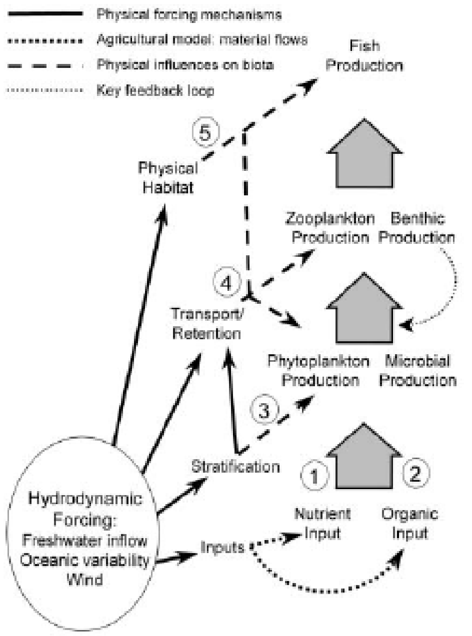


Figure 4.1 Simplified food web for the San Francisco Estuary. Wide arrows indicate material flow and all other arrows indicate causal pathways for mechanisms of flow effects. Hydrodynamic forcing affects the physical environment (solid arrows), and through various mechanisms influences the biotic environment (dotted and dashed arrows). From Kimmerer (2002).

Kimmerer (2002) re-examined long-term monitoring data (20–40 y) and concluded that, whilst several mechanisms for positive or negative flow effects on biological populations in estuaries have been proposed, positive effects appear to operate mainly through stimulation of primary production with effects propagating up the food web. The conceptual and predictive empirical models developed in later sections of this report to examine potential effects of freshwater flow extractions on floodplain and in-stream populations of magpie geese and barramundi, respectively, are contracted models of complex ecological processes that we currently have little understanding of, despite substantial research investment over the decades. This fact simply underscores the difficult nature of knowledge acquisition. Nevertheless, simplified predictive models that use the best available information at hand may be useful for 'what if' scenario simulations if model uncertainty is made explicit, and this approach essentially underpins our QERA methodology.

Climatic drivers – trends in rainfall, catchment hydrology and river flow

Many of the major waterways in the 'Top End' of the NT are being used for recreation, pastoralism, cropping, horticulture and mining (Wygralak 2006). River flow regimes are highly seasonal with more than 95% of the flow volume occurring during the wet season (October to April), with most rivers and streams ceasing flow during the dry season (May to September). The Daly River, however, is one of the Northern Territory's largest perennial rivers with dry season baseflow dominated by groundwater discharge from massive underlying limestone aquifers (Tickell 2002, Tickell et al 2002).

Data has been collected on surface water and groundwater hydrology in the Daly River catchment for over 50 years. Hence Jolly (2002a,b) was able to use long-term (1957–2000) Katherine rainfall and Daly River runoff data to construct a water balance for the Daly River catchment with reference to three coupled components: (i) inflow (rain, inflow from adjacent groundwater sources); (ii) outflow (runoff, evapotranspiration & pumping); and (iii) storage (reservoir storage, water stored above and below the water table). O'Grady et al (2002a,b) assessed water use by riparian vegetation along the Day River and revised Jollys' water balance with respect to evapotranspiration. They estimated the following means: rainfall (970 mm); runoff (220 mm); recharge (90 mm); transpiration by large trees (150 mm); understorey transpiration (510 mm); inflow from adjacent aquifers (1mm); water stored above and below the water table (6550 mm); and pumping for water supply purposes (0.4 mm or 0.2% of runoff). They concluded that the main hydrological characteristic of the Daly catchment is the great variability in annual and inter-annual rainfall, resulting in similar variability in both surface water runoff and groundwater recharge. Nevertheless, they did not examine trends in mean decadal hydrological characteristics (see below).

The following summary of catchment hydrology for the Daly River catchment is taken from Jolly (2002a): most water enters as rainfall in the wet season, and over most of the catchment inflows balance outflows; except after very intense rainfall events, true overland flow rarely occurs because of the highly permeable nature of the soil profile over most of the catchment; groundwater may discharge from off stream bank and aquifer storage areas depending on water levels during a river flow event, and this water discharges back into the river during the dry season; and groundwater discharges also from regional aquifer systems into adjacent creeks and rivers, and is a diffuse discharge-recharge process. Jolly (2002a) estimated the total annual runoff that represents the outflow from the Daly River catchment as that recorded at the river gauging station G8140040 near Mount Nancar. This station is located on the Daly River just above the upper tidal limit (see Figure 4.14), and total annual runoff values were derived from hydrographs, rating curves and flow gauging. The values for regional groundwater discharge are based on the mean annual instantaneous flow rate being 20% more than the minimum annual flow rate, and that for surface water runoff were derived by subtracting regional groundwater discharges from the total annual runoff.

Hydrology is likely a major driver of aquatic ecosystems in the Daly River catchment, hence seasonal, inter-annual and decadal patterns of surface flow and associated 'flood events' are examined in relation to rainfall, the major water balance input source. Basic statistical properties of inter-annual and seasonal flow in the Daly River at Mt Nancar gauging station between 1966 and 2006 (G8140040, see Figure 4.14) are summarised in Table 4.2. Data from the nearby gauging station (G8140041, 7 km upstream) were used to fill gaps in the record (see Appendix in Moliere 2008). Wet season flow (Oct–April) was 17 times dry season flow (May–Sept), and both are characterised by high inter-annual variability (75% coefficient of variation).

Table 4.2 Statistical characteristics of mean annual flow (ML, September – August) at Mt Nancar gauging station (combining G8140040 & G8140041, Moliere 2008) between 1966/67 and 2005/06 and, similarly, for the wet (Oct–April) and dry (May–Sept) season flows. Wet and dry season months are defined arbitrarily.

Flow (ML) Statistic	Total (Sept–Aug)	Wet season (Oct–Apr)	Dry season (May-Sept)
Number complete records	45	45	45
Mean (ML)	7 116 874	6 690 979	395 432
Standard Deviation	5 275 631	5 054 864	294 661
Coefficient of variation	0.74	0.76	0.75
Variance	2.721×10^{13}	2.498×10^{13}	8.489×10^{10}
Skewness	1.109	1.141	3.100
Kurtosis	3.663	3.861	14.948
Minimum	1 176 486	976 846	152 004
Maximum	22 051 760	21 213 129	1 858 965

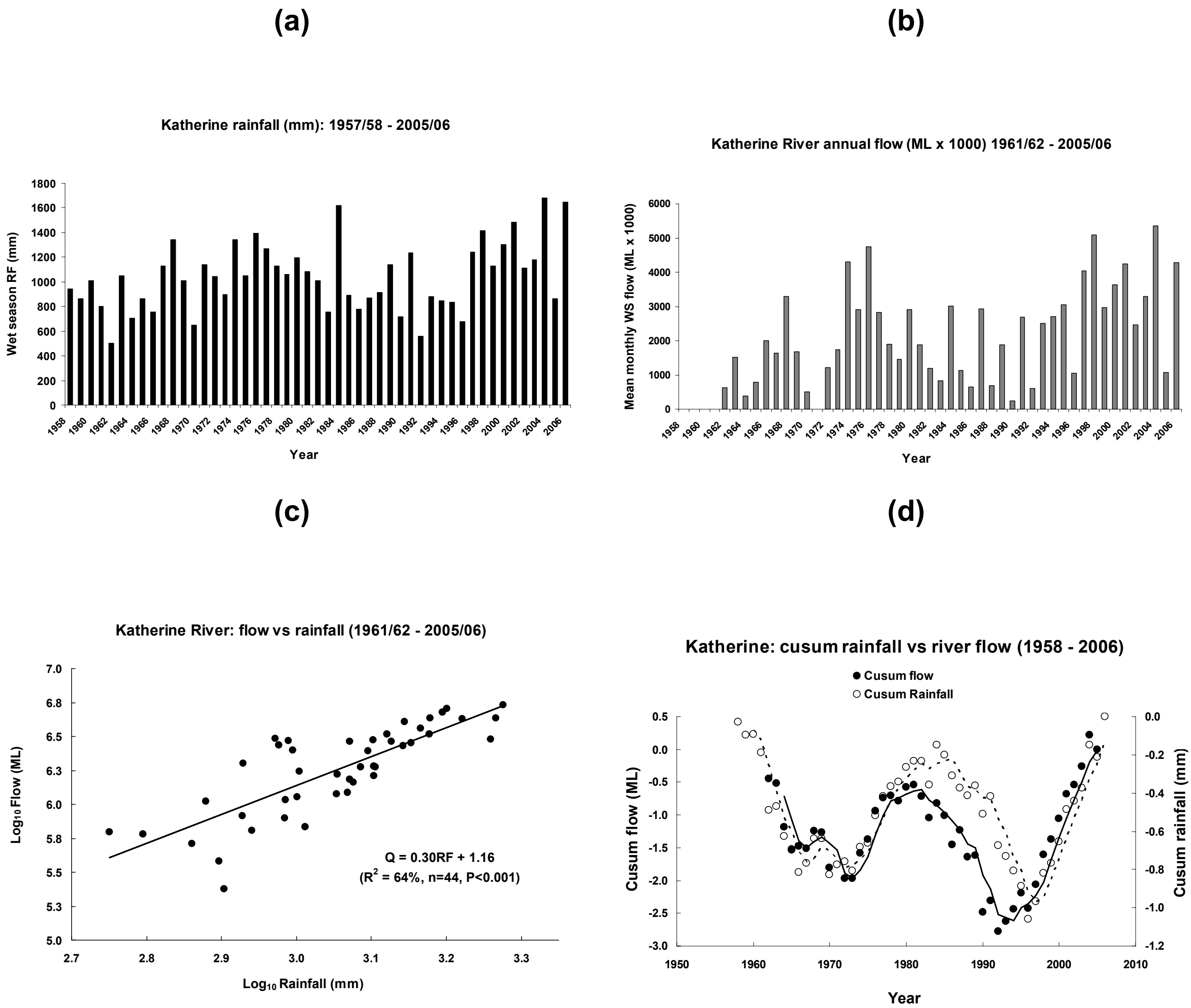
Erskine et al (2004) stated that dry season river stage is about 1m at most gauging stations on the Daly River, and that a flood is defined as a flood hydrograph with a peak stage greater than a gauge height of 7 m at most gauging stations. Additionally, they defined a floodplain flood as an event with a peak stage higher than 14 m at Mt Nancar gauging station. According to their definitions, over a 39 year period (1966/7–2004/5) channel floods occurred 87% of times, floodplain floods 46% of times and channel floods and not floodplain floods 41% of times (Table 4.3). These definitions imply that extensive flooding of the Daly River floodplain occurs every other year, with non-river flooding of the floodplain most likely occurring because of direct filling from localised rainfall, local groundwater recharge and/or local sub-catchment run-off.

Table 4.3 Characteristics of flooding of the Daly River and associated coastal floodplain (1966/67 to 2004/05) using stage height benchmarks described by Erskine et al (2004) and the mean stage height (m) at Mt Nancar gauging station (combining G8140040 & G8140041, Moliere 2008).

Flood event	Number complete records	Occurrence	% Occurrence
Floodplain flood	39	18	46
Channel flood	39	34	87
Channel flood & not floodplain flood	39	16	41

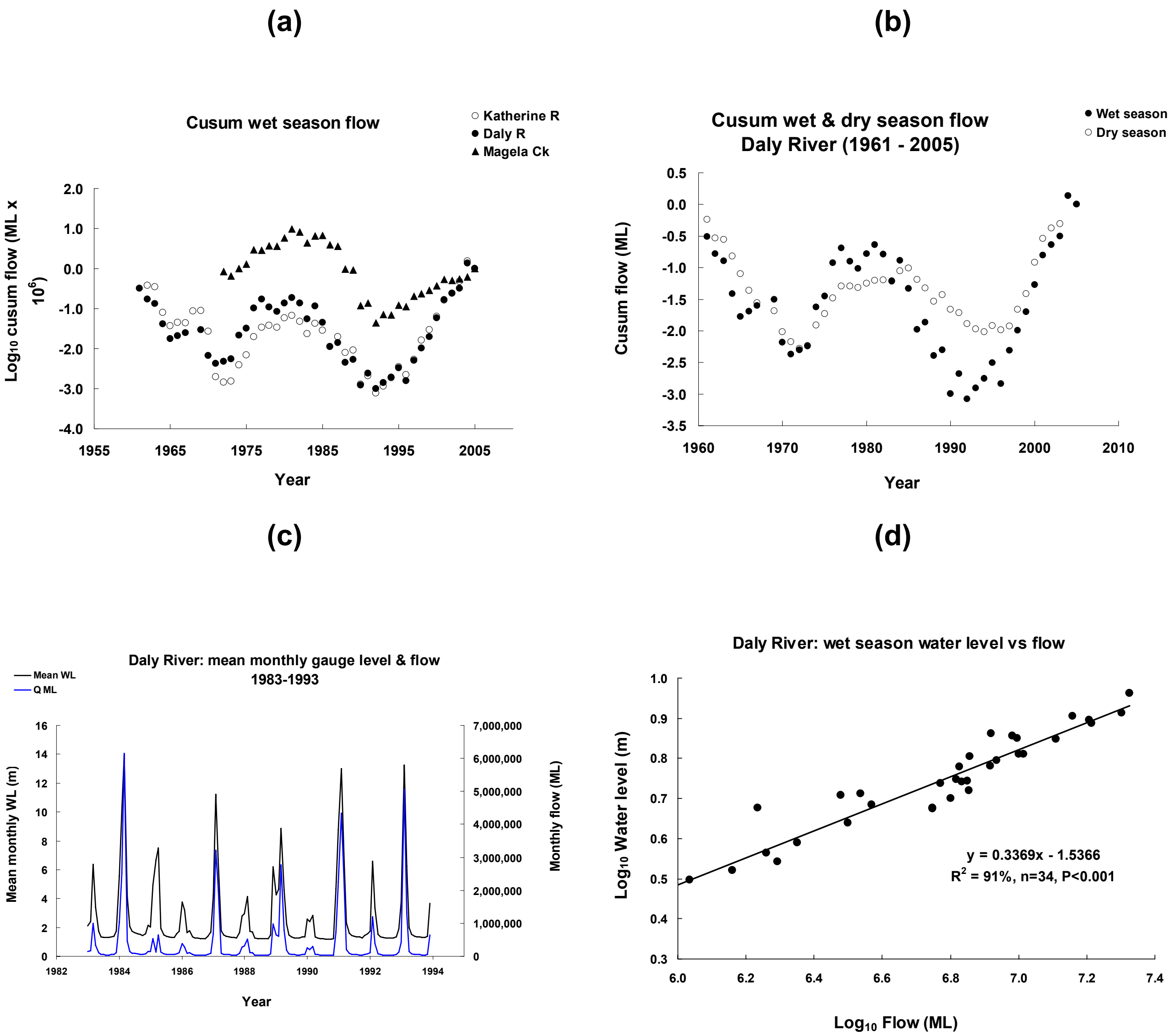
Long-term annual rainfall (mm) at Katherine Post Office and Katherine River flow (ML) show marked inter-annual variability (Figure 4.2 a & b, respectively; Oct–Sept period for

1961/62–2005/06), with 63% of the variability in flow explained by rainfall variability (Figure 4.2c). Distribution free cusum analysis (cumulative sum of the mean deviations; McGilchrist & Woodyer 1975, Mittag & Rhine 1993 p 664, Manly & MacKenzie 2003) of flow and rainfall over the same time period show similar approximate 20 y periods (Figure 4.2d); however, the instantaneous relationship between the two variables in the Daly River catchment is complex because their cusum trends are out of phase by about 3–4 years in the declining period between about 1980 and 1995, with rainfall preceding flow (Figure 4.3a).



Figures 4.2 a–d. (a) Annual (Sept-Aug) rainfall (mm) at Katherine PO (1957/58–2005/06) and (b) total Katherine River flow (ML x 1000) at G8140001 (1961/62–2005/06). (c) Regression between Katherine River flow and Katherine rainfall. (d) Cusum plots (cumulative sum of the mean deviations) of Katherine rainfall (RF mm, 1957/58–2005/06) and Katherine River flow (Q ML, 1961/62–2005/06) showing coupled rainfall-flow relationships but out-of-phase by 3–4 y between 1980 and 1995 because of response time lag between ground and surface water balance (3 point moving averages on both curves).

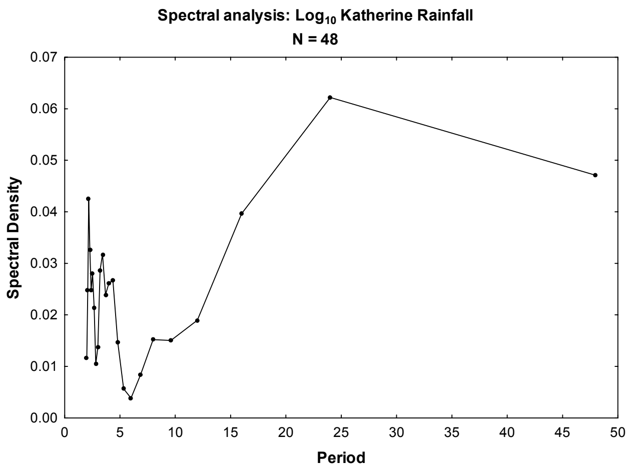
Cusum analysis of Katherine River (1961/62–2005/06, n=42 years) and Magela Creek (1970/71–2004/05, n=35 yrs) total annual flows also show approximate 20-year period trends that are concordant with Daly River flows (Figure 4.3a, Bayliss et al in prep. a) suggesting that, despite differences in catchment characteristics and regional variations in rainfall, regional trends in rainfall may act as a key driver that synchronises flow in the tropics over large spatial scales and decadal time scales. These streams represent some of the longest time series stream gauge data in the NT. Cusum plots of Daly River wet season (Oct–April) and dry season (May–Sept) flows (ML) exhibit similar period trends; however the trend for dry season flow lagged wet season flow by about 4–5 y between 1985 and 1995 (Figure 4.3b), possibly explaining the phase difference in rainfall and flow for Katherine River highlighted above. Whilst rainfall and flow are complexly related, flow and water level are highly synchronised (Figure 4.3c) and tightly correlated (Figure 4.3d). Figures 4.3c & d encompass the period when surveys were undertaken for magpie geese and their nests on the Daly River floodplain (1983–1993).



Figures 4.3 a–d. (a) Cusum plots of Katherine River (1961/62–2004/05, n=42 y), Daly River (1961/62–2004/05, n=42 y) and Magela Creek (1970/71–2004/05, n=34 y) total flows (ML, Sept–Aug) showing approximate 20-year flow periods.(b) Cusum plots of Daly River (1961/62–2004/05) wet (Oct–April) and dry (May–Sept) season flows (ML) showing similar period trends. Between 1985and1995 the period trend in dry season flow lagged wet season flow by about 4–5 y. (c) Trends in mean monthly water level (m) and monthly flow (ML) at Nancar gauging station encompassing the period of surveys for magpie geese nesting on the Daly River floodplain (1983–1993). (d) Tight regression relationship between mean monthly water level ($\log_{10}$ m) and monthly flow ($\log_{10}$ ML).

Cusum analysis tends to enhance longer-term trends at the expense of shorter-term trends, depending on the number of spectral signatures in distribution mixtures and their frequencies in relation to the length of the time series. Hence, Fourier analysis (see Brockwell & Davis 1996) in Statistica™ (StatSoft 2001) was used in combination with cusum analysis in order to tease out all spectral signatures in rainfall-flow data, and to ascertain return periods more precisely. The periodogram of spectral density (smoothed periodogram values) of Katherine rainfall (mm) vs. period shows two cycles at 4.5 y and 24 y (Figure 4.4a) and, two cycles for Katherine River flow (ML) at 8 y and 22 y (Figure 4.4b). Although spectral density values less than 5 years most likely incorporate a large component of white noise, the ‘trough’ in values at 4–6 years for both rainfall and flow correspond to periods of high El Niño–Southern Oscillation (ENSO) activity. Periodograms of cusum values of rainfall (Figure 4.4c) and flow (Figure 4.4d) enhance decadal trends in both variables showing that, on average across the time series, the 24 y and 22 y return periods respectively for rainfall and flow were the strongest signatures.

(a)



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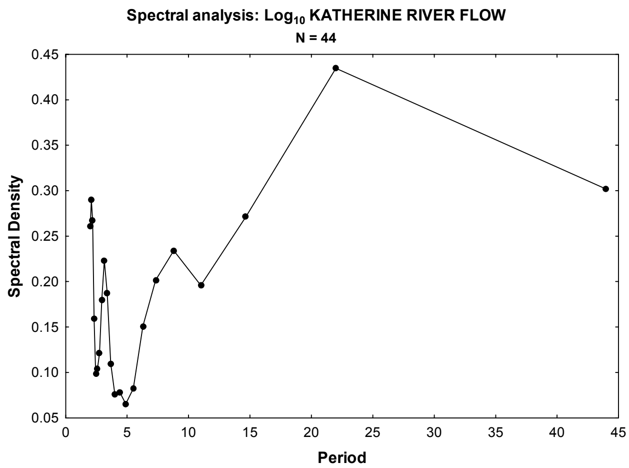
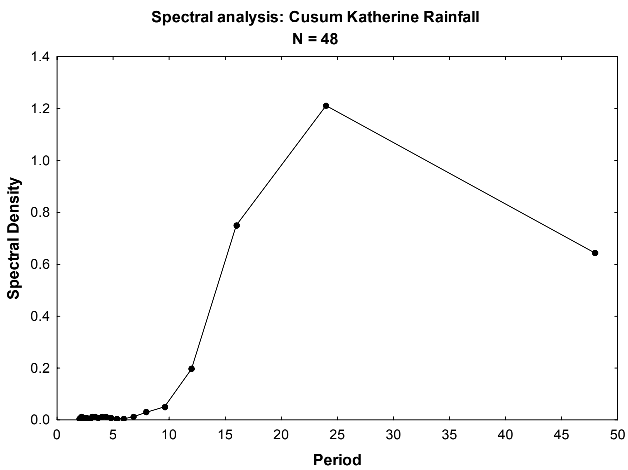
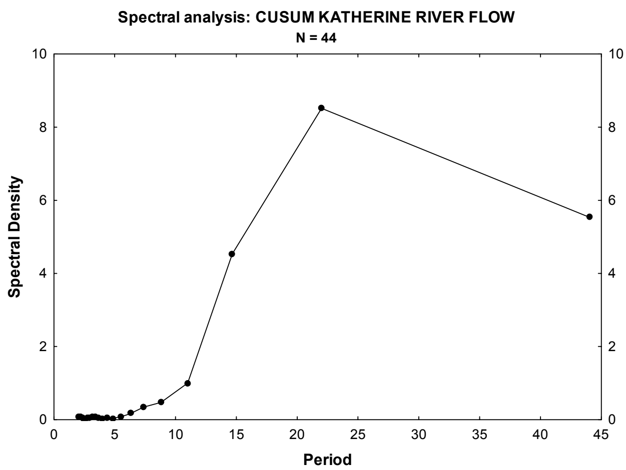


(c)



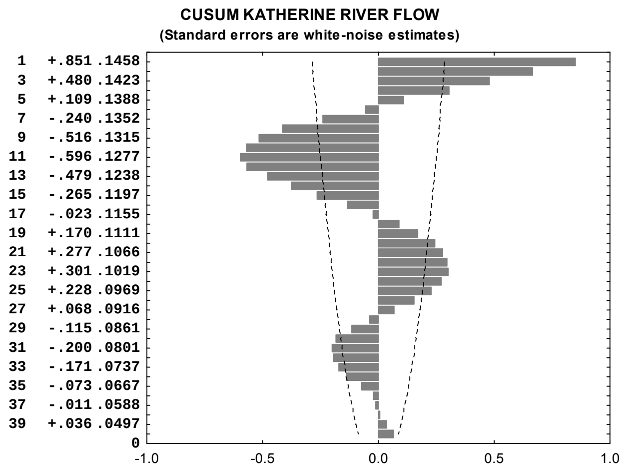
(d)



Figures 4.4 a–d. (a) Periodogram of spectral density of Katherine rainfall (mm) vs. period (y) using Fourier analysis showing two cycles in data at 4.5 y and 22 y, and (b) for Katherine River flow (ML) two cycles at 8 y and 22 y. Similar periodogram for (c) rainfall and (d) flow using cusum values that enhance the 24 and 22 year periods in rainfall and flow, respectively.

A correlogram (autocorrelation) of cusum flow values for Katherine River highlights the 22 y return period (Figure 4.5a) and, similarly, the correlogram (cross-correlation, Figure 4.5b) of Katherine River flow (ML) cusum values and Katherine rainfall (mm) cusum values show complex and probably coupled period trends. The rainfall-flow system appears tightly coupled, but because of complex discharge-recharge time lags in the groundwater storage components of the catchment water balance, the surface water flow signature could be out-of-phase by at least one period as suggested above (ie negative at 9–12 y & positive at 25–30 y).

(a)



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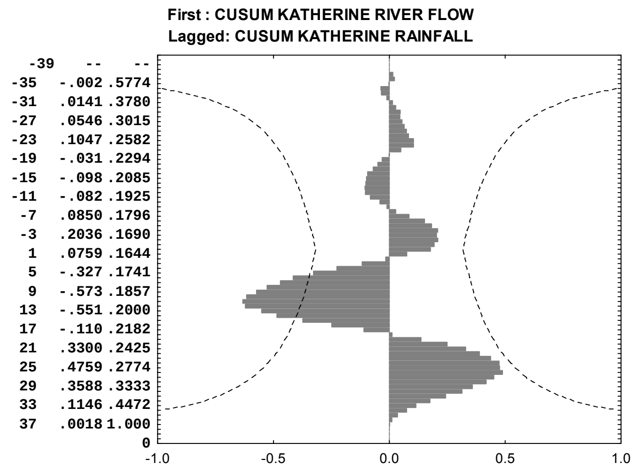


Figures 4.5 a & b. (a) Autocorrelation (Correlogram) of Katherine River cusum flow (ML) values highlighting the 20–25 year periods. (b) Cross correlation (Correlogram) of cusum values of Katherine River flow (ML) and Katherine rainfall (mm) showing that period trends in flow on average precede period trends in rainfall. Dashed lines are 95% confidence intervals and standard errors (3rd column of data left of figure) are white noise estimates.

Discussion

An additional three river systems in the NT (Adelaide, Elizabeth & Blackmore rivers) with river discharge data spanning greater than 40 years were examined for patterns in flow and show that the average return period for flow is approximately 22 y. However, despite the strong (R^2=64%) regression relationship between instantaneous Katherine River flow and Katherine rainfall, and the fact that they both exhibit similar decadal periods, the instantaneous relationship between the two variables is complex because their cusum trends were out of phase by about 3–4 y in the declining period between about 1980 and 1995, with rainfall preceding flow. This result was also reflected in the spectral analysis (24 y for rainfall & 22 y for flow). One possible explanation for the phase difference is the existence of interdecadal and decadal time lags between seasonal groundwater discharge and recharge rates, perhaps reflecting long retention times in groundwater storage components such as aquifers. In support of this suggestion, cusum analysis of wet and dry season flow over the same time period also exhibited similar period trends, but with the period trend in dry season flow lagging wet season flow by about 4–5y between 1985 and 1995. Tickell (2004) found that carbon dating and deuterium-O_{18} determinations of groundwater samples in the Oolloo Dolostone aquifer yielded broad ages between modern (<50 years) to several thousand years.

The time it takes for rainfall to travel through a catchment, or 'flushing time', is a fundamental hydrological parameter (Kirchner et al 2000). Although catchments are spatially complex and subsurface flow is invisible, Kirchner et al (2000) argued that they can, nevertheless, be characterised by a distribution of travel times reflecting the diverse flow paths (~ fractal geometry) that rainfall can take to the stream, and basically approximates a power law distribution. Whilst prediction of surface flow based on rainfall may be inherently uncertain because of complex system lags between groundwater and surface water flows spanning decades, especially in the discharge phase, it may nevertheless explain why weak or 'noisy' period trends in rainfall may lead to marked (or less 'noisy') period signatures in average flow. Taylor et al (2002) argued that ecosystems sensitive to external influences associated with nonlinearity can lead to amplification of weak climatic signals, such as vague rainfall patterns that encompass the influences of period trends in ENSO and/or the Interdecadal Pacific Oscillation (IPO or the closely related Pacific Decadal Oscillation PDO).

However, whilst rainfall and flow are complexly coupled, flow and water level are highly synchronised and exhibit a tight correlation. Hence, river flow drives water level changes that in turn determine the nature and extent of flood events and, ultimately, the seasonal dynamics of aquatic floodplain and in-stream ecosystems.

The existence of average return periods in flow represents a new finding for the NT, with implications for our understanding and management of tropical aquatic freshwater ecosystems. Even so, there are many pointers in the literature, both in Australia and overseas, that suggest decadal patterns in flow are likely because of decadal and multi-decadal climatic variability in the behaviour of global climatic systems that influence proximate drivers such as rainfall (see Rodrigo et al 2000). For example, the Pacific Decadal Oscillation (PDO) is a pattern of Pacific climate variability with a similar regional climate signature as that for the ENSO (Latif 1998, Power et al 1998, Nigam et al 1999, Power et al 1999, Mantua et al 1997, Zhang et al 1997, Power et al 2006). PDO regimes may persist for 2–3 decades, whereas ENSO phenomena have periods of approximately 3–5 y duration. The PDO index is define by Mantua et al (1997) as the November-March average of the leading monthly principal component from a Principal Components Analysis of Pacific Ocean sea-surface temperature (SST) above 20^{o}N latitude. Closer to home, Johnston and Prendergast (1999) used the cusum analysis of Carter (1990) to show that rainfall between 1932 and 1984 at Oenpelli in the Alligator Rivers Region (ARR) had an approximate 20 y period; rainfall trended to increase on average for a 10y period and then decrease on average for a 10 y period. Viles and Goudie (2003) reviewed inter-annual, decadal and multi-decadal climatic variability in relation to hydrogeomorphic processes, such as stream flow and sediment yield, and concluded that a better understanding of such processes is essential in differentiating complex relationships between natural geomorphic and anthropogenic changes. From a small sample of overseas literature: Hidalgo and Dracup (2003) found that hydroclimatic variations of the Upper Colorado River Basin were related to the ENSO and PDO, and a similar result was found by Hamlet and Lettenmaier (1999) for the Columbia River; and Rîmbu et al (2002) found that decadal variability of the Danube River flow was related to the North Atlantic Oscillation (NAO). Detecting changes in mean flow using inherently noisy time series data is challenging, and distribution free cusum analysis has the ability to enhance the signal from the noise. Nevertheless, Radziejewski and Kundzewicz (2004) suggested that, with enhanced climate change, changes in hydrological processes may be stronger and last longer and, hence, the likelihood of change detection across globe should increase.

Whilst many studies have linked PDO to stream flow patterns they are not always simple and direct. For example, Neal et al (2002) found that the effect of the PDO on south-eastern

Alaskan catchments differs from other regions of the coast of Pacific North America in that monthly/seasonal discharge patterns changed dramatically with the switch in PDO mode from warm and cold phases (or positive & negative polarities of the index, respectively). In contrast, however, annual discharge did not. Nevertheless, Delcroix et al (2007) reported a PDO-like signal and long-term trends in sea-surface salinity, and suggested a link between subtle variations in these signals and global climate change.

In northern Australia we lack understanding of how climate forcing factors such as the ENSO (via the SOI) and IPO (via the PDO) may interact to influence tropical rainfall-flow events at decadal time scales. Recent studies (eg Power et al 1999, Power et al 2006) indicate that the strength of the correlation between ENSO and precipitation may vary with the phase of the PDO and between different climate regions. Bayliss et al (2007) recently examined trends in catchment rainfall and flow of four major NT streams in relation to trends in the ENSO-IPO interaction using four *a priori* classes (PDO<0 & >=0 in combination with SOI<0 & >=0). Percentage of mean rainfall in each catchment exhibited similar linear increases with SOI (combined regression: R^2=74%, n=12, P<0.001), with the highest values (106%-111%) found in the PDO <0 and SOI>=0 class. Percentage of mean flow of each stream exhibited similar convex nonlinear trends on SOI (combined 2nd order polynomial regression: R^2=88%, n=12, P<0.001), with the highest values (118%-154%) also found in the PDO<0 and SOI >=0 class. The ENSO-IPO interaction, therefore, appears an important influence on tropical rainfall-flow events and warrants further investigation because of implications for long-term water resource management. Bayliss et al (2007) concluded also that climate forcing factors such as the ENSO and IPO may be modulated by anthropogenically-induced climate change.

4.1.2 Modelling approach

The quantitative ecological risk analysis (QERA) reported here flows from the semi-quantitative risk analysis presented in Chapter 3 that is underpinned by application of the novel spatially explicit Relative Risk Model (Bartolo & van Dam 2006). The handful of key threats and assets that were selected *a priori* for quantitative analysis (Table 4.1), and affirmed through stakeholder consultation and participation in ranking relative risks, were therefore selected from a process essentially and appropriately driven by stakeholder views. However, and as expected, a range of stakeholder perceptions on how risks are ranked relative to each other was found, and this class of uncertainty was accounted for by using interval maths (see Ferson 2002) and sensitivity analyses (see Chapter 3). Hence, once the assets and threats are prioritised and selected for more detailed quantitative analysis, the choice of method (or methods) will be essentially driven by the purpose of the assessment (ie choice of ecological assessment endpoint) in combination with the availability and quality of different types of data (ie choice of measurement endpoint). As there are many different approaches to QERA depending on purpose and data at hand (see Burgman 2005), we devised a selection process to facilitate the choice of the most appropriate methodology (Figure 4.6).

Conceptual models

All steps in a risk assessment need to be guided at the outset by decent conceptual models (Burgman 2005), and this caveat applies to both qualitative and quantitative ecological risk assessments. Conceptual models are abstractions about how we think the world works in order to answer specific questions to assist decision making, and usually takes the form of box and arrow graphs. Drewery et al (2006) suggested that conceptual models are basically process-based models hypothesising testable cause-effect relationships between stock-flow or storage-transport pathways. They suggested also that empirical models are basically data-

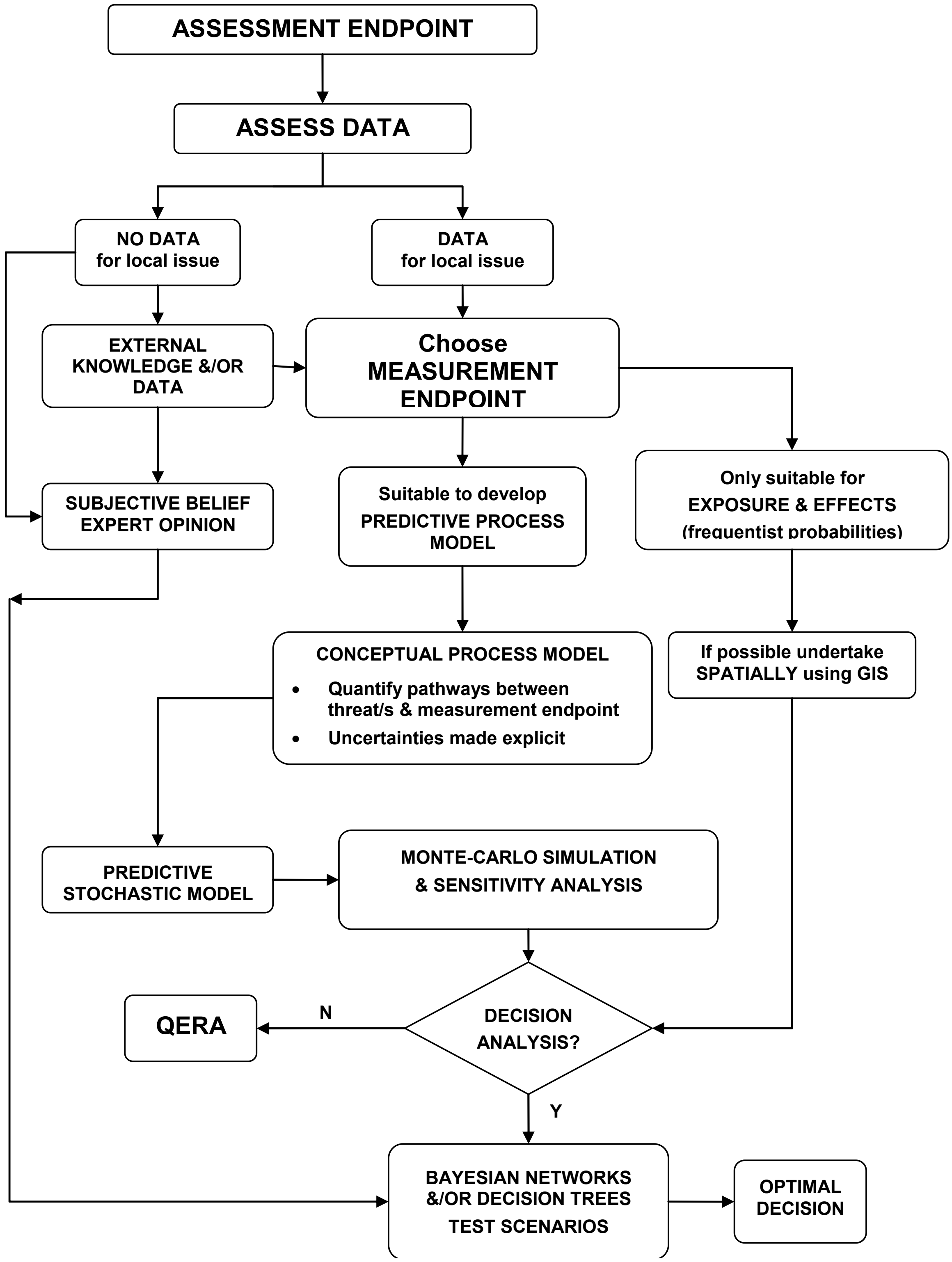


Figure 4.6 Choice of methods for Quantitative Ecological Risk Assessments determined by data availability and choice of measurement endpoint. QERA = Quantitative Ecological Risk Assessment.

based models and may suffice as statistical tests of hypotheses made explicit in conceptual models. In this Section we define conceptual models as unparameterised process models, and the empirical data-based models of Drewery et al (2006) are analogous to our parameterised stochastic process models.

Whilst there are many different types of ecological models and analytical approaches (Maynard Smith 1973), they usually compromise between complexity and utility. Hence, the purpose of a conceptual model, or any model, determines its structure and limits (Burgman 2005). For example, with an eye on predicting nuisance algal blooms, Harris (1997, 1999b) developed a conceptual theoretical framework for modeling events in aquatic ecosystems as coupled nonlinear processes in catchments, water columns and sediments, and borrowed ideas from the behaviour of complex adaptive systems.

In addition to capturing relevant current knowledge and expertise about cause-effect linkages and interactions, conceptual models used for risk assessment should communicate the full range of assumptions and uncertainties that underlie the model. Even among technical experts there are often divergent views about cause-effect relationships and, hence, structural uncertainty in models often reflects the different ideas encountered about how ecosystems work. The elicitation about causal models is, therefore, often a subjective process and the search for the right structure is often intuitive (Burgman 2005). Hence, all risk assessments in this section are preceded by a conceptual model that first defines the purpose of the analysis and, secondly, attempts to make explicit all assumptions about cause-effect relationships and possible interactions with respect to threats to assets via the measurement endpoint.

Conceptual model assumptions are treated as hypotheses to test and, hence, quantitative predictive models are developed and confronted with observed data. Predictive statistical models in themselves, however, do not constitute hypotheses (Hilborn & Mangel 1997). The predictive models and associated model errors are then used in 'what if' scenario simulations embedded in Bayesian Networks (BNs) to further clarify relationships between assets and threats within a decision making framework. It should be remembered though, that BNs are essentially conceptual models of a problem, and so should encompass the best available knowledge and/or expert views. This methodology is trialed also (Figure 4.6 & see below).

Stochastic ecological process models – knowledge uncertainty vs. natural variability

Young (1998, 2003) advocated a 'top-down' and data-based (empirical) mechanistic approach to model rainfall-flow relationships at the catchment scale, and is in contrast to the standard 'bottom-up' deterministic approach that comes complete with an *a priori* model in mind of how the system works. We adopted a similar approach with respect to testing hypotheses associated with the conceptual model itself (ie confronting models with data), but use statistics to capture model uncertainties and stochastic processes in empirically-derived mechanistic relationships. Where data were sufficient General Linear Models (GLMs; see Rushton et al 2004, Guisan & Zimmermann 2000), mostly in the form of a multiple regression equation, or a nonlinear regression equation, were used to predict the measurement endpoint as a function of environmental conditions including some measure of the threat being assessed. Hence, for the natural system being modelled, uncertainty originates from lack of perfect knowledge of the ecological relationship (eg magpie goose nests vs. flow, barramundi catch vs. flow), whilst variability arises from natural variability in driving environmental parameters (eg. flow).

Cohen et al (1996) used two-stage Monte Carlo simulation techniques to separately characterise variability and uncertainty in risk analysis, and second-order Monte Carlo uncertainty-variability analysis using correlated model parameters has recently become popular in risk assessment to separately characterise the two (see Wu & Tsang 2004). Hence, with respect to the stochastic modelling approach adopted here, innate parameter variability and model uncertainty were treated separately and modelled simultaneously. Where linear or nonlinear regression models were used, knowledge uncertainty was assumed captured in the regression model error term. For example, if a multiple regression equation predicting fish catch on flow explained 80% of the variability in observed data (R^2), then knowledge uncertainty is assumed to be the unexplained variance or 20% (ie $1-R^2$, the coefficient of determination). For the purposes of our QERAs this assumption should suffice, but in reality statistical prediction models and consonant ecological models (=model parameters consonant with reality) are not the same (Caughley 1981). For example, prediction error may be low in a statistical model such as a 4^{th} order polynomial regression, but the cause-effect knowledge that explains the data may be totally unknown.

With the above caveats in mind, statistical analyses were undertaken in Statistica™ (StatSoft 2001) and model simulations were undertaken in an Excel™ - @Risk™ software environment (Pallisade 2002b). All variables were examined for normality prior to analysis (via normal probability plots, Kolomorov-Smirnov & Shapiro-Wilk's W tests; StatSoft 2001). Where appropriate $\log_{10}$ transformations were used to normalise ordinal data (see Limpert et al 2001), and arcsine transformation (of √proportion) for percentages (Zar 1984). Multiple regression analysis was used to test multiple working hypotheses about the influence of X-independent variables chosen *a priori* on the Y-dependent or response variable, usually the measurement endpoint in risk assessment. However, the effect of one independent variable may be influenced by the levels of other intercorrelated variables, hence there would be no single level of importance. If stepwise regression was used to search for the best sub-model, then a step-down procedure was used in preference to a step-up procedure to minimise the number of possible models to test. Nevertheless, the regression P-value was adjusted for protection against Type I error using a Bonferonni correction (Wilkinson 1979). Partial regression plots were used to examine the direction, magnitude and distribution of data in the multiple regression equation. These plots describe the effects of those variables on the response variable when the intercorrelated effects of all other variables are statistically held constant. This method was used in preference to a GLM incorporating many complex interaction terms for independent variables.

Stochastic process models that included a model error term were then constructed by replacing mean parameter values with probability density functions (pdfs), which were chosen as the 'best fit' to the frequency of observed data from a range of candidate statistical distributions using Goodness of Fit (GoF) tests in Best Fit™ (Pallisade 2002a). We did not validate or test model predictions with independent data sets (see Section 4.6), hence we assume that knowledge uncertainty is essentially captured by the model error term and strictly applies to the conditions of the observed data set. In contrast, the pdfs capture innate variability of model parameters. Uncertainty and variability were therefore incorporated into the QERA and examined separately using Monte Carlo (MC) simulation. The importance of parameter inputs on risk outputs was examined using sensitivity analysis of all MC outputs. All pdfs were randomly sampled 10 000 times or more to derive a stable mean value. There was little change in mean values when > 1 simulations were run (ie 2 to 100) and, hence, results reported here only apply to the first simulation of 10^3 random samples. @Risk™ simulation results include graphical displays of the distribution of all possible results from outputs (eg via frequency distributions & cumulative probability distributions), and generates sensitivity and scenario reports that help identify those inputs that are most critical to outputs. Sensitivity analysis is undertaken using regression analysis, whereby sampled input variable values are regressed against output values, leading to a measurement of sensitivity by input variable. Results of the sensitivity analysis are displayed as a 'Tornado' type chart, with longer bars at the top representing the most significant input variables in a positive or negative direction (Pallisade 2002b).

For comparative purposes the method of Wu and Tsang (2004) was trialled for barramundi data sets by replacing 'forced' and often poorly fitted pdfs with smoothed Kernel distributions (see Silverman 1990 for methodology). Monte Carlo simulation was again used to draw a random sample from the Kernel distribution (as apposed to the pdf), and results compared to the procedure outlined above.

All model and simulation results were then incorporated into simplified BNs using Netica™ (Netica 1997) at different levels of complexity (eg the regression equation between condition & threat, the pdfs or the model inputs/outputs re-defined as state levels). Bayesian Networks constructed in Netica, whilst not amenable to advanced modelling techniques used in an Excel™ - @Risk™ environment, is a much more powerful communication tool because it is graphically based and, hence, more suitable as a decision making tool for stakeholders (see below). The cascade effect of a change in variable state, or the subjective value of a decision, and/or the uncertainty associated with it, can be observed.

Bayesian Networks and Decision Trees

The use of quantitative Bayesian Networks (BNs) as a risk management tool for stochastic complex ecosystems that are highly variable and poorly understood has recently become very popular, particularly in the face of uncertainty from multiple threats under multiple management treatments (Borsuk et al 2001, 2002a, Hart et al 2005). Additionally, BNs are flexible in that they can integrate quantitative information with qualitative expert knowledge and, hence, facilitate stakeholder engagement and communication (Baran & Jantunen 2004). Bayesian Networks have proved versatile in almost every ecological field with a decision problem that involves taking risks in the face of uncertainty, variability and complexity. For example: Borsuk et al (2002b) used BNs to model complex estuarine eutrophication processes because its graphical structure explicitly represents cause-effect assumptions between system variables that may be obscured by other modelling approaches; Lamon and Stow (2004) used a Bayesian Classification and Regression Tree (BCART) approach to link multiple environmental stressors to biological responses, and to quantify uncertainty in model predictions; Lin et al (2004) used Bayesian analysis to account for the combined uncertainty and variability of model parameters in a crayfish bioaccumulation model; Marcot et al (2001) used BNs to model habitat and population viability of selected 'at-risk' fish and wildlife species; Bryan and Garrod (2006) used rapid field assessments combined with BNs to prioritise investment in watercourse protection; and Reckhow (1999) argued that probabilistic BNs should be used for surface water quality assessments because traditional predictive ecosystem models can never hope to simulate nature.

Hence, given the good rap above, BNs are trialled here and assessed for utility. For each ecological asset under risk assessment in the Daly River catchment, therefore, a BN was developed that explicitly identifies links between hypothesised causes and effects, and highlights complexities and uncertainties in the system. BNs can embrace simple frequentist risk probabilities of exposure and effects, mechanistic or stochastic process models, predictive statistical models, expert opinion or combinations of the above. Hence, BNs recognise the dual nature of probability through chance (via frequentist statistics) and belief (via Bayesian statistics &/or expert opinion) and have a lot going for them if we don't get too carried away.

Influence Diagrams and Decision Trees are trialled also where there is a clear need to optimise a decision and where appropriate data exist (eg the barramundi fishery). Influence Diagrams are Bayesian Networks that support decision optimisation based on utility values assigned to different possible outcomes of the decision (Cain 2001), and are excellent for showing the relationship between events and the general structure of a decision clearly and concisely. In contrast, Decision Trees outline the chronological and numerical details of the decision. Hence, Influence Diagrams or BNs produce a compact summary of a problem, and Decision Trees can show the problem in greater detail (Pallisade 2000).

Scenario simulation and risk management

The influence of different interventions used to manage risks to the ecological assessment endpoint (usually a condition metric along the species-population-habitat continuum) is examined using 'what if' scenario simulation. Hence, the BNs presented here may form the start of an adaptive Decision Support System (DSS) that can be improved over time, especially with additional and/or better information flowing from targeted and well-designed future monitoring programs. The benefits of using spatially explicit QERA methods and Bayesian Networks as decision making and communication tools for environmental managers are highlighted throughout this section of the report.

4.2 Risks of water extraction and weeds on floodplain health

Executive summary

The Daly River floodplain contains important nesting habitat for the iconic magpie goose, and their breeding colonies have supported up to 36% of the NT population. The floodplain is also an important dry season refuge for magpie geese. Flow regimes that trigger floodplain floods from the Daly River are strongly correlated to peaks in nest production and, most likely, food availability throughout the dry season. Water extraction has been identified as a potential key threat to floodplain environmental flows and, hence, the condition or health of floodplain habitats. Wetland weeds have also been identified as a key threat to floodplain values, in particular mimosa and para grass; they reduce plant biodiversity and displace sedge and grass communities that magpie geese depend on for nesting success and dry season survival (Whitehead et al 1990).

The aim of this section is to develop a QERA framework to assess the 'health' of the Daly River floodplain under the combined effects of potential flow extraction and the current effect of wetland weeds. Three ecological assessment endpoints were used: (i) the health of magpie goose nesting success in the wet season (in relation to potential flow extraction and extent of weeds); and (ii) the health of magpie goose dry season refuge habitat and (iii) plant biodiversity (in relation to weeds only). A stochastic process model was developed to predict reductions in nest density from simulated flow extractions (0–100%), and results suggest that nest density will decline in direct proportion to flow extraction. A spatially explicit QERA of weeds on magpie goose nesting and dry season refuge habitats, and plant biodiversity, suggests that ecological risks from mimosa and para grass may be manageable because only 16% and 1.5%, respectively, of the floodplain was exposed at a 100% cover in 2003. Mimosa control programs since 2003 have most likely prevented colonisation of the whole floodplain. A Bayesian Network was used to assess the independent and combined risks to floodplain health from simulated flow extraction and weeds, and examined four scenarios: a 0% and 20% flow extraction in the absence and presence of weeds. A simulated 20% flow extraction had little overall influence on floodplain health either in the presence or absence of weeds.

The major influence was the extent of floodplain weeds; hence the BN was extended to compare the costs and benefits of different weed control scenarios. Results show that a control strategy that aims for a 10% residual cover of mimosa substantially increased the probability of the floodplain being in ‘Good’ condition from 4% to 72%, at an initial cost of $750 000.

We conclude that, whilst stakeholder participation is absolutely essential in developing conceptual models at the start of the QERA process, it is also critical at the very end when decisions based on value judgments about ecosystem ‘condition’ or ‘health’ need to be made. Given that our risk assessment exposed weeds as a high level real threat to floodplain values, and the fact that in-stream and floodplain aquatic ecosystems are intimately connected, we recommend that a more formal and detailed risk assessment be undertaken using the National Post Border Weed Risk Management protocols (Standards Australia/Standards New Zealand/CRC for Australian Weed Management 2006).

Technical summary

1 Magpie geese exhibit approximate 20 y population cycles in the NT that are coupled to similar periodicities in mean long-term flow for NT rivers such as the Daly River. River flow drives the spatial and temporal dynamics of magpie geese at regional and decadal time scales, most likely through its direct influence on floodplain vegetation dynamics. Flow regimes that trigger floodplain floods on the Daly River floodplain are strongly correlated to peaks in nest production and, most likely, to the availability of food throughout the dry season.

2 A conceptual model was first constructed to show explicit cause-effect relationships between threats and the ecological assessment endpoints. Multiple regression analysis was then used to develop a stochastic process model in order to predict nest density from natural flow regimes and the availability of geese to breed (R^2=67%, n=11, P<0.001). Monte Carlo simulation was used to predict nest density under increasing wet season flow extractions (0–100%). Model uncertainty and variability were explicit and treated separately, and sensitivity analysis was used to determine the relative importance of prediction variables and model error. Despite a high level of model uncertainty, simulation outputs predict that nest density will decline in direct proportion to flow extractions; for example, a 20% flow extraction will reduce average nest density by 20%.

3 The distribution and abundance of mimosa and para grass on the Daly floodplain were surveyed by the NT Weeds Branch in 2003. Hence, a spatially explicit QERA of both wetland weeds on magpie goose nesting and dry season refuge habitats, and plant biodiversity on floodplains, was undertaken to aid on-ground management of risks. Results indicate that ecological risks from mimosa and para grass may be manageable because only 16% and 1.5%, respectively, of the floodplain was exposed at a 100% cover in 2003. Mimosa control programs since 2003 may have stopped further spread and, most likely, prevented colonisation of the whole floodplain.

4 A Bayesian Network for floodplain health was constructed that incorporated the three ecological assessment endpoints (see Table 4.1). The standard approach was first adopted in that variable ranges were converted to state levels (Low, Medium & High). However, probabilities of each state level entered into the Conditional Probability Tables (CPT) were often arbitrarily determined from poorly fitted statistical frequency distributions (pdfs) and, combined with the necessity to populate large CPTs, involved much unsatisfactory guess work and creative invention. Hence, large unwieldy CPTs of

intersecting child nodes were avoided by replacing them with equations that use outputs (eg other equations, pdfs or constants) from parent nodes as input variables.

5 The predictive stochastic models developed to simulate the effects of flow extraction on nest density were incorporated into the BN, along with the QERA for wetland weeds that characterised impacts on plant biodiversity, magpie goose nesting habitat and dry season refuge habitat. Four scenarios were simulated to assess the independent and combined risks to floodplain health from flow extraction and wetland weeds, and these were 0% and 20% flow extraction in the absence and presence of weeds. A simulated 20% wet season flow extraction had little overall influence on floodplain health either in the presence or absence of weeds. The major influence on floodplain health was the extent of weeds.

6 The BN was extended to include nodes that allowed examination of the costs and benefits of different weed control scenarios. A control strategy that aimed for a 10% residual cover of mimosa significantly increased the probability of the Daly River floodplain being in 'Good' condition from 4% to 72%, at an initial cost of $0.75 million.

4.2.1 Introduction

Landform and vegetation communities on the Daly River floodplain

The Daly River floodplain, part of the Daly-Reynolds Floodplain-Estuary System, is one of the largest floodplains in the NT (Appendix 4.8) and has the largest catchment of any major freshwater floodplain system (ANCA 1996). Wilson et al (1991) identified 38 plant communities on the Daly River floodplain that included grasslands, sedgelands, paperbark forests and woodlands, and open water and mangroves. They estimated that about 71% of floodplain plant communities in the NT occur here, although none are endemic.

Broad floodplain vegetation types vary between different river systems in the NT (Frith & Davies 1961, Finlayson et al 1988, Whitehead et al 1990), most likely because of variations in the geomorphologic structure of the floodplains and past and current land use practices. Vegetation composition on floodplains, however, show marked variation with changes in micro-topography, principally through its effects on hydrology (Bowman & Wilson 1986). Distinct vegetation types are often associated with geomorphic features of the plains, such as high and low lying depressions, palaeochannels and drainage depressions (Story et al 1976). Salinity is an important factor that influences the composition of floodplain vegetation communities, as most floristic groups are associated with distinct salinity and water depth regimes. In general, diversity is low at wet and saline sites and highest in the drier sites (Wilson et al 1991). The most common community type found on the Daly River floodplain (as on all NT floodplains) is wild rice (*Oryza* spp) grasslands, followed by *Ischaemum australe* and the sedges *Eleocharis dulcis* and *E. sphacelata*. These floodplain plants are important nesting and dry season habitat components of magpie geese. For example, geese prefer *E. sphacelata* to build stages and nests (Frith & Davies 1961), although on the Daly River floodplain *I. australe* was used extensively also (Bayliss & Yeomans 1990). The bulbs of *E. dulcis* are a key dry season food for magpie geese, and *Oryza* spp is a key food source for emergent goslings and adults (Frith & Davies 1961, Whitehead & Saalfeld 2000). Marked annual and seasonal variations in floodplain vegetation composition and abundance occur also due to the alternating wetting and drying cycle between seasons (Finlayson et al 1990). Hence, floristic changes are also strongly associated with variations in annual rainfall and flow (Finlayson et al 1993). Although exotic weed species make up only a small proportion (<0.5 %) of the total number of species encountered on floodplains, they pose serious threats to their conservation values (see below).

Rainfall, river flow and magpie goose ecology

Rainfall in the seasonal tropics of northern Australia exhibits marked annual variability in both the timing and amount. Taylor and Tullock (1985) found that much of the between year variation was attributed to rainfalls occurring during the dry season and the dry-to-wet season transition. They therefore suggested that variation in timing of rainfall may be more important for driving aquatic ecosystems than the actual amount. Whitehead and Saalfeld (2000) found that, whilst the density of magpie goose nests on the Mary River floodplain fluctuated markedly between 1988 and 1993, failing almost completely in El Niño years, both nest density and nesting dates were tightly correlated with transitional rainfalls, supporting the suggestion of Taylor and Tullock (1985).

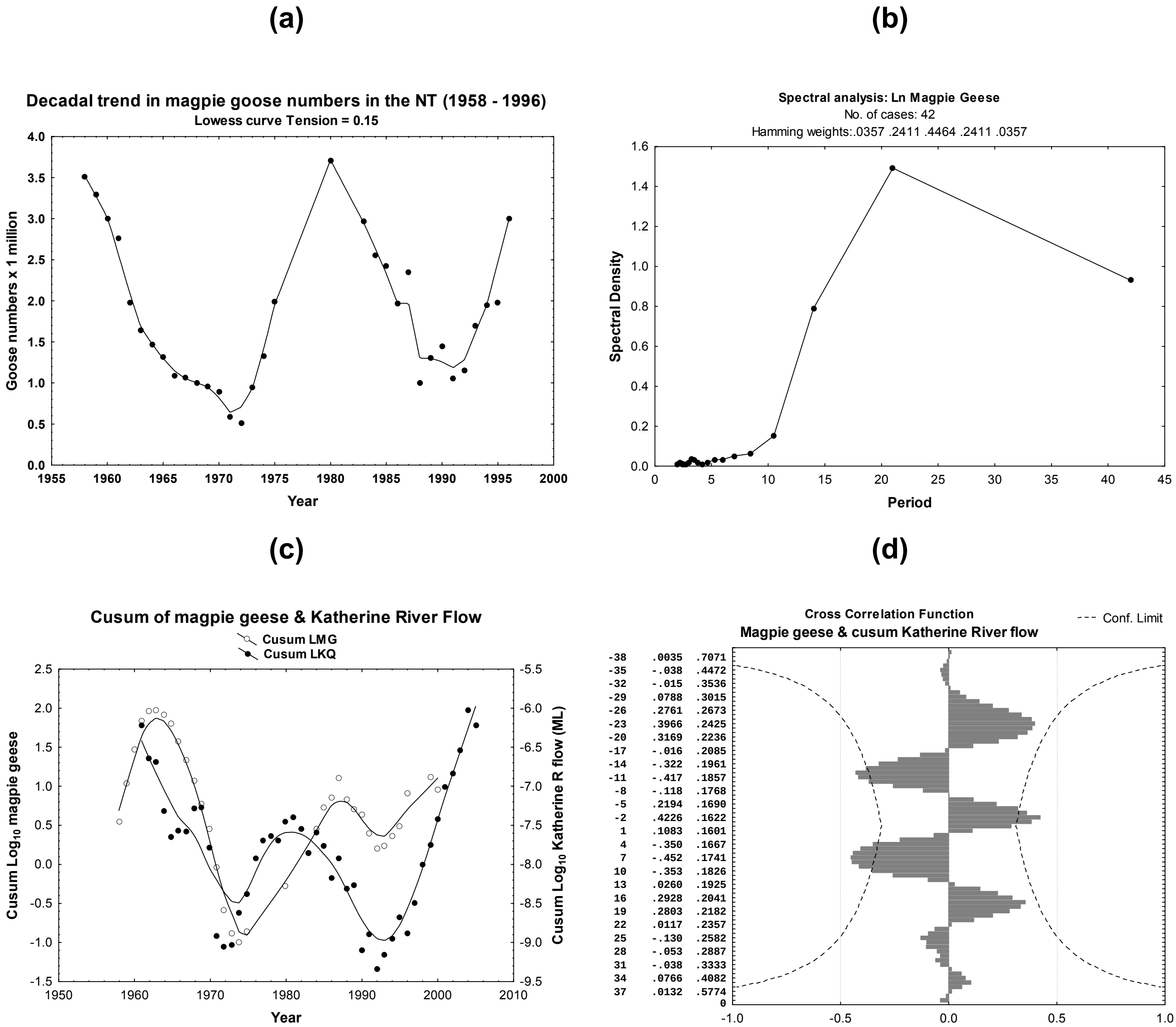
However, Bayliss (1989) showed also a tight correlation between the population dynamics of magpie geese and deviations in mean regional rainfall in the NT, suggesting that higher-order dynamics are valid also, but at spatial scales greater than floodplains within catchments. This hypothesis may also apply to higher-order dynamics over longer temporal scales, such as decadal and even inter-decadal scales, a point identified by Whitehead and Saalfeld (2000) in their comprehensive analysis of magpie goose nesting phenology. In support of this suggestion, Bayliss et al (2006) demonstrated that magpie geese across the NT exhibited approximate 20-year population cycles that were coupled to similar and generally coherent periodicities in flow of Katherine River, Daly River and Magela Creek. Figure 4.7a plots the period trends in goose numbers between 1958–1996 and combines the published early survey data of Tullock and McKean (1983; 1958-80) and Bayliss and Yeomans (1990b, 1984–1986) with later NT Parks and Wildlife Commission (P&WC) survey data between 1987 and 1996 (P&WC 2003). The same aerial survey design and methods were used between 1983 and 1996 (see Bayliss & Yeomans 1990a), whilst Tullock and McKean (1983) used a different aerial survey design. However, both survey types estimated total goose numbers and, hence, the two time series are combined. Whilst the degree of bias (ie the difference in magnitude of estimates) between both survey methods is unknown, it should not affect the underlying periodicity in numbers.

The most recent P&WC survey data (1999, 2000 & 2003) were excluded from analysis because the number of floodplains surveyed was reduced to ‘core areas’ (K Saalfeld, pers comm, P&WC 2003). Figure 4.7b shows the periodogram of spectral density of geese numbers on period (years) using Fourier analysis, and pinpoints an average 21 year period. Figure 4.7c shows cusum trends in magpie geese numbers and Katherine River flow (ML, 1958–1996, the longest NT flow series), suggesting an average response time lag between numbers and flow of about 3–5 years. Figure 4.7d (cross-correlation correlelogram) highlights the coupled 20–25-year period trends between magpie geese numbers and flow.

Water extraction, wetland weeds and magpie goose nesting ecology on the Daly floodplain

Water extraction

O’Grady et al (2002a) reviewed groundwater and surface water extraction in the Daly River catchment based on estimates of usage for the Katherine Water District. Their assessment area covered parts of the Oolloo and Tindall limestones, and water extraction was estimated at approximately 16 000 ML/year (44 ML/d) or 0.2% of the mean annual surface flow at Mt Nancar of 7.1 x 10^6 ML. The National Land and Water Audit (NLWA 2002) predicted water use in 2020 and 2050 at 80 000 ML/year (220 ML/d), and 120 000 ML/year (330 ML/d), respectively (as referenced by O’Grady et al 2002a). No figures existed at the time for the amount of water used in the Katherine-King River area, which is sourced from the aquifer in the Tindall Limestone. However, O’Grady et al (2002a) estimated current groundwater usage sourced from the Tindall Limestone (either from bores or river baseflow) at approximately 9000 ML/year (25 ML/d). Licensed surface water allocations for Katherine in the Daly River catchment at the time was 7569 ML/y. For the rest of the Daly River catchment annual allocations totalled 1180 ML, and approximately 1000 ML/y was estimated for unlicensed water usage by riparian vegetation and groundwater. Only two small weirs were used for water supply in the Daly River catchment at 1500 ML combined.



Figures 4.7 a–d. (a) Trend in magpie goose numbers on NT floodplains (1958–1996, Tullock & McKean 1983, Bayliss & Yeomans 1990b, PWCNT 2003) showing two approximate 20 y periods. (b) Periodogram of spectral density of NT magpie goose numbers vs. period (y) using Fourier analysis, showing a mean 21 y return period. (c) Cusum values for magpie goose numbers ($\log_{10}$) in the NT (1958–1996) and Katherine River flow ($\log_{10}$ ML) showing an average response time lag of 3–5 y. (d) Cross correlation (Correlogram) between cusum values for goose numbers ($\log_{10}$) and flow ($\log_{10}$ ML) showing coupled 20-year periods. Dashed lines are 95% confidence intervals and standard errors (3rd column) are white noise estimates.

Future water allocation in the Daly Region has received considerable government and community attention over the last several years despite the low current and projected water usage rates cited above. Early community stakeholder consultations (DRCRG 2004) identified water extraction as a potential key threat to in-stream and floodplain environmental flows and, hence, the ‘condition’ of their associated habitats and biotic communities. For example, Georges et al (2002, 2003) modelled the negative impact that dry season flow extractions in the Daly River may have on populations of the iconic pig-nose turtle (*Carettochelys insculpta*), whereby the effects of flow reduction were mediated through changes in ambient water temperatures through to temperature-dependent sex ratios. Erskine et al (2003) made detailed and comprehensive recommendations on environmental water allocation, which were revised at a Daly Region water allocation workshop held in May 2004. Erskine et al (2004) then proposed changes to the original recommendations based on additional data, information and consultation, summarised below:

1 *Floods and Wet Season Environmental Water Requirements:* the rising limb and flood peaks should be protected because they cue important biotic responses and because they serve important geo-ecological functions, such as channel maintenance, reworking of sand bars for pig-nosed turtle nesting sites and lateral connection of floodplains.

2 *Minimum Streamflows and Dry Season Environmental Water Requirements:* minimum streamflows should be maintained to protect *Vallisneria nana*, *Spirogyra* spp, pig-nose turtles and other aquatic flora and fauna, and to ensure that the water requirements of riparian vegetation can be supplied at times of extreme water stress.

3 *Maintenance of Groundwater Discharge to the Daly River and Dry Season Environmental Water Requirements:* groundwater levels and spring inflows to the Daly River during the dry season should be maintained to ensure that current base flows persist.

4 *Water Quality Environmental Water Requirements:* existing groundwater and surface water quality in the Daly Basin should be maintained to protect aquatic ecosystem structure and function.

5 *Recognition of, and allocation for, cultural flows:* protection of both environmental and cultural values (see Jackson et al 2005) as recommended by Jackson (2004).

With respect to (2) above, Erskine et al (2004) further recommended that the following minimum streamflows should be adopted at the relevant locations:

- Dorisvale Crossing – 6.2 cumecs (m3/s) or 536 ML/d
- Oolloo Crossing – 12 cumecs (m3/s) or 1,037 ML/d
- Mt Nancar – 12 cumecs (m3/s) or 1,037 ML/d

In addition to the above minimum management thresholds, Erskine et al (2004) identified the following hydrological event thresholds (see recommendations 1.2 & 1.6, & Section 4.1.1 above): dry season river stage is about 1m at most gauging stations on the Daly River; a flood is defined as a flood hydrograph with a peak stage greater than a gauge height of 7m at most gauging stations; and a floodplain flood is an event with a peak stage higher than 14m at Mt Nancar gauging station (ie accommodating the difference between a channel flood only & a channel flood plus floodplain flood). Erskine et al (2004) further recommended that at discharges greater than the above minimum thresholds, but less than the flood event and baseflow event thresholds specified above, at least 80% of the streamflow should be protected for the maintenance of streamflow, water quality, flow hydraulics, aquatic habitats and flora and fauna. Additionally, they recommended that at discharges less than the above minimum thresholds, at least 92% of the streamflow at these locations must be protected for the maintenance of critical aquatic habitats and their associated flora and fauna.

Needless to say, the extraction of Daly River flow will entail a complex mix of cultural, political, socio-economic, regulatory and biophysical issues, as exemplified by the fact that the flow extraction recommendations summarised above have yet to be adopted. Hence, the approach adopted here for simulating water extraction scenarios does not follow the complex extraction rules underpinning the recommendations of Erskine et al (2004). Flow extraction rules are here simplified in order to examine, in principle, potential impacts on the chosen ecological assessment endpoints for magpie geese and barramundi. Given that there is strong correlation (and most likely interaction) between wet season flow and subsequent dry season flow, and total annual flow, we assume that our results are generally relevant regardless of the specifics of seasonal extraction rules. Hence, the following two water extraction rules are used:

1 Only surface flow extraction is simulated although there is strong connectivity between surface water and groundwater flows; and

2 Only wet season flow extraction is simulated, irrespective of whether or not flow is rising, at the flood peak or recessional.

Wet season flow at Mt Nancar gauging station (Figure 4.14) is arbitrarily defined as flow (ML) that occurs between October and April. Hence, the 20% cap on environmental flow extraction during the recessional phase of a flood is here applied to all of wet season flow and comprises one of the scenarios examined in the BN. A 'No' flow extraction is simulated also to provide a benchmark for each ecological assessment endpoint. The BN has options for flow extractions greater than 20%, representing worse case but familiar scenarios that typify many rivers in Australia.

Wetland weeds

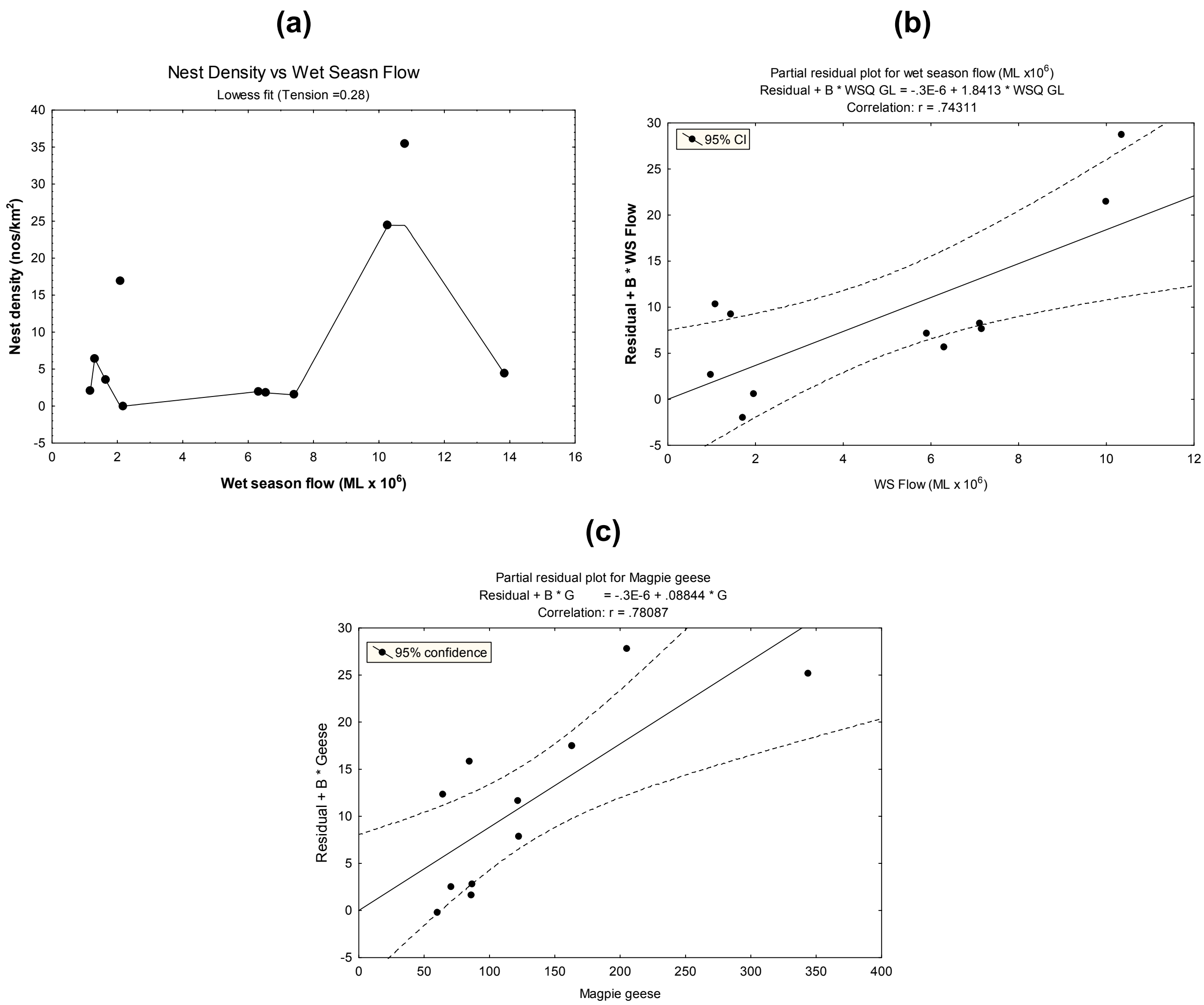
In contrast to the overwhelming ecological pressures due to flow extraction, drainage and habitat alteration experienced by wetlands and waterways in south-eastern Australia, wetland weeds have been identified as possibly the key threat to our relatively 'pristine' northern aquatic ecosystems (Finlayson et al 1988, Douglas et al 1998, Douglas et al 2001). The three most important wetland weed species in the tropics are currently *Mimosa pigra* (mimosa), *Salvinia molesta* (salvinia) and *Urochloa mutica* (para grass), although other aquatic weed species are rapidly emerging as significant threats. Mimosa and para grass were chosen for risk analysis because they occur on the Daly River floodplain and they have the ability to rapidly colonise most wetland habitats. Additionally, they form dense monocultures and so have maximum impact on native plant biodiversity (see Lonsdale et al 1985, 1988, 1995; Cook & Setterfield 1996; Miller 1983; Walden & Bayliss 2003; Walden et al 2004 for mimosa impacts; & Douglas & O'Connor 2003, 2004; & Ferdinands 2006 for para grass impacts). Ferdinands et al (2001) demonstrated that para grass is a major risk to the biodiversity of the Mary River floodplains, and Bayliss et al (2006) showed also that it is currently the major ecological risk on Magela floodplain, Kakadu National Park, because of its extent (11% of the floodplain at 100% cover, 35% extent), effect (a monoculture that displaces native vegetation) and relatively rapid spread rate (14% p.a. on average). Bayliss et al (2006) highlighted also that the potential spread rate and impacts of mimosa, which is well documented on the adjacent Oenpelli floodplain (Cook 1996, Cook & Setterfield 1996; Lonsdale 1993), is currently controlled on Kakadu through an annual 'search and destroy' investment of about $0.5 million. In addition to biodiversity impacts on plants, the displacement of wild rice and water chestnut (*E. dulcis*) dominant communities will have major impacts on many native wildlife species that use floodplains, such as the iconic magpie goose (Bayliss & Yeomans 1990b, Whitehead & Saalfeld 2000) because these plants play critical roles in food webs by providing seasonally abundant, high-energy food sources.

Magpie geese nesting ecology on the Daly floodplain

The Daly River floodplain contains key wet season nesting habitat for magpie geese in the NT (1983-1986: 21%-36% of the NT population) and at times supported up to 13% of NT geese in the dry season (Bayliss & Yeomans 1990b, PWCNT 2003, Chatto 2006). Hence, their nest density and population size on the floodplain, as ascertained by systematic aerial surveys between 1984 and 2000, are now examined in detail in relation to river flow. This analytical step is a necessary prerequisite to a comprehensive risk assessment of the impacts of potential future water extractions on top of existing impacts from wetland weeds. River flow is strongly correlated to river stage height (Section 4.1.1) and, hence, the extent and period of inundation of floodplain floods and associated availability of seasonal nesting habitat. We assume, therefore, that river flow is a reliable index of the quantity and quality of available magpie goose nesting habitat.

As discussed above, whilst the timing of onset of wet season rainfall may trigger nesting in magpie geese and so ultimately influence annual nest density and recruitment success, the amount of rainfall throughout the remainder of the wet season is also critical in that it needs to be sufficient in order to ensure completion of the nesting cycle, and to produce adequate early dry season food (eg wild rice) for emergent goslings, and late dry season food such as *E. dulcis* bulbs, critical for the survival of adults and yearlings before the next wet season rains. The amount of rainfall is critical also in that there can be too little (eg a poor wet season with no floodplain flooding) or too much (eg extensive floodplain floods causing nests & goslings to drown). Hence, the relationship between nest density and flow may exhibit 'threshold' effects, and observed data for the Daly River floodplain supports this suggestion (Figure 4.8a). A Lowess (StatSoft 2001) curve fitted to the trend indicates that, with the exception of one outlier, nest density falls between 0–6.0/km^2 at wet season flows less than average (6.9 x 10^6 ML, or a mean monthly stage height at Mt Nancar of 4.8 m). However, soon after average flow is passed (& hence the mean monthly water level that triggers the occurrence of floodplain floods), a threshold is reached whereby nest density peaks at 25–35/km^2, encompassing a broad range of high flows (9–11 x 10^6 ML). Although data are few (one point), a second threshold may exist whereby at flows greater than 11.0 x 10^6 ML (mean monthly stage height at Mt Nancar, 5.6–5.9 m) nest density declines, possibly due to nest drowning (pers obs).

Although the above double threshold nest-flow relationship is intuitive, a simple multiple linear regression model (Table 4.4) was first fitted to the data because they were assumed too few to provide a reliable fit to more complex nonlinear multivariate models. The model hypothesis is that magpie geese nest density is both a function of wet season flow (indexing the quantity and quality of nesting habitat on the floodplain) and the availability of adult geese to breed. Overall explained variance was 67% (untransformed data), which we consider an acceptable level of model uncertainty for simulation. However, the partial regression residuals of nest density on flow (Figure 4.8b), and nest density on goose density (Figure 4.8c), suggest that both variables may in fact exhibit complex nonlinear relationships. Hence, second order polynomials of flow and goose density were added to the previous linear model and both were highly significant additions (Table 4.5). The overall regression explained 98% of variance in the data and is highly significant after protection for Type I error rate using a Bonferonni correction (Wilkinson 1979).



Figures 4.8 a–c. (a) Threshold relationship in magpie goose nest density (nos.km^{-2}) and flow at Mt Nancar gauging station on the Daly River floodplain (1984–2000; Bayliss & Yeomans 1990b, K. Saalfeld P&WC NT unpubl data). A Lowess curve (Tension = 0.28) was fitted to the trend. Partial residual plots for Y-partial nest density (numbers/km^2) vs. (b) X-partial wet season flow (ML) and (c) X-partial goose density (numbers/km^2). The scatter plots indicate that the relationship between both variables and nest density may be nonlinear.

Table 4.4 Multiple regression summary of magpie geese nest density (N, numbers/km^2) vs. wet season flow (WSQ, ML x 10^6) and density of available breeders (G, numbers / km^2). Daly River floodplain (1984 – 2000), n = 11 (some years not surveyed), see data summary in Appendix 8.2.1. Data are untransformed.

R= 0.8572, adjusted R^2 = 67%, n= 11, P< 0.005, SE regression = 6.67

Variable	Beta	SE Beta	B	SE B	P
Intercept			-11.45	4.77	0.043
WSQ (ML x 10^^{6})	0.572	0.182	1.84	0.59	0.012
G	0.644	0.182	0.09	0.02	0.008

Table 4.5 Multiple regression summary of magpie nest geese density (N, numbers/km^2) vs. wet season flow (WSQ, ML x 10^6) and current breeding density (G, numbers/km^2) using quadratic polynomials to capture nonlinear effects. Daly River floodplain (1984 – 2000), n = 10 (some years not surveyed), see data summary in Appendix 8.2.1.

R= 0.8572, adjusted R^2 = 98%, n= 11, P< 0.005 (Bonferroni adjustment), SE regression = 1.76

Variable	Beta	SE Beta	B	SE B	P
Intercept			-7.24	2.23	0.023
WSQ	-2.028	0.265	-5×10^{-6}	1×10^{-6}	<0.001
WSQ^2	2.737	0.273	8×10^{-13}	8×10^{-13}	<0.001
G	0.409	0.053	0.06	0.0073	<0.001
G^2	-0.218	0.055	-4.6×10^{-5}	1.1×10^{-5}	0.011

4.2.2 Conceptual model for floodplain health

A conceptual model was constructed (Figure 4.9) to guide assessment of the 'health' of the Daly River floodplain under different wet season flow extraction scenarios (eg 0% & 20%) and simultaneously impacted by the wetland weeds mimosa and para grass. The following three ecological endpoints were used to assess potential ecological impacts: (i) the health of magpie goose nesting success in the wet season in relation to potential flow extraction and extent of weeds; and (ii) the health of magpie geese dry season refuge habitat and (iii) plant biodiversity in relation to weed extent only. A stochastic ecological process model (goose nests vs. flow & density of breeding adults) and a BN were then developed from the conceptual model (see Sections 4.2.3 & 4.2.7 below, respectively). The BN incorporated results from a spatially explicit weeds risk assessment (Section 4.2.5) and was then extended to include modules and pathways to: simulate the annual spread of both weeds and, hence, increasing ecological risk over time; and to estimate control costs of both weeds to a specified target objective so that improvements in floodplain health at different levels of investment can be assessed. Control cost scenarios can be simulated in any year since the 2003 baseline weed survey.

4.2.3 Water extraction simulation, model uncertainties and sensitivity analyses

The stochastic process model used to predict the impact of simulated wet season flow extractions (0–100%) on magpie goose nest density (see multiple regression equation in Table 4.4) is conceptually illustrated in Figure 4.10 and shows all model uncertainties. The frequency distribution of observed flow data (ML) during nest surveys used in the regression model is best described by an exponential probability density function (pdf) (Figure 4.11a & b) and, similarly, for goose density (numbers/km^2, Figure 4.11c & d). Mean values were derived by Monte Carlo (MC) simulation (10 000 iterations) using @Risk™ software (Pallisade 2002b) and incorporated parameter variability and overall model uncertainty as outlined in Figure 4.10.

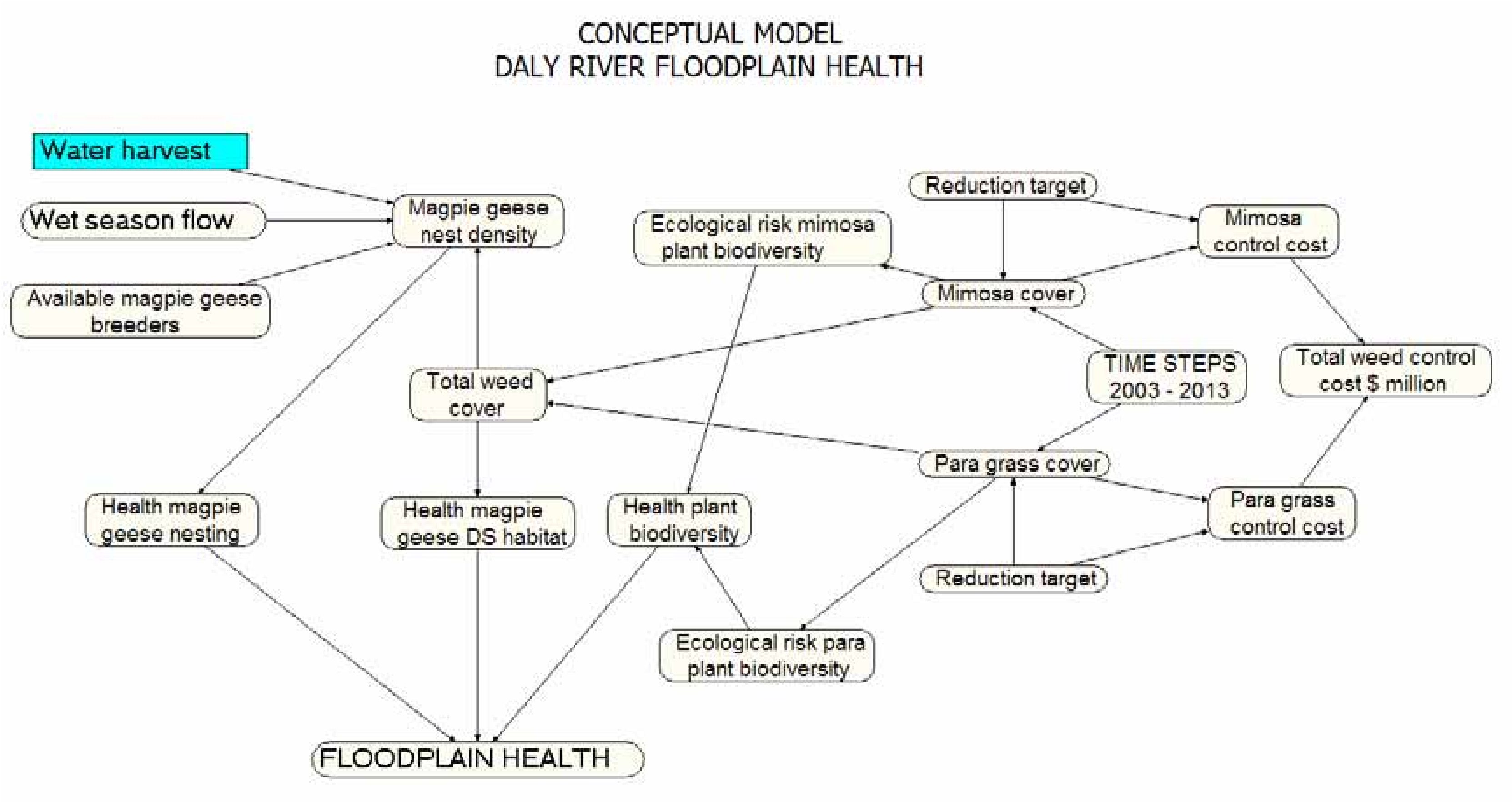


Figure 4.9 Conceptual model used to construct a Bayesian Network to assess the 'health' of the Daly River floodplain under different wet season flow extraction scenarios (eg 0%, 20%) and impacted by the wetland weeds mimosa and para grass. Three ecological endpoints were used to assess potential impacts: (i) the health of magpie goose nest density in the wet season; (ii) the health of magpie goose dry season refuge habitat; and (iii) the health of floodplain plant biodiversity. The Floodplain BN includes also pathways to estimate cost of control of mimosa and para grass in order to assess improvements in floodplain health with different levels of investment. Control cost scenarios can be simulated in any year since the baseline weed survey in 2003 (= 16% cover of floodplain), and encompass predicted annual spread rate (ha).

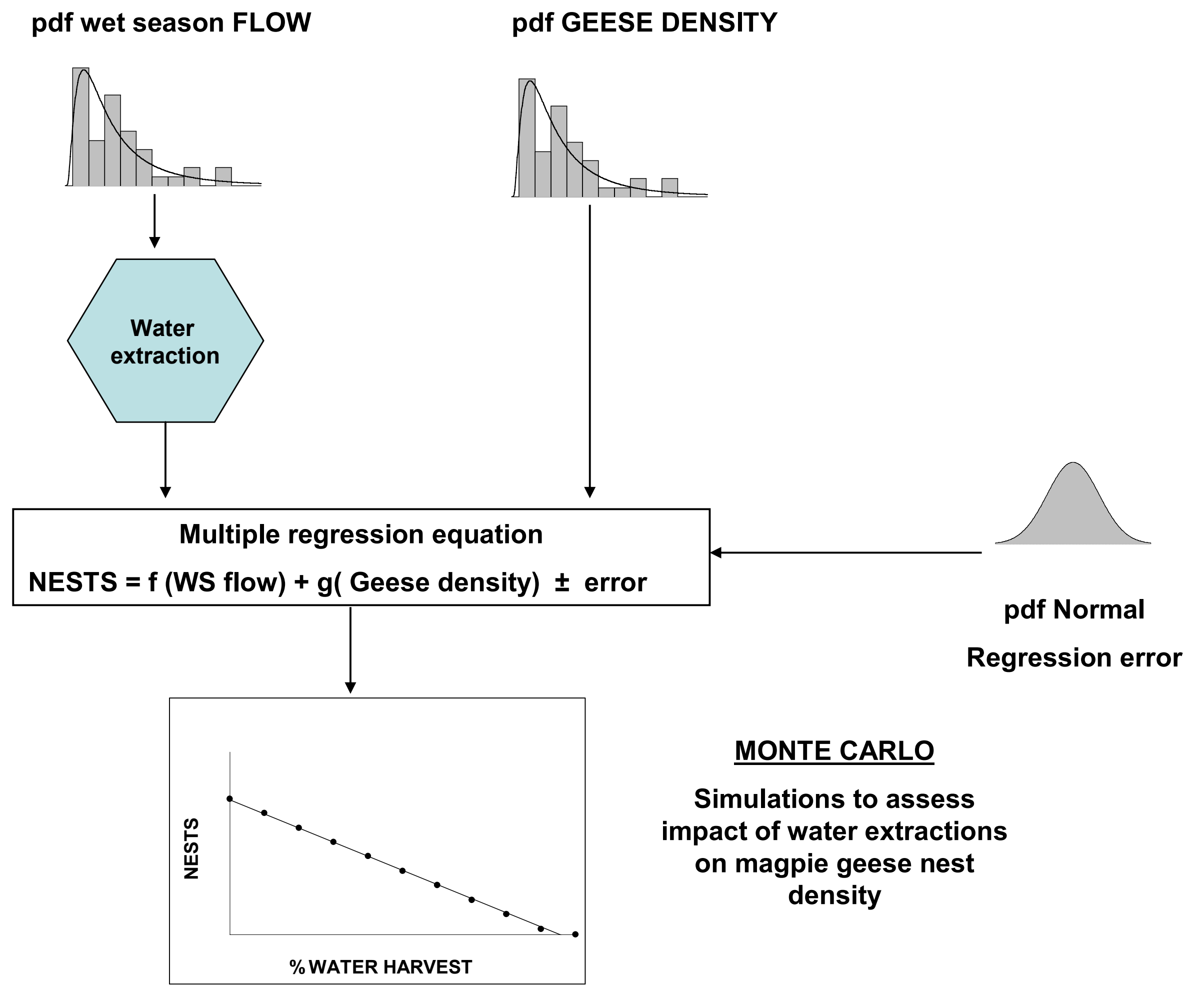
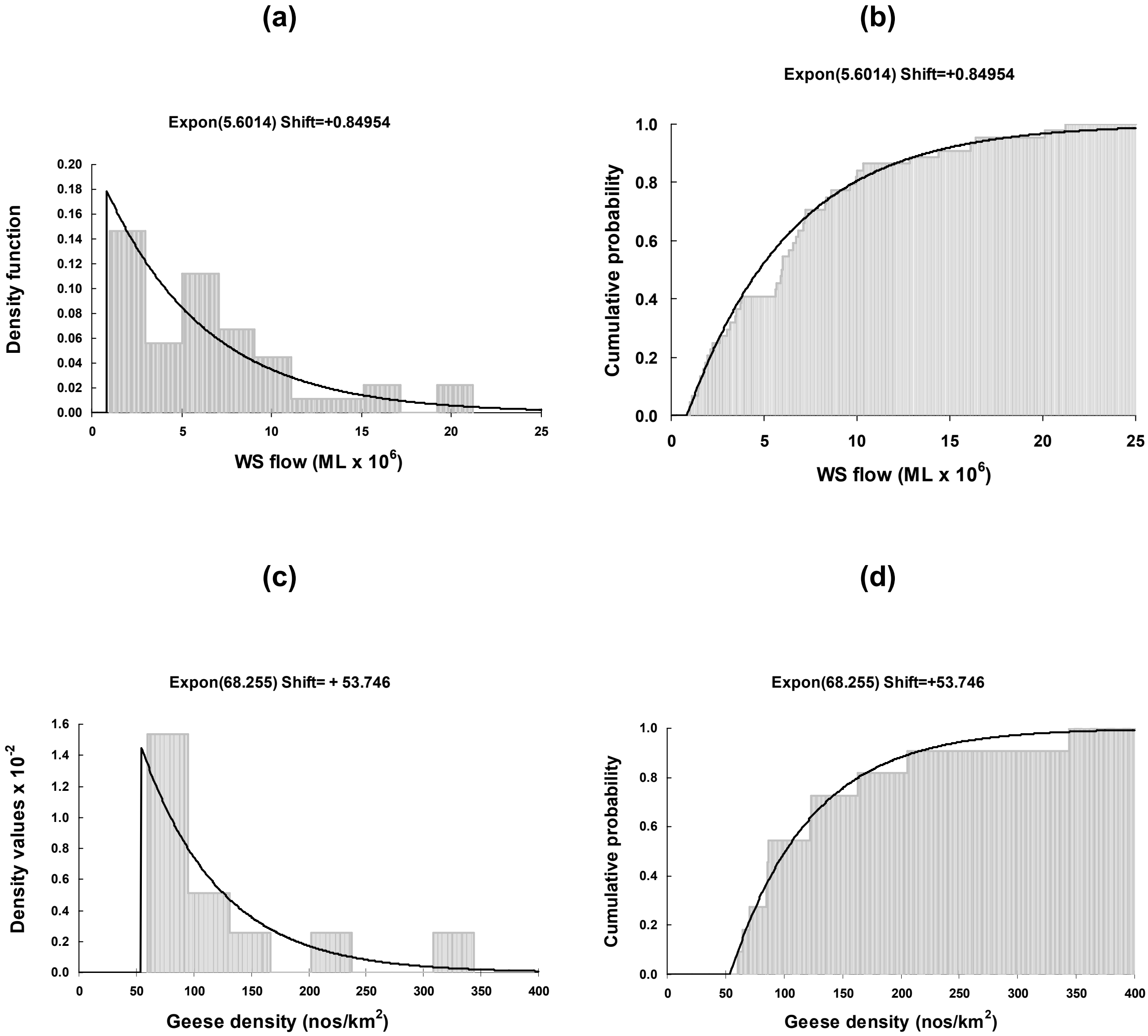


Figure 4.10 Stochastic process sub-model of the simulated impact of wet season flow extraction (0-100%) on magpie goose nest density showing all model uncertainties. The statistical model is used to underpin the Bayesian Network for floodplain health using magpie goose nesting success as an ecological endpoint and nest density (numbers/km^2) as the measurement endpoint. The multiple regression equation was developed from NT Parks & Wildlife Commission aerial survey data to predict nest density based on available geese to breed and current flow.



Figures 4.11 a–d. Statistical distributions fitted to observed data used in the regression equation (& subsequent Bayesian Network) to predict nest density (numbers/km^2) as a function of available wet season flow and numbers of geese to breed (numbers/km^2). (a) Probability density function (pdf, Exponential) and (b) cumulative probability curve of wet season flow (ML). (c) Probability density function (pdf, Exponential) and (d) cumulative probability curve of available geese to breed (numbers/km^2).

Whilst MC Simulations predict that mean nest density will decrease as the proportion of wet season flow extraction is increased (Figure 4.12a), the prediction appears highly uncertain (Figure 4.12b). Nevertheless, additional model outputs indicate that, despite model uncertainty, results are coherent with observed data. For example, the pdf of observed nest density is best described by a Lognormal distribution (Figure 4.13a & b) and is similar to the distribution of predicted nests including model error (Figure 4.13c). Additionally, a sensitivity analysis (regression method) indicates that model outputs were more influenced by wet season flow inputs than either model error or goose density (Figure 4.13d). Simulation results predict that, in general, magpie goose nest density will decrease in proportion to the amount of wet season flow extracted; for example, a 20% reduction in flow will lead to a 20% reduction in nests.

4.2.4 Spatially explicit ecological risk assessment of weeds

The extent and location of the Daly River freshwater floodplain in the Daly River catchment that encompasses both wet season magpie goose nesting colonies and their dry season refuge habitats is illustrated in Figure 4.14. The location of the two stream gauging stations that provided flow data for all subsequent analyses is shown also (G8140040 & G814001, see Moliere 2008).

Distribution and abundance of mimosa and para grass on the Daly River floodplain

In 2003 the NT Weeds Branch undertook a systematic helicopter survey of weeds on the Daly River floodplain and used a similar methodology as that used by Wilson et al (1991) to survey the distribution and abundance of native floodplain vegetation (S Wingrave pers. comm.). The percentage cover of mimosa was recorded on transects spaced about 2.7 km apart, and the presence/absence of para grass in four 25% cover abundance classes was recorded also. In a GIS the Daly River floodplain was divided into 250 m x 250 m cells (6.25 ha) and all continuously recorded vector point GPS data for mimosa and para grass were averaged for each cell. This procedure facilitated mapping of cover abundances and a spatially-based risk assessment of exposure and effects after 13 years (1990 to 2003). The distribution and cover abundance of mimosa and para grass on the Daly River floodplain in 2003 are illustrated in Figures 4.15 a & b, respectively.

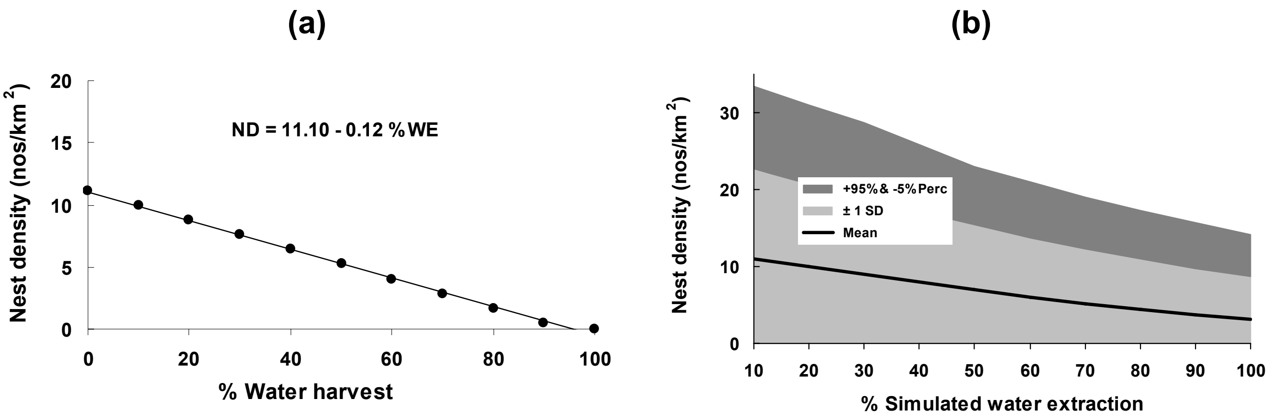


Figure 4.12 a & b Simulated declining mean trend in (a) nest density as a function of increasing wet season flow extraction using the simple regression equation summarised in Table 5.3. Mean values were determined using Monte Carlo simulation (10,000 iterations) using @Risk software (Palisade 2002) and incorporated model uncertainty as outlined in Figure 6. (b) As for (a) above but with uncertainty levels illustrated using one standard deviation (SD) about the mean trend and the + 95% and – 5% percentiles.

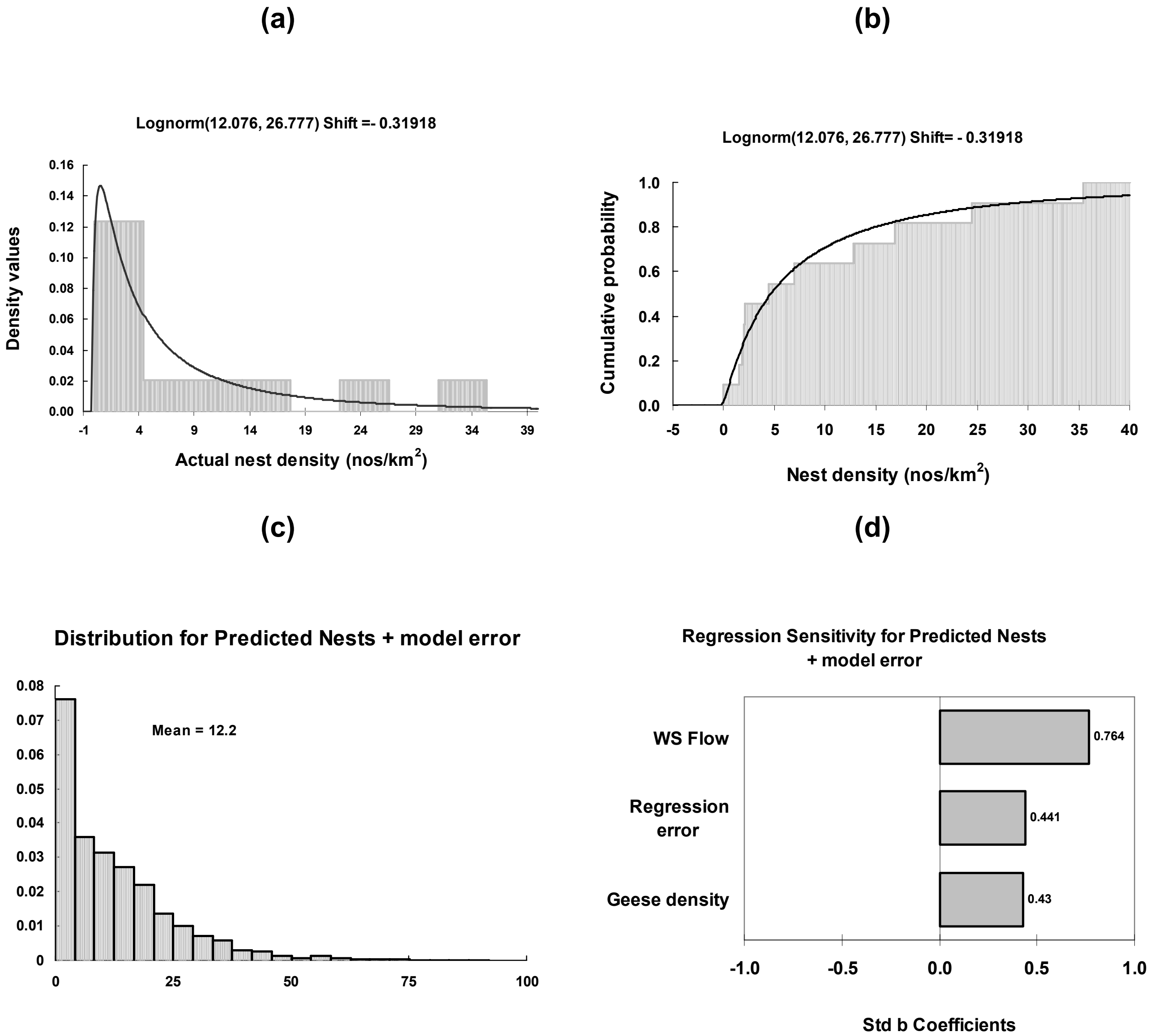


Figure 4.13 a–d Statistical distributions fitted to observed densities (numbers/km^2) of magpie goose nests and simulated data. Probability density function (pdf, Exponential) and (b) cumulative probability curve of observed nest density data (numbers/km^2). (c) Probability density function (pdf, Exponential) of simulated data, and (d) Tornado graph summarising sensitivity analyses of variable inputs into the regression equation predicting nest density on flow and available geese to breed, showing that flow contributed most to simulated model outputs, followed by regression error and the density of available geese to breed.

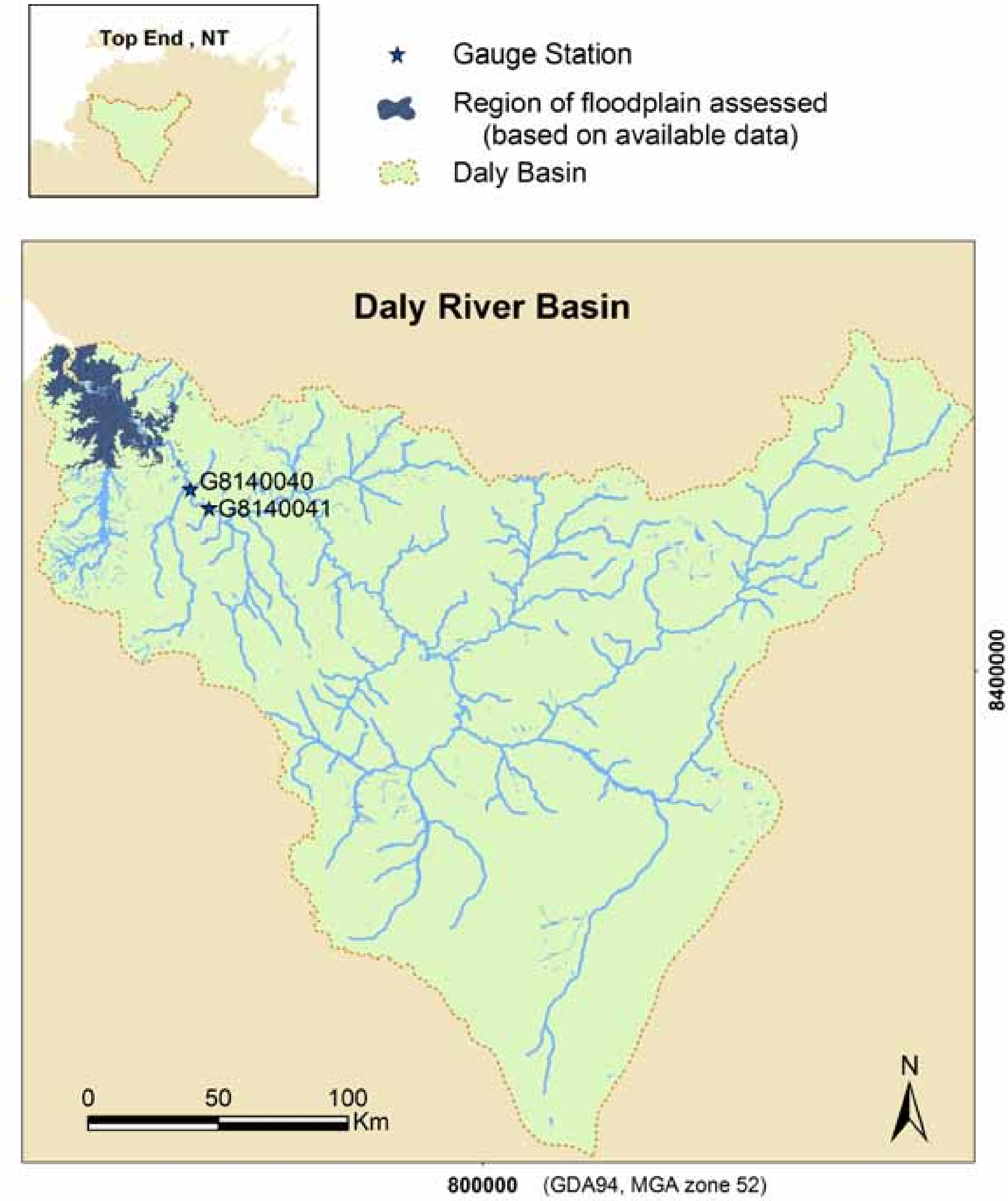


Figure 4.14 Map of Daly River catchment showing extent of floodplain (dark blue shading) used in the weeds risk assessment, and the location of the two gauging stations (G8140040 & G8140041) used in all analyses of flow above (see text).

Distribution and abundance of key wetland native plants on the Daly River floodplain

To facilitate assessment of likely effects of weed exposure on native floodplain vegetation *per se* and magpie goose habitats, the distribution and abundance of four plant groups important to their nesting and dry season survival were also mapped on the same spatial grid using data from Wilson et al (1991). These were: *Eleocharis sphacelata* and *Ischaemum australe* (nesting material); *Oryza* spp (food for adults & emergent goslings); and *E. dulcis* (bulbs provide preferred dry season food for adults and yearlings, see Bayliss & Yeomans 1990b), respectively (Figure 4.16a–d). As expected, all major plant groups are spatially correlated and so exhibit similar habitat requirements (Table 4.6). For example, *E. sphacelata* and *E. dulcis* sedges generally co-occur (R=0.4120, n=140, P<0.001), and wild rice is highly correlated to both sedges, particularly *E. sphacelata* (R=0.6311, n=140, P<0.001). Hence, to simplify examination of associations between nests and floodplain vegetation types, the cover abundances of wild rice and both *Eleocharis* sedges were combined (see Table 4.6).

Distribution and abundance of magpie goose nest colonies and dry season refuge sites

To examine the effects of weed exposure on magpie goose nesting success and dry season survival, their distribution and abundance across the floodplain are mapped for selected seasonal surveys using data from Bayliss & Yeomans (1990b) and unpublished NT Parks & Wildlife Commission data (K Saalfeld pers comm). All observed densities were corrected to estimates of absolute densities using the seasonal visibility correction factors derived by Bayliss and Yeomans (1990a). Figures 4.17 a–d show the distribution and density (numbers/km^2) of magpie goose nests on the Daly River floodplain in the late wet seasons of (a) 1984, (b) 1989, (c) 1993 and (d) 2000, respectively.

Multiple regression analysis was used to examine relationships between nest density estimated in 1993 and floodplain vegetation type estimated in 1990. Vegetation type is used here to index suitable nesting habitat. Results (Table 4.7) show that geese prefer to nest in and around vegetation types dominated by *E. sphacelata* and wild rice (*Oryza* spp). Although explained variance is low (R^2=10%), and the analysis did not account for scale effects or spatial autocorrelations, the results support the findings of many previous studies (eg Frith & Davis 1961, Bayliss & Yeomans 1990b, Whitehead & Saalfeld 2000) with respect to nesting habitat preference.

The distribution and abundance (numbers/km^2) of magpie geese on the Daly River floodplain in the dry seasons of 1984 (data from Bayliss & Yeomans 1990b) and 1996 (data from NT P&WC, K Saalfeld pers comm) are shown in Figure 4.18 a & b, respectively.

Dry season goose density ($\log_{10}$ numbers/km^2) on the Daly River floodplain was correlated to the cover abundances of *Hymenachne* spp and *Eleocharis sphacelata* (Table 4.8), species that prefer deep water (~1–2 m, Finlayson 1993). Whilst the covers of both plants were spatially intercorrelated (R=0.2042, n=135, P<0.01), they were entered as relatively independent variables in the multiple regression equation. As with the nesting analysis above, explained variance is low (R2=6%), and the analysis did not account for scale effects or spatial autocorrelations. Nevertheless, these results also support the findings of previous studies (eg Bayliss & Yeomans 1990b).

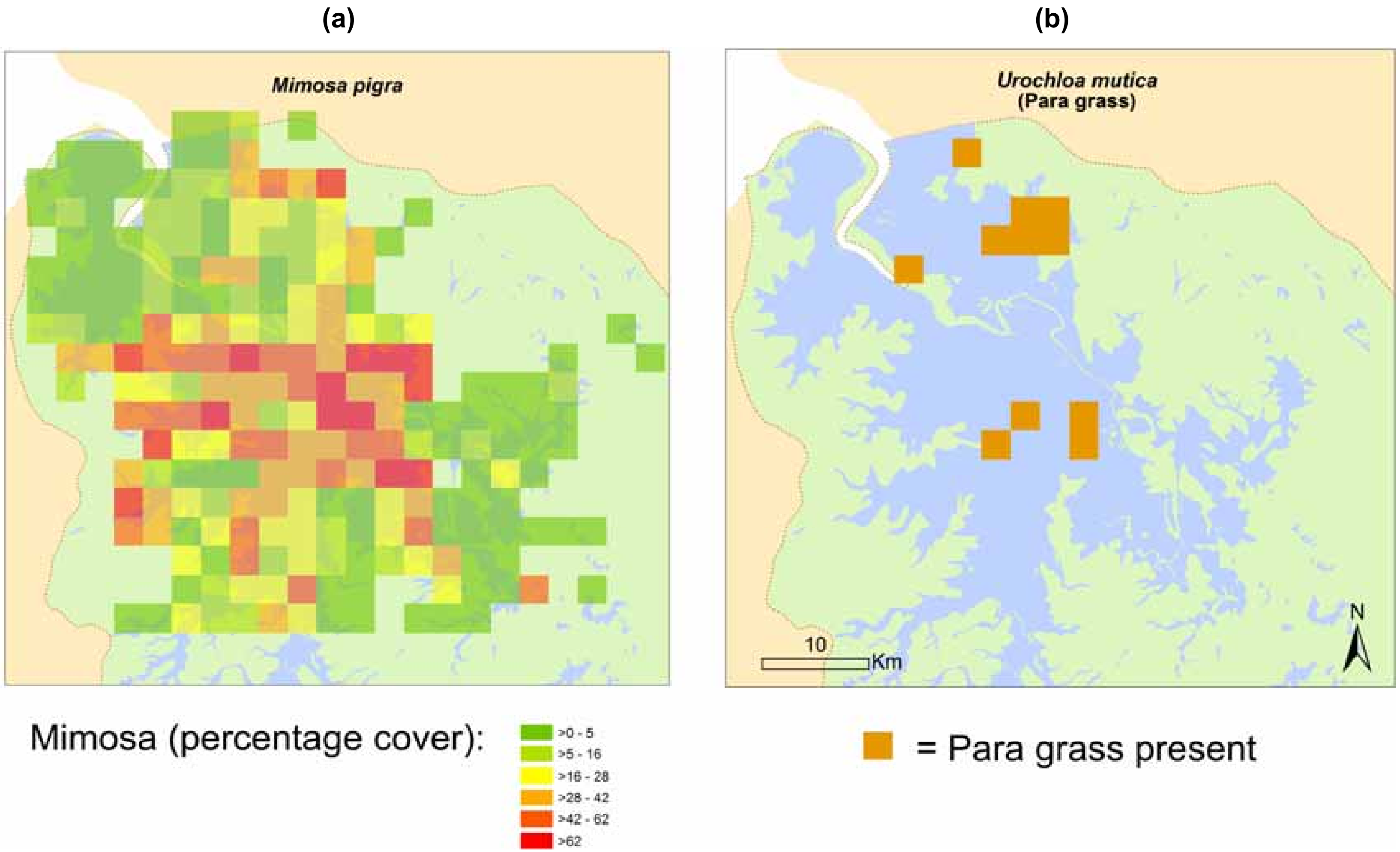


Figure 4.15 a & b Distribution and abundance of (a) mimosa (percentage cover) and (b) para grass (presence/absence) on the Daly River floodplain in 2003 (NT DPIFM, S Wingrave unpubl survey data). Cell sizes are 250 m x 250 m (6.25 ha).

Table 4.6 Correlation matrix between vegetation plant groups on the Daly River floodplain (% cover arcsine transformation, wet season 1990, Wilson et al1991), magpie geese nest density (log_{10} numbers/km^2) estimated in 1993 and magpie geese density (log_{10} numbers/km^2) estimated in the dry season of 1996 (K Saalfeld, pers comm, Parks & Wildlife Commission unpubl data), and the wetland weeds mimosa and para grass estimated in 2003 (S Wingrave pers comm, DPIFM unpubl Data). Codes for plant groups are outlined below. Bold text are significant at P<0.05. See data in Appendix 8.2.2 and 8.2.3.

	Mim_AS	**Para_AS**	**L_{10} Nest93**	**L_{10} DG96**	**Hym_AS**	**Isch_AS**	**Rice_AS**	**EleoS_AS**	**EleoD_AS**	**TNV_rs_AS**	**RESED_AS**
Mim_AS	1.00	0.11	-0.15	0.12	0.09	**0.20**	0.03	0.06	0.03	**0.30**	0.06
Para_AS	0.11	1.00	0.00	-0.03	-0.07	-0.02	0.06	0.00	0.00	0.03	0.07
L_{10} Nest_93	-0.15	0.00	1.00	0.04	**0.19**	-0.11	**0.31**	**0.29**	0.16	**0.25**	**0.32**
L_{10} DG_96	0.12	-0.03	0.04	1.00	**0.21**	-0.11	0.11	**0.19**	0.12	0.12	0.12
Hym_AS	0.09	-0.07	**0.19**	**0.21**	1.00	**-0.28**	0.10	**0.21**	-0.06	0.15	**0.23**
Isch_AS	**0.20**	-0.02	-0.11	-0.11	-0.28	1.00	**-0.55**	**-0.42**	**-0.46**	**0.40**	**-0.55**
Rice_AS	0.03	0.06	**0.31**	0.11	0.10	**-0.55**	1.00	**0.62**	**0.83**	**0.46**	**0.94**
EleoS_AS	0.06	0.00	**0.29**	**0.19**	**0.21**	**-0.42**	**0.62**	1.00	**0.40**	**0.34**	**0.69**
EleoD_AS	0.03	0.00	0.16	0.12	-0.06	**-0.46**	**0.83**	**0.40**	1.00	**0.38**	**0.83**
TNV_rs_AS	**0.30**	0.03	**0.25**	0.12	0.15	**0.40**	**0.46**	**0.34**	**0.38**	**1.00**	**0.48**
RESED_AS	0.06	0.07	**0.32**	0.12	**0.23**	**-0.55**	**0.94**	**0.69**	**0.83**	**0.48**	1.00

Plant group	**Code**	**Plant group**	**Code**	**Plant group**	**Code**
Mimosa pigra	Mim_AS	*Hymenachne* spp	Hym_AS	*Eleocharis dulcis*	EleoD_AS
Para grass	Para_AS	*Ischaemum*	Isch_AS	Total native vegetation (re-scaled to 100% cover)	TNV_rs_AS
Nest density	L_{10} Nes_t93	Wild rice (*Oryza* spp)	Rice_AS	Wild rice + *Eleocharis* sedges	RESED_AS
Geese density	L_{10} DG_96	*Eleocharis sphacelata*	EleoS_AS		

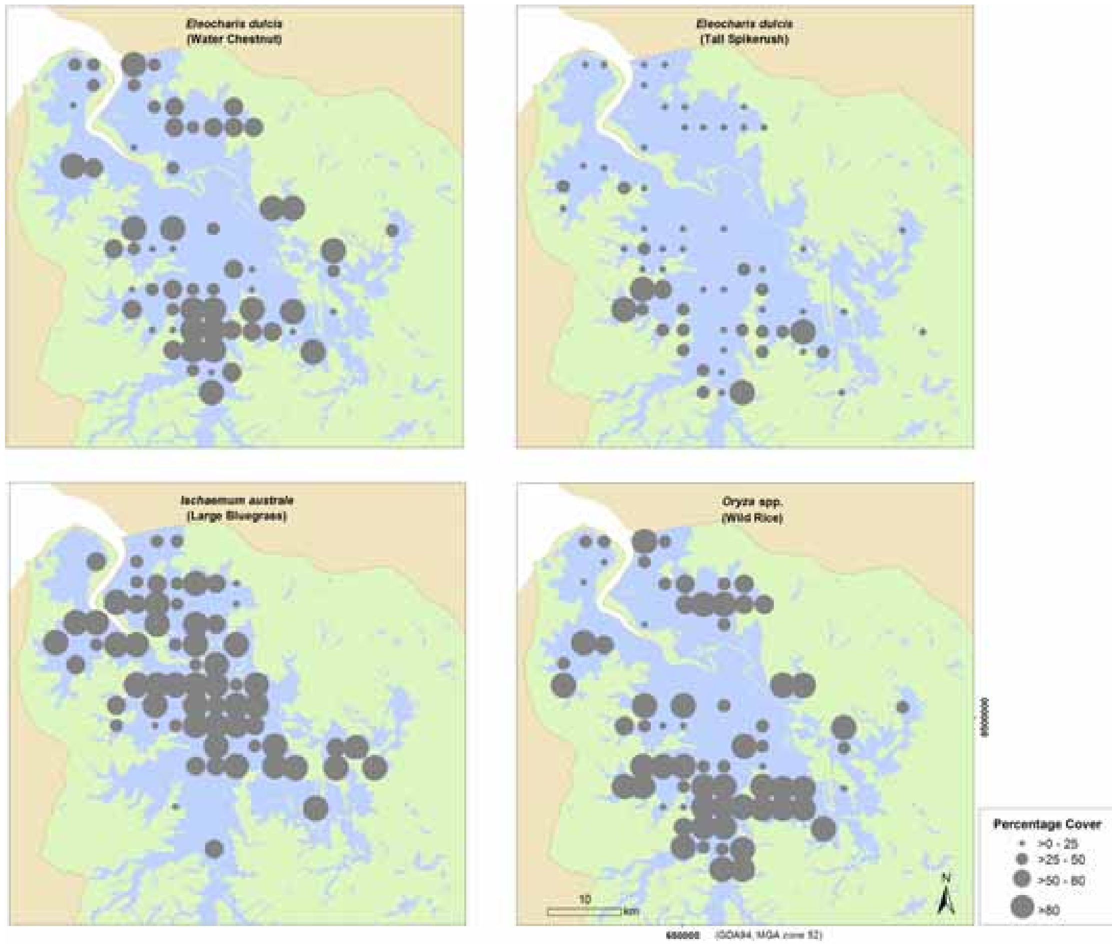


Figure 4.16 a–d (top left to bottom right). Distribution and abundance (percentage cover) of (a) *Eleocharis dulcis*, (b) *Eleocharis sphacelata*, (c) *Ischaemum australe* and (d) *Oryza* spp on the Daly River floodplain in 1990 (using data from Wilson et al 1991).

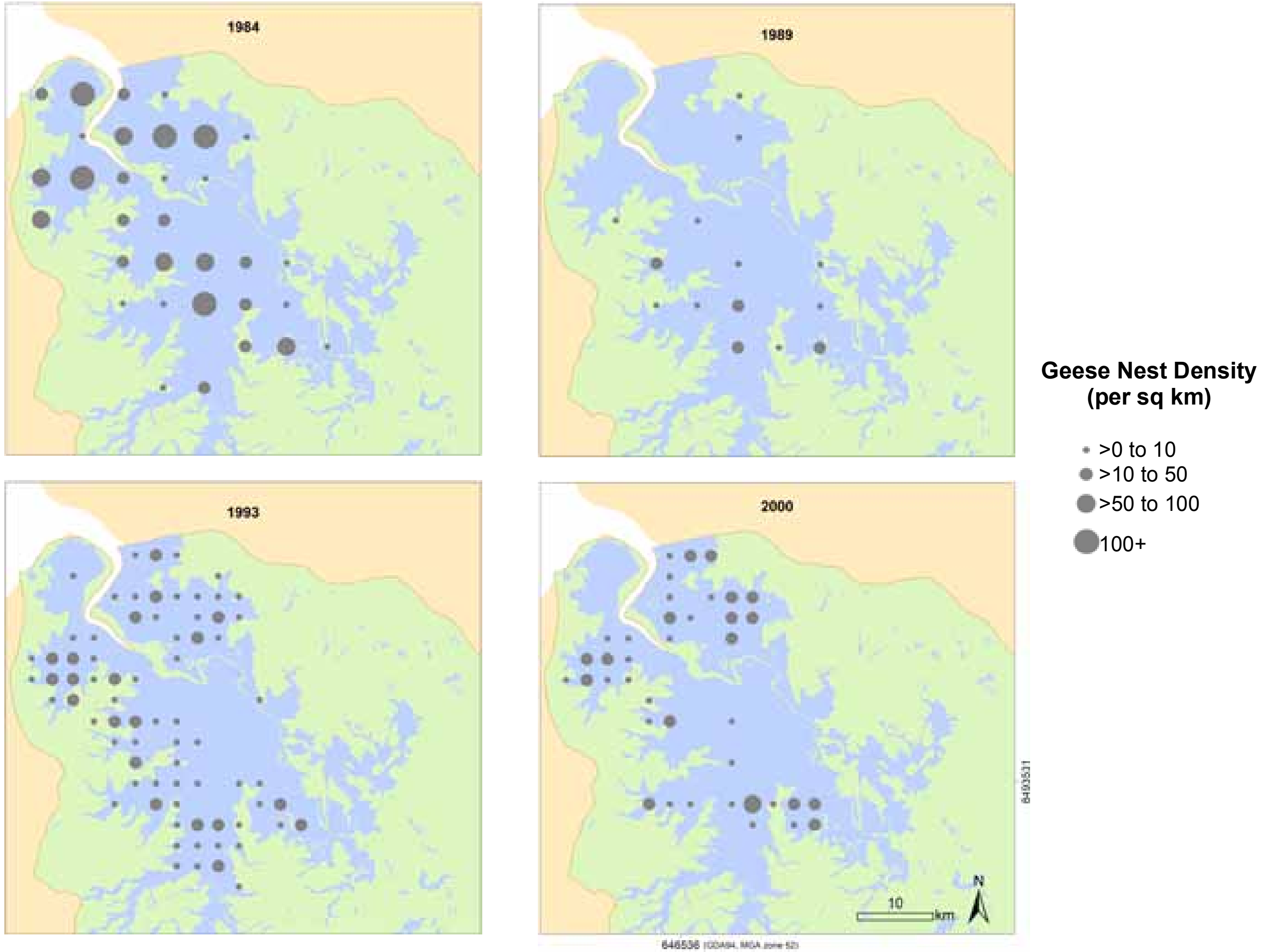


Figure 4.17 a–d (top left to bottom right). Distribution and abundance (numbers/km^2) of magpie goose nests on the Daly River floodplain in the late wet seasons of (a) 1984 (Bayliss & Yeomans 1990), (b) 1989, (c) 1993 and (d) 2000 (K Saalfeld pers comm, unpubl PWCNT data).

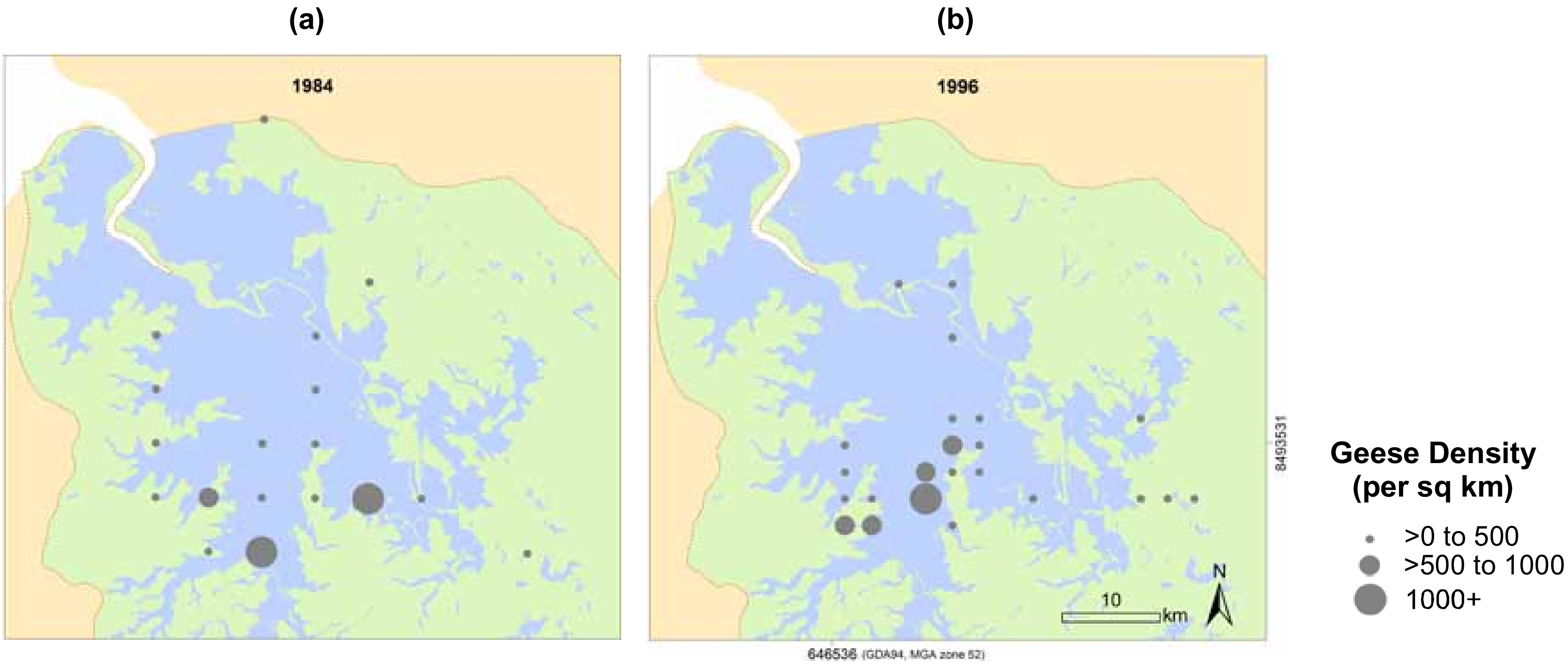


Figure 4.18 a & b Distribution and abundance (numbers/km^2) of magpie geese on the Daly River floodplain in the dry seasons of (a) 1984 (Bayliss & Yeomans 1990b) and (b) 1996 (K Saalfeld pers comm, unpubl PWCNT data).

Table 4.7 Multiple regression summary of magpie goose nest density (log_{10} N, numbers/km^2) estimated on the Daly River floodplain in 1993, and the percentage covers (arcsine) of wild rice and *Eleocharis* sedges combined (RES_AS) as estimated in the 1990 wet season (Wilson et al 1991; see data in Appendix 8.2.1 to 8.2.3).

R= 0.3269, adjusted R^2 = 10%, n= 135, P< 0.001, SE regression = 0.43

Variable	B	SE B	P
Intercept	0.241	0.049	<0.001
RES_AS	0.217	0.054	<0.001

Table 4.8 Multiple regression summary of magpie goose density (log_{10} G, numbers/km^2) estimated on the Daly River floodplain in the dry season of 1996, and the percentage covers (untransformed) of *Eleocharis sphacelata* sedge and *Hymenachne* spp estimated in the 1990 wet season (Wilson et al 1991). See data in Appendix 8.2.1 to 8.2.3.

R= 0.2541, adjusted R^2 = 6.1%, n= 133, P= 0.006, SE regression = 0.75

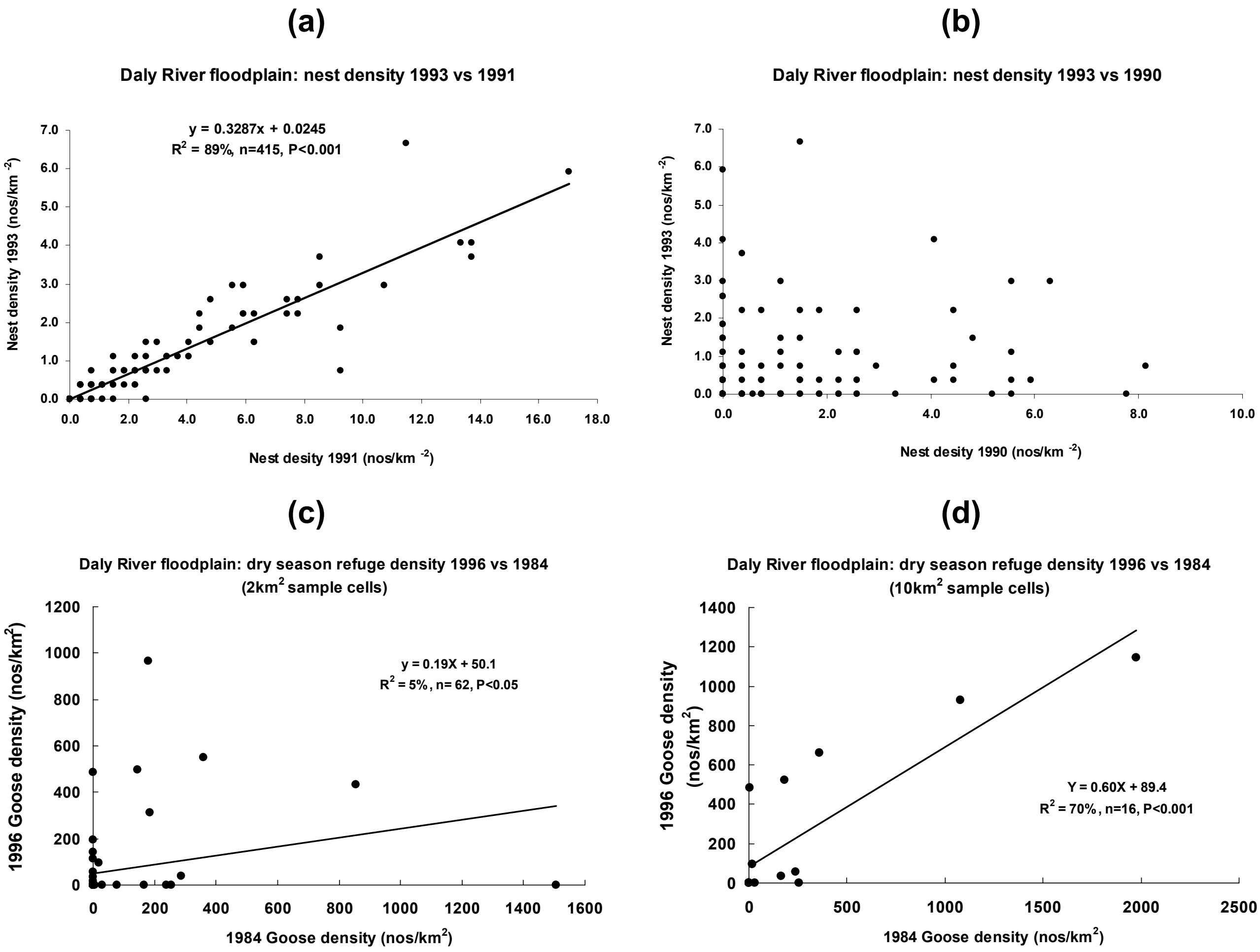
Variable	Beta	SE Beta	B	SE B	P
Intercept			0.168	0.077	0.029
HYM	0.186	0.086	0.0068	0.0028	0.032
ELEOS	0.170	0.086	0.0063	0.0031	0.049

The temporal stability of habitat use by geese for wet season nests and dry season refuges has management implications with respect to loss of favoured sites due to weed cover, and are therefore examined in detail here. A tight regression relationship was found between spatial nest densities in 1993 and 1991 using 2.7 km x 0.4 km sample cells (Figure 4.19a). In contrast, however, there was no correlation between 1993 and 1990 spatial nest densities (Figure 4.19b), possibly because the 1990 survey did not coincide with peak nesting. A weak regression relationship was found between dry season refuge use in 1994 and 1984 using 5.0 km x 0.4 km sample cells (2.0 km^2). However, the relationship tightens when data are scaled up to 25 km x 0.4 sample cells (10.0 km^2). Magpie geese, therefore, seem to return to generally the same areas to nest in colonies and to seek refuge in the dry season, suggesting that loss of preferred habitat across the floodplain from weed colonisation would have a major impact on their abilities to reproduce and survive.

4.2.5 Ecological risk of two floodplain weeds

The frequency distributions of observed mimosa and para grass cover abundance data per grid cell across the Daly River floodplain in 2003 are best described by Beta General probability density functions (pdfs; Figures 4.20 a & b and c & d, respectively). These statistical distributions were used to define state levels of threats in the initial Bayesian Network of floodplain health introduced in Section 4.2.7.

Two multivariate methods were used to examine relationships between mimosa and floodplain vegetation. The first used presence-absence data to fit a logistic regression model (Table 4.9) and the second used percentage cover abundance estimates to fit a multiple regression equation (Table 4.10).



Figures 4.19 a–d. Temporal stability of wet season nesting use and dry season refuge use by magpie geese on the Daly River floodplain. (a) Tight regression relationship between spatial nest densities in 1993 and 1991 using 2.7 km x 0.40 km sample cells. (b) No correlation between 1993 and 1990 spatial nest densities. (c) Weak regression relationship between dry season refuge use in 1996 and 1984 using 5 km x 0.4 km sample cells (2 km^2). (d) Strong relationship when data are scaled-up to 25 km x 0.4 sample cells (10 km^2). Data are from Parks & Wildlife Commission NT waterbird aerial surveys between 1983 and 2000 (Bayliss & Yeomans 1990b, K Saalfeld pers comm, unpubl PWCNT data).

Logistic regression results show that, with respect to the occurrence of mimosa, there is strong preference for the relatively deepwater habitats once dominated by *Hymenachne* (Table 4.9). No preference is apparent for habitats dominated by either *Eleocharis* sedge, whilst some preference is shown for habitats once dominated by *Ischameum australes* and wild rice. In contrast, multiple regression results (beta coefficients in Table 4.10) show that, with respect to both occurrence and abundance after a 13 year establishment period, mimosa had greater preference for habitats once dominated by *I. australe* grass, wild rice and *E.sphacelata* combined, with a lesser preference for habitats once occupied by *Hymenachne*. The contrast in results between the Logistic regression and multiple regression models most likely reflect greater resolution and, hence, accuracy in continuous data compared with presence/absence data. Regardless of statistical model, however, the habitat preferences of mimosa ascertained here reflect the degree to which different native vegetation types have been displaced after 13 years of colonisation and, hence, directly measures impact on plant biodiversity. There are too few observations of para grass occurrence in grid cells across the floodplain to undertake a similar multivariate analysis of habitat suitability.

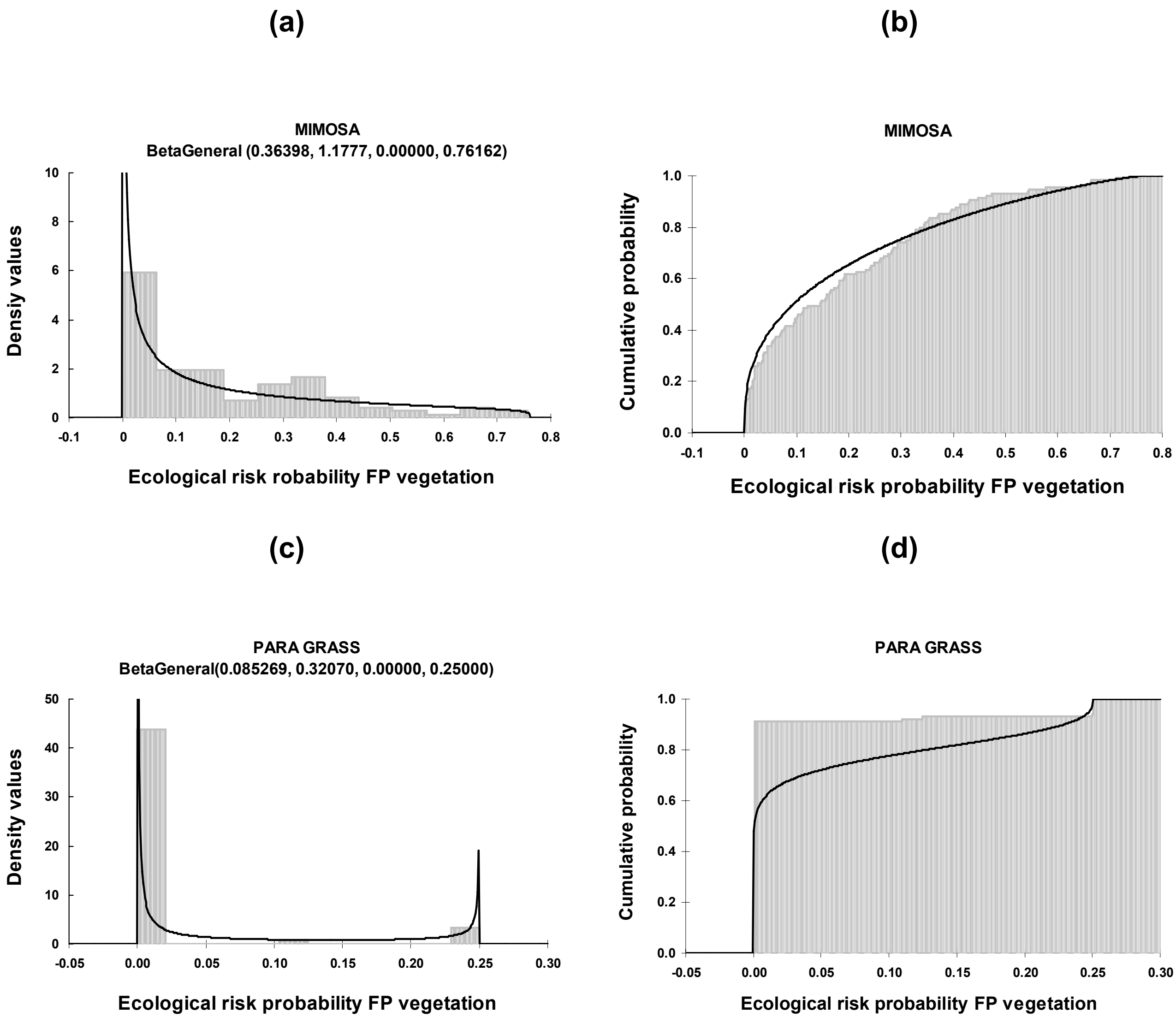


Figure 4.20 a–d. Statistical distributions fitted to observed data on wetland weed abundance (% cover) on the Daly River floodplain, used to define state levels of threats in the Bayesian Network of floodplain health. (a) Probability density function (pdf, Beta General) and (b) cumulative probability curve of mimosa abundance (mean %cover in 250 m x 250 m cells). (c) Probability density function (pdf, Beta General) and (b) cumulative probability curve of para grass abundance (mean %cover in 250 m x 250 m cells, see text for method).

Table 4.9 Logistic regression (logit) equation predicting the occurrence (presence-absence) of mimosa on the Daly River floodplain from the percentage cover of different native wetland plants. N of 24 0's (absence) and 1's (presence). A Quasi-Newton Maximum Likelihood function was used to estimate the final loss.

Final loss= 556.21, $X^2(4) = 15.9$, P=0.003

Variable	Constant B0	Hym_AS	Isch_AS	Rice_AS	EleoS_AS	EleoD_AS
Estimate	0.398	22.013	0.977	1.357	-0.443	-0.245
Odds ratio (unit change)	1.489	3.6×10^3	2.657	3.886	0.642	0.783
Odds ratio (range)			4.640	8.432	0.499t	0.681

Classification of cases: Observed 0's (absences) = 0% correct; observed 1's (presence) = 100% correct.

Table 4.10 Multiple regression summary of the relationship between the distribution and abundance (arcsine % cover) of mimosa in 2003 and the abundance (arcsine % cover) of native vegetation across the Daly River floodplain in 1990.

R= 0.3702, adjusted R^2 = 12%, n= 140, P< 0.001, SE regression = 0.282

Variable	Beta	SE Beta	B	SE B	P
Intercept			0.199	0.051	<0.001
Hym_AS	0.168	0.083	0.145	0.072	=0.044
Isch_AS	0.427	0.097	0.194	0.044	<0.001
RES_AS	0.263	0.095	0.117	0.042	=0.007
(Rice & *E. sphacelata*)					

The above analyses suggest that mimosa clearly has had an impact on magpie goose nesting and dry season refuge habitats on the Daly River floodplain and, by inference, on native wetland vegetation per se. Even so, there are no site-specific data for the Daly River floodplain that can be used to quantify the direct impact of either mimosa or para grass on plant biodiversity, here defined as species richness (number species/unit area). There are, however, sufficient good quality quantitative data from many previous studies over the decades in other floodplains across the NT that can be used to extrapolate to the Daly floodplain (eg see Cook & Setterfield 1996, Walden et al 2004 & Walden & Bayliss 2003 for mimosa impacts; & Douglas & O'Connor 2004 for para grass impacts). This knowledge, summarised in Section 4.2.6 below, is used here to undertake a QERA of the effects of both weeds combined on plant biodiversity. Bayliss et al (in prep. b) re-analysed experimental mimosa data on the Oenpelli floodplain (Cook 1992) and found that, at a 100% cover, 86% of floodplain plant species are lost (effects probability=0.86). Bayliss et al (in prep. a) found also that on the Magela floodplain a 100% cover of para grass will lead to a 100% loss of native plant species because it forms dense monocultures (effects probability = 1.00).

A Venn diagram (Figure 4.21) illustrates how conditional probability theory (Bayesian statistics) is used to derive the combined ecological risk of mimosa and para grass wetland weeds to Daly River plant biodiversity values. The independent probability of ecological risk is simply the probability or likelihood of exposure times the probability of effects, as demonstrated for mimosa and para grass below. However, when independent risks are combined the joint probability, or interaction term, needs to be subtracted to avoid double dipping of effects (ie species already lost to mimosa cannot be lost to para grass or vice versa) as illustrated below.

Pr ecological risk mimosa (P_m) = *Pr (Exposure) x Pr (Effects)*

Pr ecological risk para grass (P_{pg}) = *Pr (Exposure) x Pr (Effects)*

Pr ecological risk floodplain weeds = *Pm + Ppg – (Pm x Ppg)*

Results for the combined QERA of mimosa and para grass on the Daly River floodplain is summarised in Table 4.11, and is relevant at the time of the 2003 baseline weed survey. For current ecological risk in 2007 we assume that no control has taken place since 2003 and that average spread rates estimated on the Mary and Oenpelli floodplains applies. Without control since 2003 the ecological risk values predicted for 2007 are substantial for both weeds.

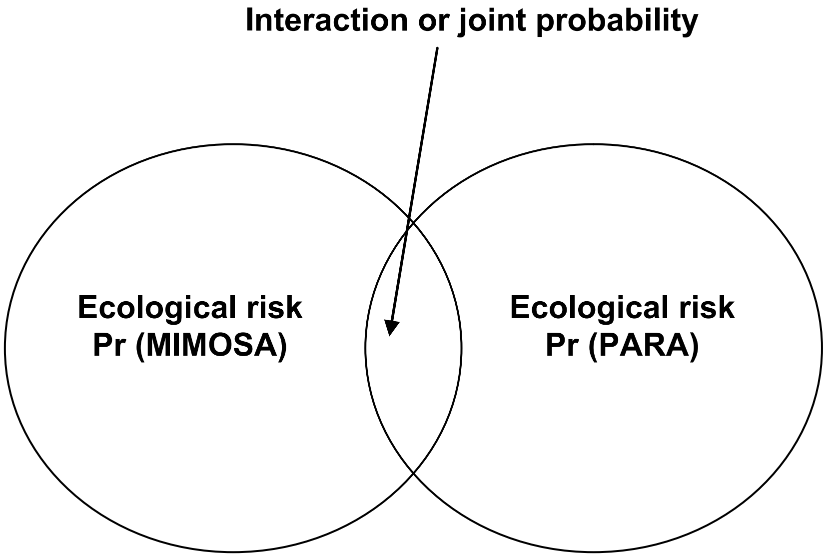


Figure 4.21 Venn diagram showing how conditional probability theory (Bayesian statistics) is used to derive the combined ecological risk of mimosa and para grass wetland weeds to Daly River plant biodiversity values. The joint probability or interaction term is subtracted to avoid double dipping of effects (see formulae in text).

Table 4.11 Derivation of combined ecological risk assessments (ERA) of mimosa and para grass to plant biodiversity (species richness) on the Daly River floodplain in 2003, and that predicted for 2007 in the absence of control. The effect of a 100% cover of para grass is assume 1.0 (Bayliss et al in prep a) and, that for mimosa 0.86 using re-analysed experimental data from Cook (1992). See Section 4.2.6 for estimates of effects probabilities and spread rates derived for other NT floodplains.

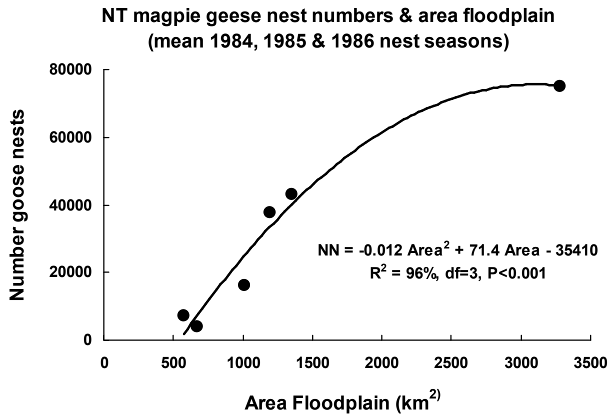
Year	**Mimosa**			**Para grass**			**Combined ERA**
	Exposure	Effects	ERA	Exposure	Effects	ERA	
2003	0.156	0.860	0.134	0.012	1.000	0.012	0.142
2007	1.000	0.860	0.860	0.084	1.000	0.084	0.870

Walden et al (2004) used broad habitat requirements of mimosa and a CLIMEX model to predict the future distribution and, hence, ecological risk of mimosa across northern Australia in the absence of a national control program. Most wetlands appear suitable mimosa habitat; hence, all magpie goose nesting colonies in the NT are at risk from mimosa. Bayliss et al (in prep a) found that the total number of goose nests, estimated in NT catchments west of Arnhem Land and averaged across the 1984 to 1986 breeding seasons, increased with increasing area of floodplain habitat (Figure 4.22a). This relationship was converted to an NT-wide impacts curve that predicts the percentage loss of magpie goose nest production and, hence, annual recruitment, due to the loss of nesting habitat from uncontrolled mimosa colonisation (Figure 4.22b).

4.2.6 Key bioeconomic functions for mimosa & para grass control

Key bioeconomic functions derived for mimosa and para grass control on other NT floodplains are summarised here because they are used to simulate management scenarios in the Bayesian Network for floodplain health. The bioeconomic functions for mimosa are illustrated in Figures 4.23a–c, and were derived from the Oenpelli experience in the late 1980s and early 1990s. The negative linear relationship found between percentage plant species loss and increasing mimosa cover in CSIRO experimental plots (Figure 4.23a, data in Cook 1992) is effectively a typical damage-abundance

(a)



(b)

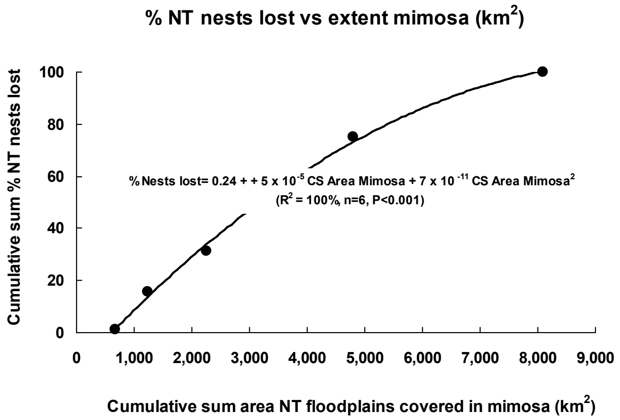


Figure 4.22 a & b Potential impact of mimosa on nesting success of magpie geese in the NT (data from Bayliss & Yeomans (1990b). (a) The increasing and ameliorating nonlinear regression relationship between total number of goose nests (average across the 1984, 1985 and 1986 breeding seasons) and increasing area of floodplain habitat in NT catchments in the Western Domain. Assuming that mimosa is a floodplain generalist and will occupy most habitats it invades, (b) transposes the relationship in (a) to estimate the percentage of nests lost as a function of increasing (accumulated) extent of mimosa (ha) in the NT.

function (Bayliss et al in prep b). The regression model predicts that, at a 100% mimosa cover, 86% of floodplain plant species will be lost. Most wetlands are suitable mimosa habitat and, accordingly, Walden and Bayliss (2003) used empirical data from the Oenpelli and Mary River floodplains to model its increasing spread and colonisation as an exponential function (Figure 4.23b), capped by a ceiling only when the floodplain extent is reached (data from Miller et al 1981, Miller & Lonsdale 1987, Lonsdale 1993). Most cost functions for invasive species show unit costs to increase exponentially with declining abundance, reflecting more time searching than destroying. The control cost curve for mimosa (Figure 4.23c; \$cost/ha vs. mimosa cover in ha) was derived from NT Department of Primary Industries and Fisheries reports on mimosa control on Aboriginal lands (DPIF 1991–1997).

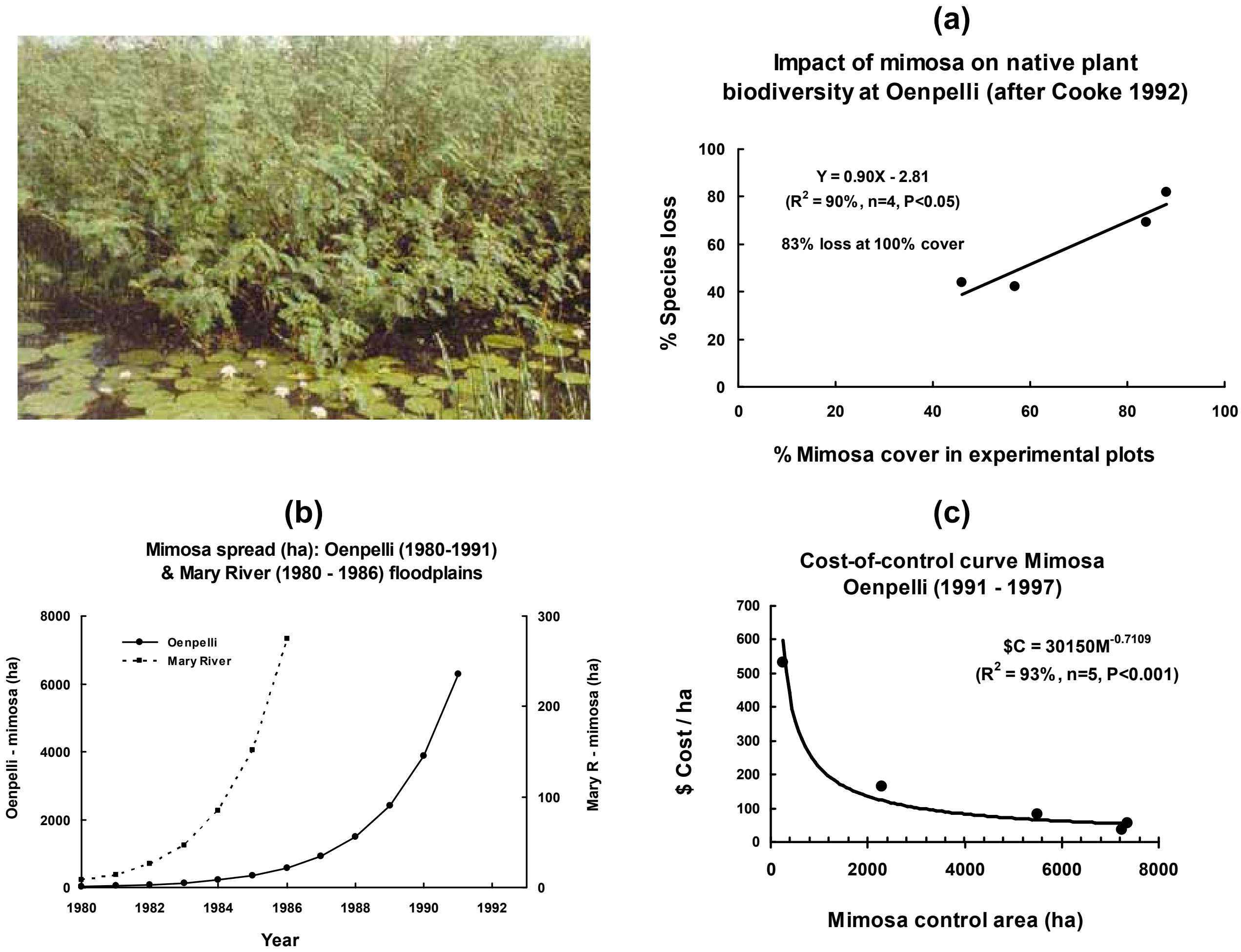


Figure 4.23 a–c. Three key functions needed to cost-effectively manage mimosa on NT floodplains, and are derived from the Oenpelli experience in the late 1980s and early 1990s. (a) The damage-abundance function. The negative relationship between percentage plant species loss and increasing mimosa cover (CSIRO experimental plots, modified from Cook 1992), indicating that at 100% mimosa cover 86% of floodplain plant species are lost. (b) The rate of spread, colonisation or increase function. Mimosa spread (ha) across a new floodplain habitat is exponential with a ceiling generally defined by the extent of the floodplain itself (data from Miller et al 1981, Miller & Lonsdale 1987, Lonsdale 1993). (c) Cost-of-control function ($Cost/ha vs. mimosa cover in ha). Most invasive species cost functions show unit costs to increase exponentially with declining abundance, indicating more time searching than destroying. The control cost curve was derived from Oenpelli mimosa control financial reports (DPIF 1991–1997).

The three key functions needed to cost-effectively manage para grass in conservation areas of NT floodplains are shown in Figures 4.24a–c and were derived from the Magela experience (Walden et al in prep., Bayliss et al in prep. b). A GLM was derived (multiple dependent response equation) from sample plot data across the Magela floodplain in the 2003–04 wet season, and predicts a negative association between the cover abundances of native vegetation and the cover abundance of para grass. The model was then converted into a typical damage-abundance function (Figure 4.24a) that, albeit indirectly, predicts the loss of native floodplain plant cover as a function of increasing para grass cover (Bayliss et al in prep. b). Note the threshold damage detection responses for most plant groups suggest that a 15–20% cover control target would be pragmatic and cost-effective. The average rate of spread or colonisation (ha) of para grass across the Magela floodplain was estimated at 14% pa, or a doubling rate of about 5 years (Figure 4.24b). The control cost curve for para grass (Figure 4.24c) was derived from unpublished Kakadu National Park ground control data for the Noulangie floodplain, South Alligator River (Bayliss et al in prep. b).

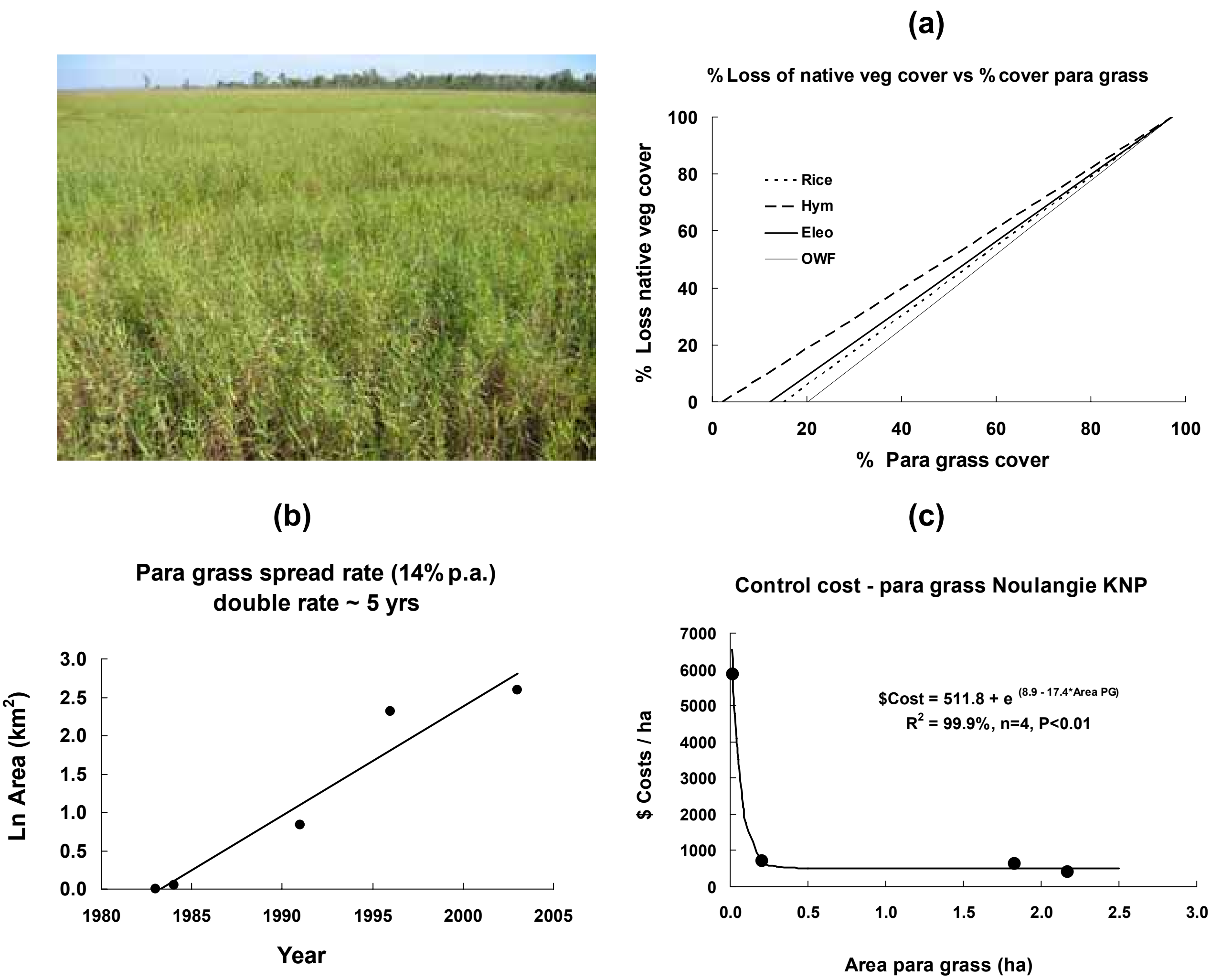


Figure 4.24 a–c. Three key functions needed to cost-effectively manage para grass in conservation areas of NT floodplains (see Walden et al in press). (a) The damage-abundance function. The relationship between percentage losses of native floodplain plants as a function of increasing percentage para grass cover (obtained from sample plots). Note the threshold damage detection responses for most plant groups (Bayliss et al in prep a) (b) The rate of spread, colonisation or increase function. Average rate of spread of para grass (ha) across the Magela floodplain (Bayliss et al in prep). (c) The cost-of-control function. The control cost curve was derived from unpubl Kakadu National Park ground control data from the Noulangie floodplain, South Alligator Rivers (Bayliss et al in prep b).

4.2.7 Bayesian Network for floodplain health

A Bayesian Network for floodplain health was constructed that incorporates the following three ecological assessment endpoints, and associated measurement endpoints, as outlined in the conceptual model (Figure 4.9): (i) the health of magpie goose nesting success in the wet season in relation to potential flow extraction and the current extent of floodplain weeds; and (ii) the health of magpie goose dry season refuge habitat and (iii) floodplain plant biodiversity in relation to the current extent of weeds only. An initial BN was constructed that used variable ranges converted to state levels (Low, Medium & High). The associated probabilities of each state level entered in a Conditional Probability Table (CPT) depended on the form of their probability density functions (pdfs) determined by Best Fit™ (Pallisade 2002a) and an examination of natural breaks in the data (Table 4.12). This process is adopted by many practitioners of BN construction and recommended by Cain (2001). However, probabilities of each state level entered in the CPTs were often arbitrarily determined from poorly fitted statistical frequency distributions (the pdfs) and, combined with the necessity to populate large CPTs, involved much unsatisfactory guess work and creative invention. Hence, large unwieldy CPTs of intersecting child nodes were avoided by replacing them with equations that use outputs (eg other equations, pdfs or constants) from parent nodes as input variables.

Table 4.12 Variable ranges (Low, Medium & High) and associated probabilities used in the initial Bayesian Network to assess risk to floodplain health from simulated water extractions and extent of weeds. Probability density functions (pdfs) and cumulative probability distributions were examined in Best Fit (Pallisade 2002a) for natural breaks in the data. State levels highlighted in bold represent current state levels of variables.

Variable	State Level	Range	Best Fit Distribution	Prob.
Magpie geese nest density	LOW	0 – 11.8 (mean)	Log Normal	0.73
(nos.km^{-2})	HIGH	11.8 +		0.27
Magpie geese density (nos.km^{-2})	LOW	0 – 122 (mean)	Exponential	0.63
	HIGH	122 +		0.27
Wet season flow (log_{10} ML)	LOW	0.00 – 0.73	Exponential	0.50
	MEDIUM	0.73 – 10.00		0.30
	HIGH	10.00 +		0.20
Mimosa ERA	LOW	0.00 – 0.10	Beta General	0.43
(derived spatially)	MEDIUM	0.10 – 0.50		0.50
	HIGH	0.50 – 1.00		0.07
Para grass ERA	LOW	0.00 – 0.10	Beta General	0.73
(derived spatially)	MEDIUM	0.10 – 0.50		0.27
	HIGH	0.50 – 1.00		0.00

The following four scenarios were simulated to assess the independent and combined risks to floodplain health from potential flow extraction and the existing occurrence of wetland weeds: (a) a zero and (b) 20% simulated flow extraction in the absence of weeds; and a (c) zero and (d) 20% flow extraction in the presence of weeds (see Figures 4.25 a–d, respectively). The presence of weeds was set at 2003 levels (mimosa = 16% cover & para grass = 1.5% cover).

Scenario simulations (Table 4.13) show that a 20% flow extraction taken randomly throughout the wet season had little overall influence on floodplain health, with and without weeds. Whilst magpie goose nest density declined in direct proportion to flow extractions, it comprised only one of three floodplain health indicators and the only one directly linked to flow. More importantly perhaps, the threshold level for a ‘Good’ state of nest density was set high (> 12 nest/km^2) and, although arbitrary, it concords with densities obtained after a floodplain flood event and the observed frequency distribution of nests.

Nevertheless, the major influence on floodplain health was the extent of floodplain weeds (Table 4.13) and, hence, the BN was extended to include nodes that allow examination of the costs and benefits of different control scenarios (Figure 4.26a & b). The following two weed control scenarios were examined in the absence of simulated wet season flow extractions: (a) no weed control (floodplain health 10% Poor, 86% Ok & 4% Good; and (b) mimosa control to a target of 10% cover (floodplain health 10% Poor, 18% Ok & 72% Good). Scenario (b) applies to the 2003 baseline for mimosa cover (16% cover) and no para grass control. A 10%

reduction target leaves a residual 5% cover of mimosa, thus avoiding exponentially high control costs associated with eradication or reduction to trace levels. According to our BN, the benefit of the chosen control strategy is a significant increase in the probability of floodplains being in 'Good' condition (72% cf 4%), at an initial cost of $0.75 million.

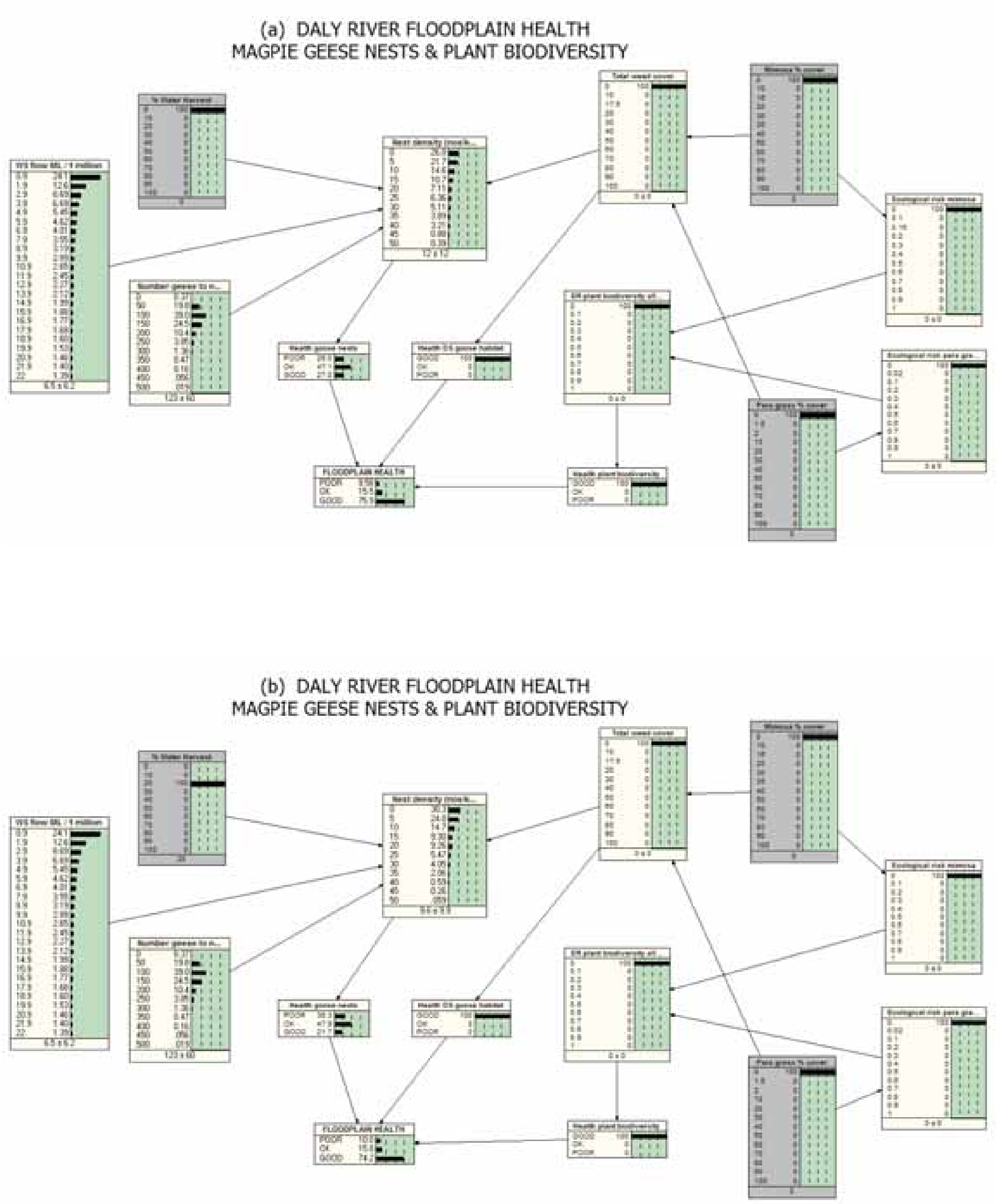
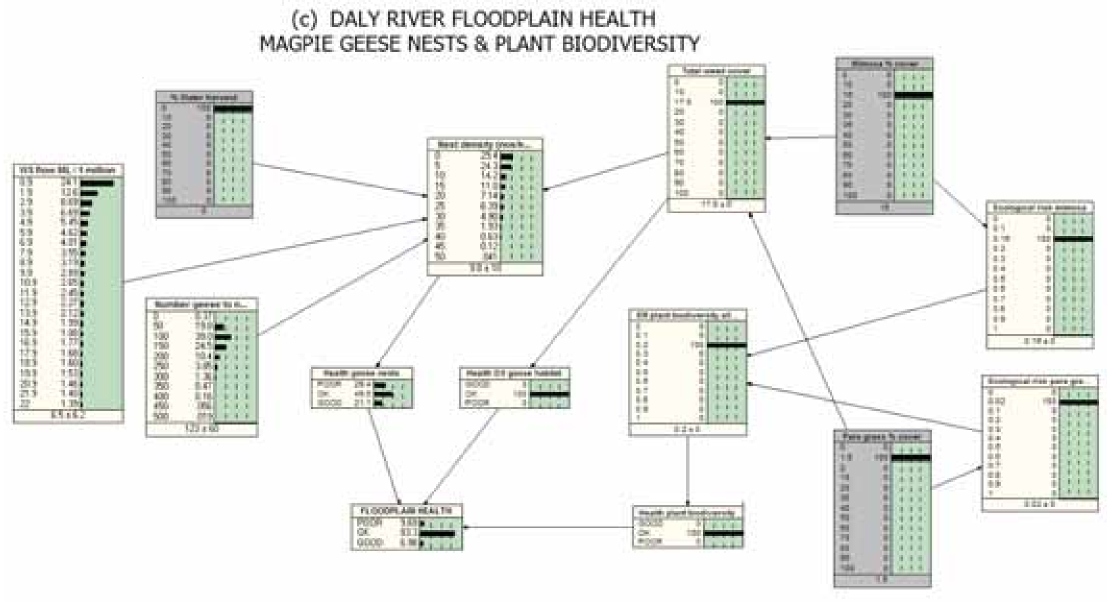


Figure 4.25 a & b Bayesian Network for Daly River Floodplain Health for no weed effects and two wet season flow extraction scenarios: (a) no extraction; and (b) a 20% extraction.



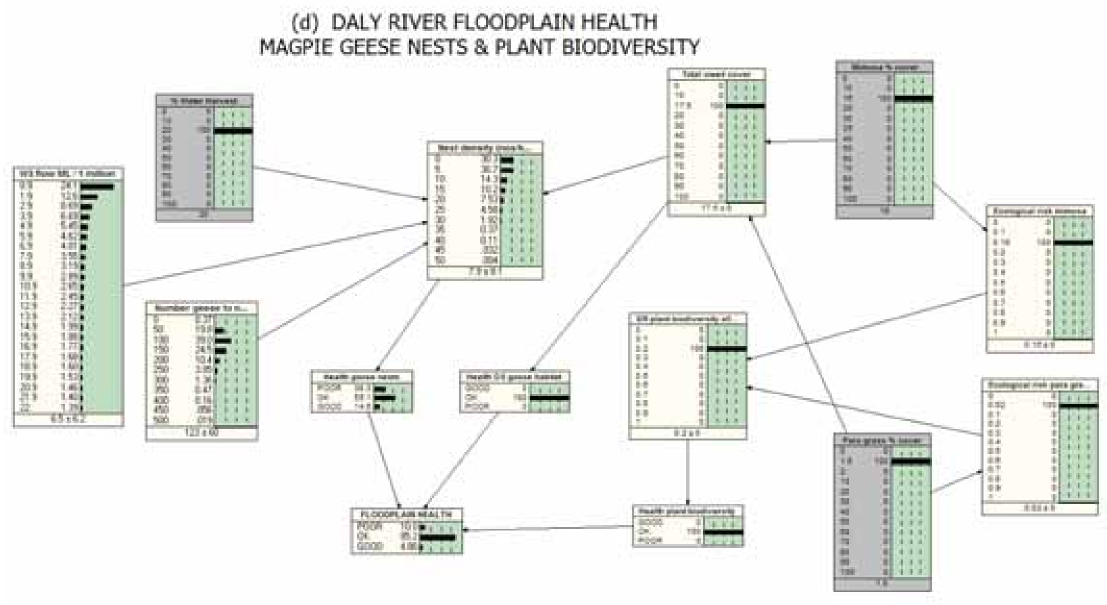


Figure 4.25 c & d Bayesian Network for Daly River Floodplain Health with weed effects and two wet season flow extraction scenarios: (a) no extraction; and (b) a 20% extraction.

Table 4.13 Summary of scenario simulations for the Bayesian Network of floodplain health. Four scenarios were simulated (0 & 20% wet season flow extraction in the presence and absence of floodplain weeds). The probabilities of state levels are presented as percentages.

% wet season flow extraction	Floodplain health	Weeds	
		Absent	Present
0	Poor	9	10
0	Ok	15	83
0	Good	76	7
20	Poor	10	9
20	Ok	16	86
20	Good	74	5

4.2.8 Discussion and recommendations

The Daly River has reliable flows of good quality water throughout the year because dry season baseflow is fed by vast underground limestone aquifers (Jolly et al 2000, Tickell et al 2002), and the surrounding catchment has reasonable quality soils. Hence, in addition to possessing intrinsic conservation and cultural values, the Daly River region has potential for expanded pastoral, cropping, irrigated agriculture and horticulture primary industries. In late 2003 the Daly River Community Reference Group (DRCRG 2004) was established by the NT government to help develop an Integrated Regional Land Use Plan based on an analysis of issues and underpinned by stakeholder engagement. Early community stakeholder consultations (DRCRG 2004) identified water extraction as a potential key threat to in-stream and floodplain environmental flows and, hence, the 'condition' of associated habitats such as riparian, floodplains and estuaries. Wetland weeds were also identified as a key threat to the health of the Daly River floodplain, in particular *Mimosa pigra* (mimosa) and para grass (*Urochloa mutica*). In addition to biodiversity impacts on plants, both wetland weeds will have major impacts on the biodiversity of floodplain fauna. For example, the displacement of tall sedge (*Eleocharis sphacelata & E. dulcis*) and wild rice (*Oryza* spp) dominant communities by mimosa and para grass will have major landscape-wide impacts on magpie goose nesting success and dry season survival (Whitehead et al 1990, this report). Magpie geese are an iconic wildlife species in the NT and an important customary food of Aboriginal people.

Magpie geese in the NT exhibit approximate 20-year population cycles that are coupled to similar periodicities in mean long-term flow for NT rivers such as the Daly River (Bayliss et al in prep a). River flow drives the spatial and temporal dynamics of magpie geese at regional and decadal time scales, most likely through its direct influence on floodplain vegetation dynamics. Flow regimes that trigger floodplain floods on the Daly River are strongly correlated to peaks in nest production (Section 4.2.1) and, most likely, to the availability of food throughout the dry season. The Daly River floodplain contains important wet season nesting habitat for waterbirds, particularly the iconic magpie goose where their nest colonies have supported up to 36% of the NT population (Bayliss & Yeomans 1990b). Additionally, the floodplain is a key regional dry season refuge for many waterbird species, including magpie geese, because of its diversity of perennial river-floodplain habitats (ANCA 1996).

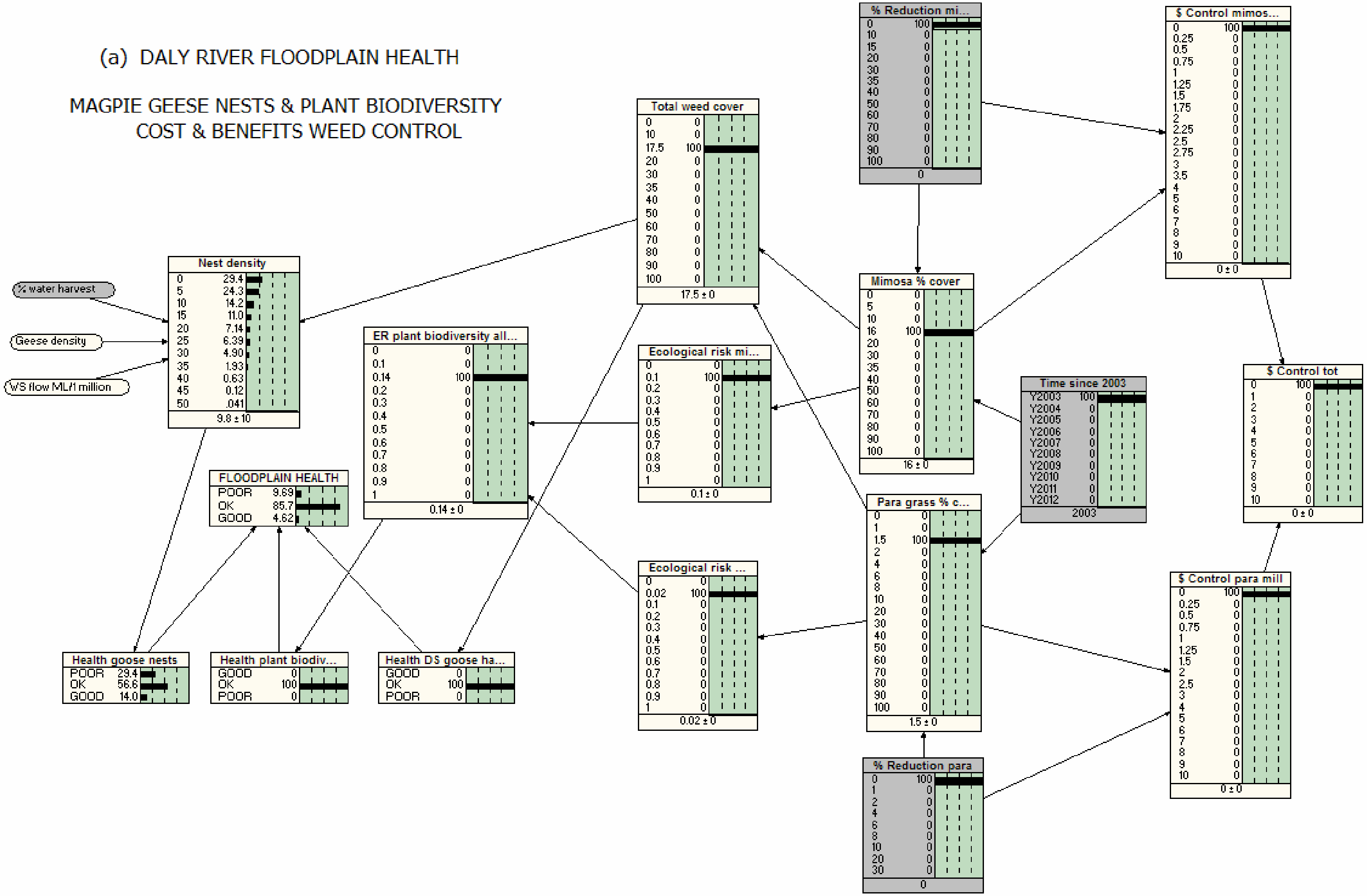


Figure 4.26 a & b Bayesian Network for Daly River floodplain health for two weed control scenarios and no wet season flow extraction: (a) no weed control (Floodplain health 10% POOR, 86% OK & 4% GOOD); and (b) mimosa control to a target of 10% cover (Floodplain health 10% POOR, 18% OK & 72% GOOD) at an initial investment of $0.75 million. Assumes 2003 baseline for weed cover and no control for para grass.

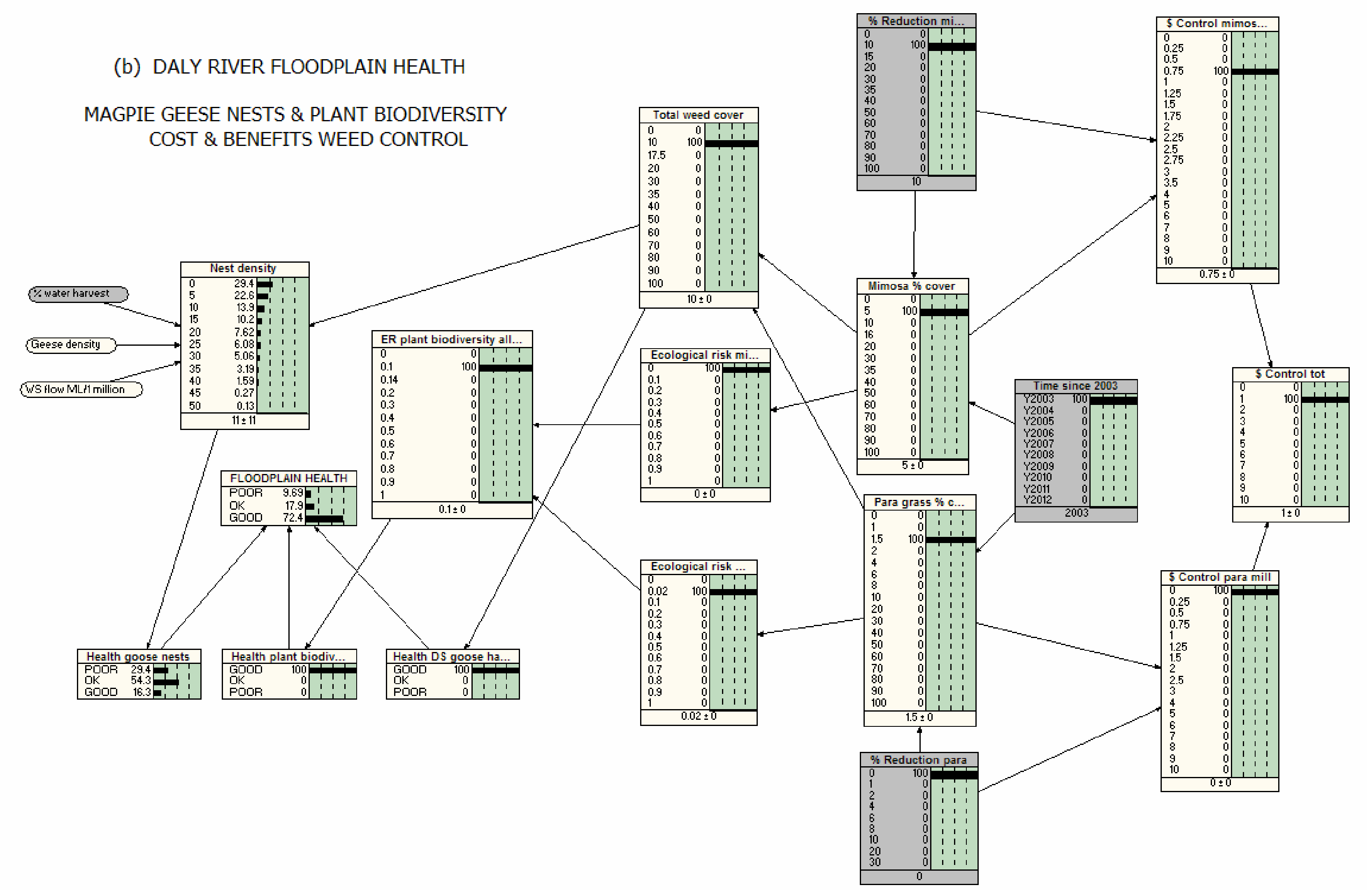


Figure 4.26 a & b (continued). Bayesian Network for Daly River floodplain health showing outcomes for scenario (b) mimosa control to a target of 10% cover (Floodplain health 10% POOR, 18% OK & 72% GOOD) at an initial investment of $0.75 million. Assumes 2003 baseline for weed cover and no control for para grass.

We used the following three ecological assessment endpoints to assess overall floodplain health: (i) the health of magpie goose nesting success in the wet season in relation to potential flow extraction and extent of weeds; and (ii) the health of magpie geese dry season refuge habitat and (iii) plant biodiversity in relation to weeds only. To facilitate our quantitative risk assessment we developed a predictive stochastic model to simulate the effects of flow extraction on nest density, and undertook a spatially-explicit QERA for wetland weeds that characterised impacts on plant biodiversity, nesting habitat and dry season refuge habitat. Despite the high level of uncertainty associated with our nest-flow model, simulation results predict that nest density will decline in direct proportion to flow extractions. Hence, a 20% flow extraction rate will lead to an approximate 20% reduction in nest density. Our spatially-based QERA for mimosa and para grass showed that 16% and 1.2%, respectively, of the floodplain was exposed at a 100% cover in 2003. We then incorporated these quantitative results into a BN to assess the independent and combined risks to floodplain health from simulated flow extraction and wetland weeds. We examined four key scenarios; a 0% and 20% flow extractions in the absence and presence of weeds.

A simulated 20% wet season flow extraction had little overall influence on floodplain health, either in the presence or absence of weeds, because it was only one of three indicators of floodplain health and the only one directly related to flow. More importantly, however, our definition of 'Good' nesting success was weighted towards mean nest density, which did not account for the accumulated population benefit of peak nest densities resulting from extensive floodplain floods on average every second year. The importance of the threshold relationship between magpie geese density and river flow (Section 4.2.1) can only be captured if scenario simulations account for decadal-scale period trends in flow and then goose numbers. Future risk simulations should account for decadal-scale time trends in river flow because a random draw from single pdf may be inappropriate. Nevertheless, in our overall assessment, the major influence on floodplain health was the extent of floodplain weeds. Both mimosa and para grass have major impacts on plant biodiversity per se and loss of vegetation results in loss of wildlife habitat. Magpie geese, for example, seem to return to the same areas to nest in colonies or to seek refuge in the dry season, suggesting that any site-specific loss of preferred habitat on the floodplain from weed colonisation would have major impacts on their abilities to reproduce and survive. Our analysis of habitat preference by geese and weeds supports this suggestion. However, whilst our spatial analyses suggests that magpie geese are site specific in seasonal habitat use and so very susceptible to ubiquitous weeds, protection of their preferred sites can now be more targeted and so possibly more affordable.

The BN was extended to include nodes that allowed examination of the costs and benefits of different broad-scale weed control scenarios. A control strategy that aimed for a 10% residual cover of mimosa significantly increased the probability of the Daly River floodplain being in 'Good' condition (72% cf 4%) at an initial cost of $0.75 million. Intensive mimosa control programs on the Daly floodplain since 2003 appear to have stopped further spread (Neil Schmidt, pers comm, NT Weeds Branch) and, most likely, prevented colonisation of the whole floodplain. The actual costs used to achieve this commendable result could be used to modify the control-cost equations in the BN that is dependent on Oenpelli data 10 years old.

Hence, the ecological risk of mimosa in 2003 and applied to 2007 may be manageable with relatively moderate investment levels because only about 16% of the floodplain was exposed at a 100% cover. And surprisingly, given the extent of cattle stations that encompass the Daly floodplain, only 1.2% of the floodplain was exposed to 100% para grass cover in 2003, although this needs to be re-assessed. Bayliss et al (in prep. b) estimated that the doubling rate of mimosa without control is 1.5 y, and that for para grass 5 y. The ecological risk values predicted for both weeds in 2007 (Table 4.1.1) without control since 2003 are substantial and, most likely, beyond hope because the investment levels needed to reduce cover to manageable levels would be out of reach to government, industry and community groups alike. Given that our risk assessment has exposed weeds as a serious key threat to floodplain values, we recommend strongly that a more formal and detailed weeds risk assessment is undertaken using the National Post Border Weed Risk Management protocols (Standards Australia/Standards New Zealand/CRC for Australian Weed Management 2006).

Whilst we have simulated flow extractions for the Daly River, actual large-scale extractions have not occurred, although that possibility is on the horizon (Hamilton & Gehrke 2005). Hence, there are no local lessons to be learnt to sustainably manage water, nor has the limit to catchment development been identified. Nevertheless, the fate of south-eastern Australian and North Queensland catchments are constantly and perhaps appropriately called upon as examples of what not to do (& see Section 4.2, risks of land clearing to water quality). For example, river flows in the lower Murray were highly variable before regulation, with major floods promoting large-scale recruitment of biota and lower levels of recruitment associated with more seasonal floods (Walker & Thoms 1993). Walker and Thoms (1993) argued that the reduced flood variation in the lower Murray may have reduced the resilience of aquatic species to recover from disturbances, such as invasive species, in particular weeds. This argument seems a snug fit for magpie geese. Future wet season flow extractions may clip the peaks off high nest densities because the frequency of extensive floodplain floods will reduce. The loss of such recruitment bursts to the natural 20 y high-low-high cycles over decadal and multi-decadal time scales may be substantial as indicated above, and warrants investigation, particularly in combination with loss of nesting and dry season refuge habitats from uncontrolled wetland weeds. Chapin et al (1998) argued that changes in biodiversity can have significant impacts on ecosystem and landscape processes, both on a daily basis and in response to extreme events.

Taken at face value our BN produced useful and intuitive results. However a couple of caveats need to be highlighted. Hidden amongst the equations and probabilities is the inescapable fact that the risk modelers' own subjective value judgments currently determines what levels of each assessment endpoint constitutes 'Poor, Ok and Good'. Hence, this is probably the point where comprehensive stakeholder consultation needs to take over from objective probabilities in order to elicit subjective values and opinions from the people most affected by any investment decision based on risk. It is highly recommended, therefore, that before assessment endpoints are reached stakeholders should again be consulted in order to incorporate more relevant subjective values and opinions with regards to what constitutes 'ecological health'. Hence, whilst stakeholder participation is absolutely essential in developing conceptual models at the start of the QERA process, it is also critical at the very end. The middle bit of the BN construction involves scientifically based technical value judgments. With regards to social value judgments there can be no right or wrong as all views (within limits) are equally valid. However, with regards to technical value judgments there is a right and wrong.

Irrespective of the above conclusions, the risk assessment approach adopted here is compatible with our main overall aim of developing appropriate analytical tools and establishing a framework that can be modified to accommodate changes to underlying model assumptions, qualitative definitions of condition and/or management rules. For example, we simulated wet season flow extraction based on a couple of rules and assumptions (Section 4.2.1) that err towards simplicity and, hopefully, practicality in the face of highly variable and, hence, uncertain flow events (P Jolly pers comm, NT NRETA). We were attracted to these simplifying assumptions because even over short monthly time steps we could not identify the 'peak' in a flood event with certainty, given that some time must first pass before, on average, recessional flow is said to be occurring. In retrospect it can be defined precisely and, in contrast, instantaneously it would be a probability. Hence, only wet season flow extraction is simulated and irrespective of hydrograph stage. The 20% cap on environmental flow extraction recommended by Erskine et al (2004) was applied to all of the wet season flow. For magpie geese this may over-rate the impact of simulated flow extractions if most future extractions are in the dry season as the frequency of floodplain floods will be reduced. In contrast, however, it may under-rate the impact on barramundi (see Section 4.3) because recessional flows may be just as ecologically important in maintaining connectivity as rising flood waters and, hence, movements of fish between nursery habitats on floodplains and in-stream habitats in rivers. Nevertheless, the flow extraction recommendations of Erskine et al (2004), although complex, are based on well considered assessments of current knowledge of environmental requirements of key aquatic ecological processes in the Daly River. Hence, they should be assessed at some stage in the future should they be endorsed, and our QERA framework could accommodate such an assessment.

4.3 Risks of water extraction on in-stream health

Executive summary

Water extraction has been identified as a potential key threat to environmental flows in the Daly River and, hence, the condition or health of in-stream habitats. The aim of this section is to develop a QERA framework to assess in-stream 'health' from the threat of potential wet season flow extractions. The barramundi is an important recreational and commercial fish in the NT, and has a life cycle dependent on the connection between freshwater and estuarine ecosystems. Two assessment endpoints for barramundi were used to assess in-stream health; catch and population abundance as indexed by Catch Per Unit Effort (CPUE). Whilst both endpoints are related they have different social contexts with respect to river value. Catch is a socio-economic assessment endpoint and, in contrast, population abundance is an ecological assessment endpoint. Commercial barramundi catch and effort declined on average between 1983 and 2005, and preceded the 1989 closure to commercial fishing. In contrast, recreational catch and effort increased on average between 1985 and 2005. The CPUE population abundance indices of all barramundi fisheries (angler Classic Tournament, angler Tour operators & commercial gillnet) increased on average over time and in tandem with an average increase in wet season flow. As for magpie geese, barramundi most likely exhibit decadal trends in abundance in sympathy with the 22y period trend in river flow.

Stochastic process models were developed to predict barramundi catch from fishing effort and wet season flow, and population abundance from wet season flow. The relationship between barramundi catch and natural flow was used to predict potential tradeoffs between reduced flow from extractions and lost fisheries value. Similarly, the relationship between CPUE and natural flow was used to predict potential impacts on barramundi abundance from flow extractions. Monte Carlo simulation and sensitivity analysis were used to separately account for model uncertainty and parameter variability in model predictions. A Bayesian Network (BN) was then developed to examine the influence of 20% and 50% flow extraction scenarios on in-stream health, as indexed by catch and population abundance. The stochastic process model for catch vs. effort and flow, and CPUE vs. flow, were used in the intersecting child node rather than resorting to an unwieldy Condition Probability Table. The BN for barramundi catch was converted into a Decision Tree to help clarify the influence pathways and trade-offs between different water extraction and fishing effort policies.

In summary, commercial and recreational barramundi catches in the Daly River, and the abundance of the catchable population, appear highly sensitive to flow extraction; the greater the extraction rate the greater the negative impact on both barramundi socio-economic and ecological assessment endpoints. However, research is required on seasonal flow relationships for a range of fish species encompassing the diversity of life histories and functional community groups found in aquatic habitats of the Daly River-floodplain ecosystem (M. Douglas pers. comm.). We recommend that indirect impacts of water extraction on fish communities be examined also in future risk assessments, such as disruption of biophysical processes at the terrestrial-aquatic interface associated with land use dependent on water, and the impacts of invasive species such as aquatic weeds and exotic fish. To conclude, we highlight the importance of being able to differentiate between sudden changes in fish catches due to a climate regime shift with those due to changes in fishing effort. Therefore, we recommend also that future barramundi management should account for the strongly coupled decadal trends in climate-river flow-abundance in order to minimise risk to stock levels and sustainable catches during low-flow periods.

Technical summary

1 Two assessment endpoints for barramundi were used to assess 'in-stream health' of the Daly River; catch and population size as indexed by Catch Per Unit Effort (CPUE). Commercial and recreational barramundi catches are viewed here as socio-economic endpoints and, in contrast, the size of the barramundi stock is viewed as an ecological assessment endpoint. Whilst both endpoints are related they are treated independently because their associated measurement endpoints depend on the same data.

2 Commercial barramundi catch and effort show parallel exponential declines between 1983 and 2005 that preceded the 1989 closure. In contrast, recreational Classic Tournament and recreational Tour operator catch and effort increased in the periods 1985–2005 and 1994–2005, respectively.

3 The CPUEs of all barramundi fisheries in the Daly River were highly inter-correlated and, strongly and positively correlated to wet season flow (eg angler Classic R^2=51%, n=18, P<0.001). Additionally, cusum (cumulative sum of the mean deviations) trends in flow and cusum trends in all CPUE population indices were highly coherent (eg angler Classic R^2=93%, n=18, P<0.001) suggesting 22 y period trends in abundance. Barramundi stock increased on average between 1983 and 2005 in tandem with an average increase in wet season flow. Although the average population rate of increase was only 2% p.a., this translates to a 55% increase in stock level in 2005 above 1983.

4 Cross-correlation correlograms between 'Classic' recreational barramundi CPUE and wet season flow, and between commercial barramundi CPUE and wet season flow, show significant positive correlations at 0, 2 and 3 year time lags, and 0 and 2 year time lags, respectively. The positive correlation between instantaneous catch and flow may encompass a large component of increased catchability due to increased fish movements, in addition to increased recruitment and/or survival effects. In contrast, positive correlations at 2 and 3 year time lags suggest enhanced recruitment and/or survival of the cohort that has reached the size limit to enter the fishery (currently 55 cm total length in commercial & recreational barramundi fisheries). This cohort may comprise mostly upstream males migrating to the river mouth to spawn.

5 Multiple regression analysis shows that commercial barramundi catch is strongly related to both effort and flow (R^2=67%, n=23, P<0.001; both variables significant) and, similarly, for recreational Classic barramundi catch (R^2=78%, n=18, P<0.001; both variables significant). Total recreational catch was strongly related to total recreational effort (angler Classic hours + Tour line hours) and flow (R^2=94%, n=18, P<0.001; both variables significant), and assumes that both angler efforts are equivalent. In all multiple regression models effort had 1.5–3.0 times more influence on catch than flow.

6 The strong relationship between barramundi catch and natural flow was used to indirectly predict potential tradeoffs between reduced flow from extractions and lost fisheries value, in terms of either revenue generated from the commercial and recreational fisheries, or simply catches (socio-economic assessment endpoints). Similarly, the strong relationship between barramundi CPUE and natural flow was used to indirectly predict potential impacts on population abundance from reduced flow due to extractions (the ecological assessment endpoint). Hence, two conceptual models were developed to guide assessment of Daly River in-stream health under scenarios of increasing wet season flow extraction. The multiple regression equations for catch vs. flow and effort, and the regression equation for angler Classic CPUE vs. flow (& representing also the commercial and Tour fisheries), were used as stochastic process models to simulate the effects of flow extraction. Monte Carlo simulation and sensitivity analysis were used in model predictions to separately account for model uncertainty and parameter variability.

7 Simulation results predict that the percentage reduction in mean total catch of all Daly River barramundi fisheries will initially rapidly increase as the proportion of wet season flow extraction increases (0–100%), but that the reduction in catch will ameliorate and then asymptote at 100%. The model predictions are highly certain. Simulation results predict that barramundi populations in the Daly River (adjusted for a 15% reduction in stock due to current offtake) will linearly decrease as the percentage of wet season flow extraction increases (0–100%), but that the rate of decrease is not directly proportional. For example, a 20% simulated wet season flow extraction will reduce barramundi populations by 32%, and that for a 50% flow extraction by 58%. Certainty level is high at low levels of percentage flow extractions but decreases with the level of extraction.

8 A Bayesian Network (BN) for in-stream health was constructed that incorporates commercial and total recreational barramundi catches. Version 1 used standard methods in that flow, effort and catch variable ranges were converted to state levels (Low, Medium & High). However, probabilities of each state level entered in the Conditional Probability Tables (CPT) were often arbitrarily determined and, combined with the necessity to populate large CPTs, involved much unsatisfactory guess work. Hence, large unwieldy CPTs of intersecting child nodes were avoided by replacing them with equations that use outputs (eg other equations, constants, probability density functions or pdfs) from parent nodes as input variables. The stochastic process models developed above to simulate the effects of flow extraction on catch were therefore combined into the one BN.

9 Two scenarios were examined for the recreational barramundi fishery only because commercial barramundi fishing in the river reach of the Daly River fishing zone is closed; no flow extraction and a 20% wet season flow extraction. Commercial effort was set to the lowest level recorded in 2005 and, hence, their catch is set to 'Poor'. Recreational effort and wet season flow are characterised by their pdfs and, hence, node inputs and outputs encompass the whole range of effort and flow conditions encountered during the operation of the fishery. For the 'no flow' extraction scenario 65% of the assessment of

in-stream health is classified as 'Ok and Excellent', with the majority being 'Excellent' (48%). In contrast, for the 20% wet season flow extraction scenario only 17% of the assessment of in-stream health is classified as 'Ok and Excellent', with the majority being 'Poor' (83%).

10 The above BN for barramundi catch was converted into a Decision Tree (DT) to more clearly examine the influence pathways and trade-offs of different water extraction and fishing effort policies on commercial and recreational catch. An attempt was made to use a monetary value ($) as the assessment endpoint in the DT so that the benefits and costs of alternative policy options for flow could be compared (eg commercial vs. recreational fisheries, fisheries vs. agricultural production, etc) and, needless to say, is only a starting point. The DT optimal policy for both commercial and recreational barramundi fishery in terms of either total catch (numbers or weight) or dollar value is, as expected, no water harvest, high flow and high effort. More importantly, DT analysis showed that the recreational fishery decreased in value by 77% with a 20% water harvest ($129K c.f. $29K) under high flow and high effort states (present condition). Similarly, the DT for a commercial fishery assumed to be still operating decreased in value by 67% with a 20% water harvest ($1.9 million c.f. $0.7 million p.a.) under high flow and effort states. However, under high flow and low effort state (present condition) the decrease in value is 64% ($957K c.f. $343K p.a.).

A similar approach as for catch was used to develop a BN to assess barramundi population abundance under different flow extraction scenarios. State levels for in-stream health based on barramundi population size are arbitrary but, nevertheless, underpinned by basic harvesting dynamics theory and the precautionary principle. Two simulation scenarios were undertaken: no flow extraction and a 20% wet season flow extraction. Under a scenario entailing natural flow conditions and no flow extraction, most (97%) of the barramundi population is classified as 'Ok'. In contrast, under a 20% flow extraction scenario 100% of the barramundi population is classified as being in 'Poor' condition.

There were sufficient commercial catch data for nine other fish species to examine relationships between catch in combination with effort and flow. Linear regression analysis was used also to examine trends in population size as indexed by CPUE over time (years). Catch significantly increased with effort only for cod, jewfish and mackerel. Flow was positively correlated with catches of shark, mackerel and snapper, although these relationships appear complex and not as direct as that for barramundi. No fish species examined declined significantly between 1983 and 2005, and only snapper and mixed fish significantly increased on average over time.

In summary, commercial and recreational barramundi catches, and the abundance of the catchable population, appear highly sensitive to flow extraction; the greater the extraction rate the greater the negative impacts on both barramundi socio-economic and ecological assessment endpoints.

4.3.1 Introduction

Life history of barramundi

The barramundi (*Lates calcarifer* (Bloch)) is a large perch growing to 150 cm total length and weighing up to 40 kg, and inhabits coastal rivers, estuaries and inland waters accessible to the sea throughout the tropical and semi-tropical waters of the Indo-pacific Region (Davis 1982). Barramundi are caladromous, generally spending the first years of life in the upper part of rivers and move downstream to estuaries to spawn as 3–5 year old mature males (Griffin 1987). Most downstream movement in the NT commences in August–September, and spawning is thought to precede the onset of the wet season and associated flows, or coincidental to it. Most males that spawn are thought to remain in the tidal, brackish parts of rivers where they eventually become females (Griffin 1987, see Moore 1979 & Davis 1982). Despite variable spawning times documented for barramundi across its northern Australian range (Davis 1982, Russell & Garrett 1985), Pusey et al (2004) suggested that spawning precedes the wet season and the onset of monsoonal flows. However, maximum gonadal activity in the NT occurs from October to December (Davis 1982), coinciding with the onset of the wet season rains but preceding peak rains in February and peak flows in February–March.

In the mid-1980s barramundi was an important commercial and recreational fish species (Russell & Garret 1985) in the NT, and this continues to be true today but with greater emphasis on the potential impact of recreational fishing on wild stocks (Coleman 2004, Pusey et al 2004). Hence, because of its economic and social importance, much research has been undertaken since the early 1950s into their life history (age & growth, sexual maturity, food), seasonal migrations, salinity impacts and response to harvesting (Davis 1982, 1985a&b, 1986; Davis & Kirkwood 1984; Russell & Garrett 1985; Griffin 1987, 1988; de Lestang & Griffin 2000). The descriptive links between barramundi life history, large-scale movements within river basins and macro-scale habitat use are generally well understood, although little quantitative research has been undertaken into habitat use at micro and meso-scales (Pusey et al 2004). Pusey et al (2004) reviewed the movement ecology of barramundi, summarised here because it is key knowledge needed to assess the health of in-stream ecosystems based on fish catch and population abundance. Larvae hatch in estuarine and near shore habitats and are passively delivered by tidal action to supralittoral swamps near the river mouth. In the NT larvae rely more on floodplain and billabong systems many kilometres from the coast, and so may be assisted in accessing these habitats by very large spring tides (Davis 1982). By the end of February or peak rains 0+ juveniles or post-larvae recruits depart tidal creek habitats (stimulus unknown) and occupy nursery swamps, where yearlings remain until the mid-dry season. Extensive active movement becomes a feature of the biology of juvenile barramundi in the 1+ age class, comprising mostly immature males that move far upstream colonising a range of freshwater habitats including billabongs, floodplain lagoons and wetlands. Access to these habitats is largely governed by flooding regime, and juvenile barramundi remain in upstream freshwater habitats for 3 to 5 years before migrating to spawning grounds over a number of months.

The commercial and recreational fisheries in the Daly River

The aim of this section is to develop a QERA framework to assess in-stream 'health' in the Daly River from the threat of potential wet season flow extractions. A necessary pre-requisite is an understanding of the nature and dynamics of the barramundi fishery in the NT and, in particular, the Daly River fishing zone.

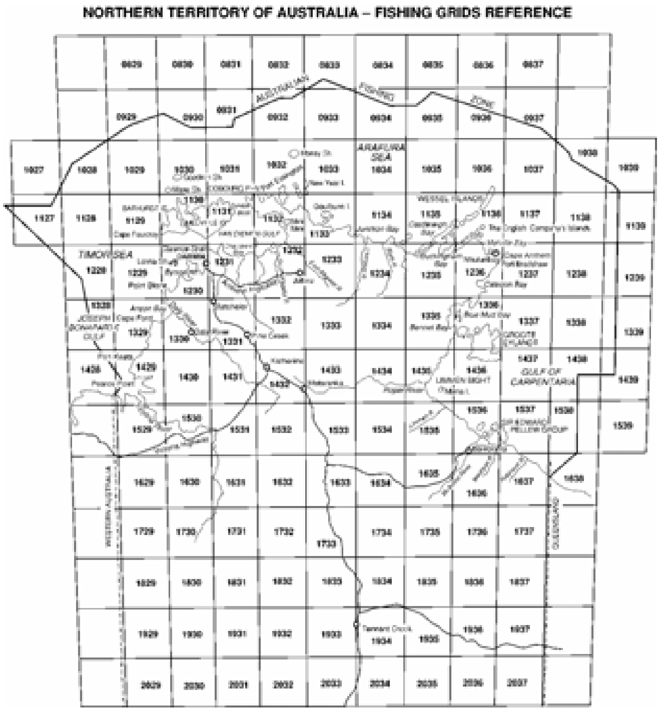
Griffin (2007) provided an updated chronology of barramundi management in the NT based on previous chronologies from the 1985 Barramundi Task Force Report (Hill & Grey 1979, Grey 1985) and the 1991 Barramundi Management Plan (& see Walters et al 1997). Commercial barramundi fishing is one of the most important inshore gillnet fisheries in the NT with a history of management dating back to the early 1960s. Management of commercial barramundi catch has involved controls on effort, and current restrictions adopted in 1991 include limits on gillnet length and net length units, limits on net mesh size, length of fish, licence fees and fee structures, a closure season (October to January inclusive) and declared closure areas. The commercial fishing season operates each year from 1 February through to 30 September, and the majority of commercial fishing in 2005 took place in the Murganella, North Arnhem, Blue

Mud Bay, Roper River and Daly River regions (DBIRD 2005). The current size limit for barramundi for both commercial and recreational fishing is 55 cm. In 1989 the Daly River reach and the eastern part of Anson Bay (Figure 4.27a, fishing zones 1329 & 1330) were closed to commercial barramundi fishing because of concerns about over-fishing and potential impacts on future stock levels. All commercial barramundi data post-1989 are for Anson Bay west of the closure line at the river mouth. In 1989 the recreational fishing season closed in the lower Daly River and Anson Bay from 1st October to 31st January. The Daly River Seasonally Closed Area is defined as the area downstream form the outlet of Moon Billabong to the commercial closure line (see unshaded lower river reach in Figure 4.27b). Concerns about over harvesting led to multiple river closures in the NT by 2004 (eg Daly River & Anson Bay, parts of the Mary River, Darwin Harbour & Shoal Bay, McArthur River & Adelaide River; see DBIRD 2003). The NT Fishery Status Reports 2003 (DBIRD 2004) for barramundi state that commercial catch has decreased over the last few years and may be a function of many factors including a run of poor wet seasons and, hence, limited recruitment. The total commercial harvests of barramundi for the NT in 2003 and 2005 was 660 and 552 tonnes, respectively, which were apparently well within the range of estimated sustained yields. However, the 2003 Report notes that in accessible and heavily used river systems such as the Daly River, recreational fishing pressure in conjunction with commercial fishing may have increased total harvest to levels approaching maximum sustainable yield.

Griffin (1979) surveyed recreational catch and effort rates of barramundi in 1978 and 1979 as a balanced response to increasing concern about declining commercial catch rates since 1976, and concluded that recreational barramundi fishing was important economically and socially. The only survey of recreational fishing since then is that by Coleman (2004) as part of the National Recreational Fishing Survey. She stated enthusiastically that *'Recreational fishing has always been a popular pursuit of residents and visitors to the Northern Territory. Abundant fish stocks, accessible waterways and favourable weather conditions combine to provide a fishing experience unparalleled in Australia'*. The 2000 survey identified barramundi as the most popular target species in the NT, with an estimated total catch of over 400,000 and an annual harvest of 100,400. The total number of barramundi caught increased by 60% between 1995 and 2000 and, similarly, barramundi effort increased from being 38% of the total recreational effort to 43%. Despite such figures, Coleman (2004) argued that reliable assessment of the catch and effort contribution of recreational fishing to total catch has always been hampered by lack of reliable statistics. The same issue also plagues reliable assessment of the socio-economic benefits that recreational fishing contributes to the national and local economies. Whilst commercial fishing and fishing tour operator activities are quantified through a reporting system attached to licence requirements, previous to the recent High Court decision on Native Title rights at Blue Mud Bay and requirements for all recreational fishers to obtain licences from the Northern Land Council (NLC), data on recreational fishing effort in the past has always been imprecise and anecdotal at best (Coleman 2004). She concluded that to make informed management decisions on recreational fishing would require estimates of the total amount of fish being caught, where they were being taken and by whom.

Commercial (1983–2005) and recreational barramundi catch-effort data were obtained from NT Fisheries (Paul De Lestang pers. comm.) in order to undertake a QERA of Daly River flow extraction scenarios on catch and population size as indexed by Catch Per Unit Effort (CPUE). The Daly commercial catches are defined as catches in NT Fishing grids 1330 and 1329 east of Cape Ford (Figure 4.27a). Although reliable recreational catch-effort fishing data

(a)



(b)

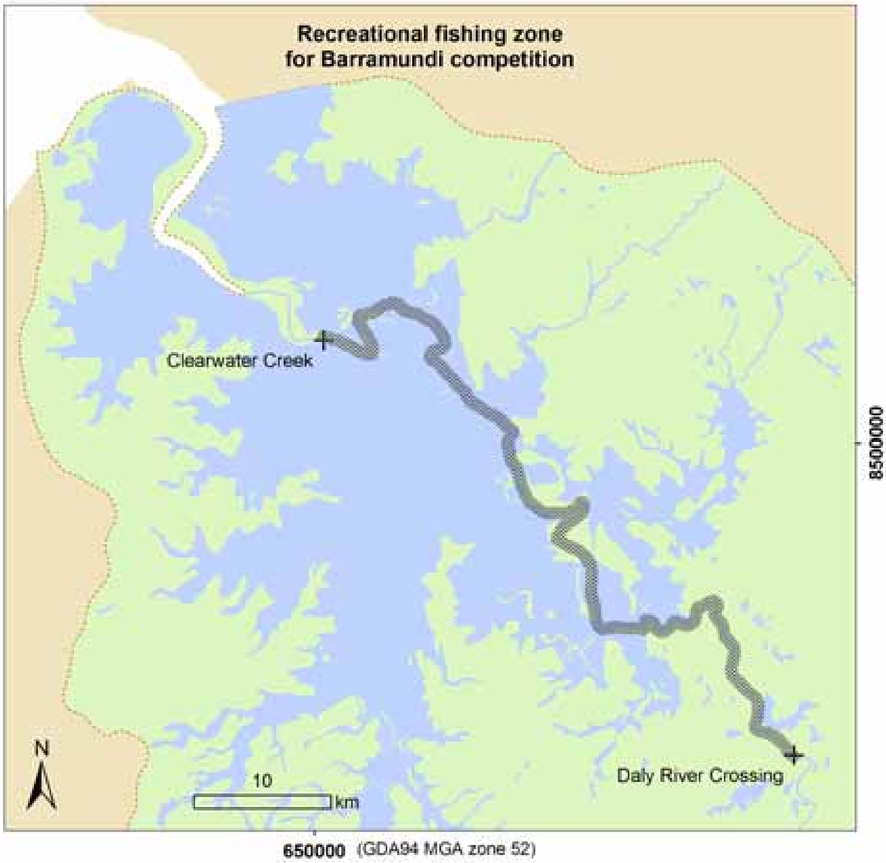


Figure 4.27 a & b (a) Fisheries Grid Reference Numbers for commercial management zones and (b) the location of the Recreational Fishing Zone on the Daly River for the annual 'Classic' barramundi competition.

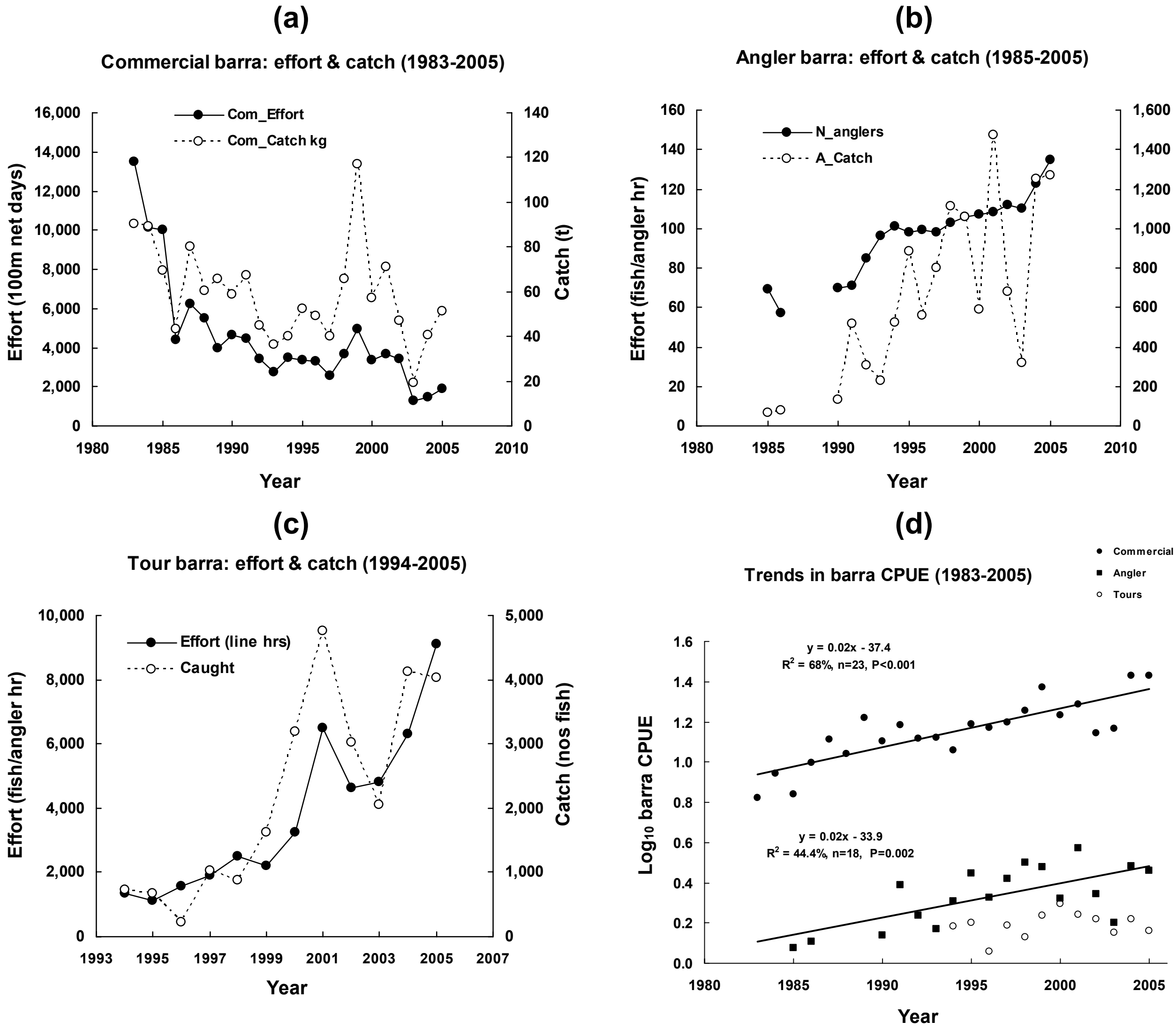
for barramundi are difficult to obtain, the Daly offers two opportunities. The first is catch-effort data from Fishing Tour Operators (1994-2005) who are required to lodge returns as part of their licence requirement (Tour recreational catches are here defined as catches in NT Fishing Grids 1331, 1431, 1330 and Grid 1329 east of Cape Ford, see Figure 4.27a). The second is voluntary survey data obtained from the NT Barramundi Classic Fishing Tournament (1985–2005, excluding 1987–1989). The Tournament is an annual tag and release fishing competition carried out over a five day period during the dry season (April-Sept), and has occurred at three separate venues (Mary River/Corroboree 1982–84, 1987–88; Daly River 1985–86, 1990–2005+; Port Hurd 1989). White (1998) undertook a detailed analysis and assessment of catch-effort survey data for the period 1982-1997. The location of the Recreational Fishing Zone on the Daly River for the annual barramundi Classic competition is shown on Figure 4.27b.

Commercial barramundi catch (t) and effort (100 m net sets/day) shows a marked decline between 1983 and 2005 (Figure 4.28a). The decline precedes the 1989 closure to commercial fishing and, nevertheless, there is a small increase in effort and a corresponding marked peak in catch at the time of closure. In contrast, both recreational Tour operator and recreational angler Classic catch and effort increased in the periods 1985–2005 and 1994–2005, respectively (Figure 4.28b & c).

Despite the catch-effort histories of all barramundi fisheries described above that suggest barramundi populations in the Daly River have sustained fishing pressure in one form or another for a couple of decades, their abundance as indexed by commercial and Classic CPUE significantly increased on average over time (Figure 4.28d). The regression for Tour CPUE vs. time is not significant, but this is because a nonlinear trend is averaged over a short time interval (n=11 years). Additionally, and as expected, commercial and Classic CPUEs are highly correlated (see Figure 4.29a), suggesting that they are indexing the same population trend. Hence, barramundi populations have been increasing on average by 2% p.a., which translates to a 55% increase in stock abundance in 2005 from 1983 levels. The results suggest also that this trend may be independent of the reduction in commercial fishing effort and associated catch (but see below). Hence, the interaction between commercial and recreational catch and effort is examined more closely.

The time trends (Figure 4.29b) in commercial and total recreational barramundi catches (Tour plus Classic numbers caught) show no apparent correlation (Figure 4.29c). Additionally, time series analysis (cross correlation on $\log_{10}$ transformed data, detrended & mean subtracted) showed that there was no lagged effect or interaction between commercial and recreational catches (Figure 4.29d).

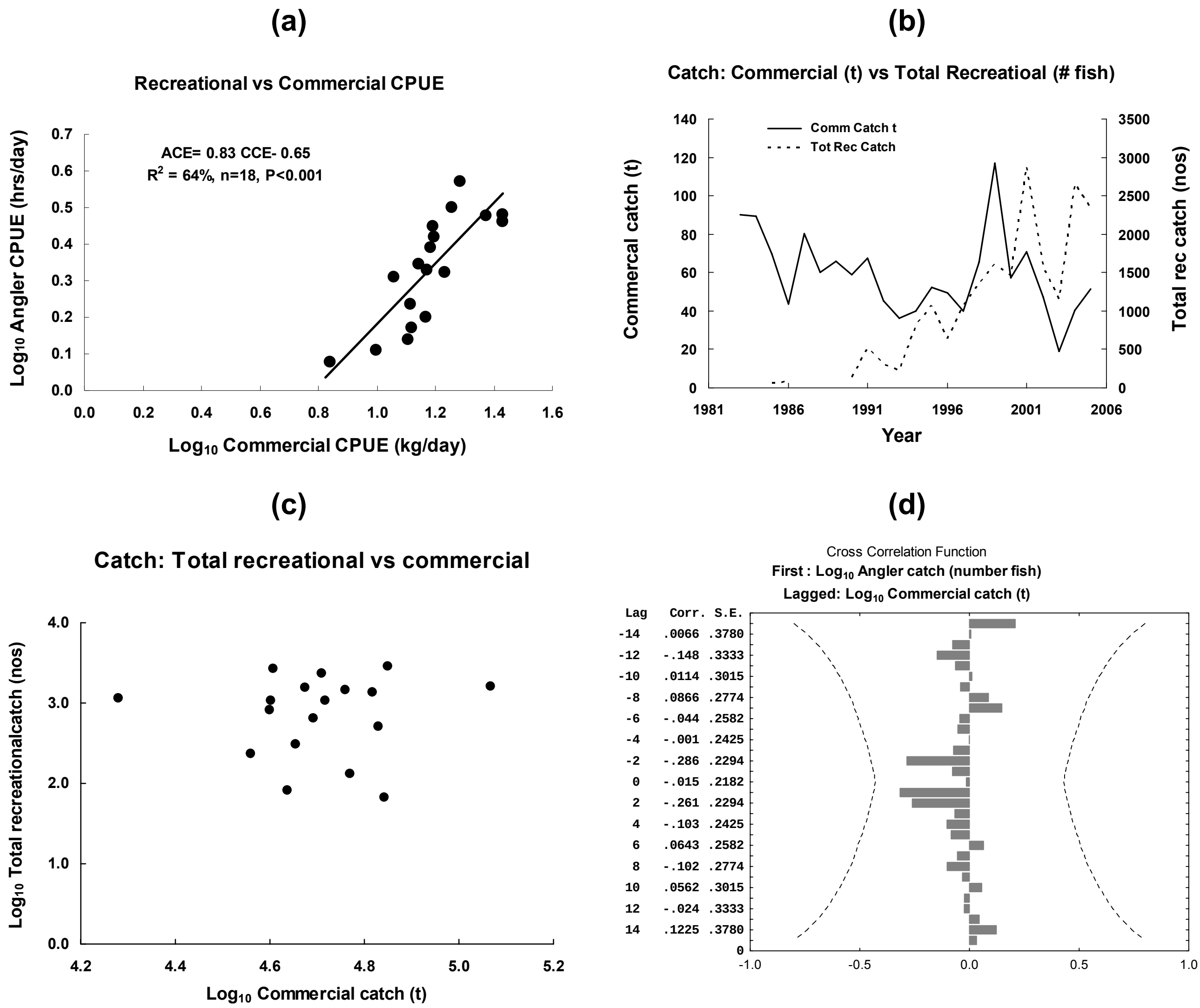
Paradoxically though, a traditional catch-effort analysis (Figure 4.30) indicated that commercial catches were just to the right of maximum sustained yield, suggesting over-fishing and a need to reduce effort to ‘safer’ levels to the left of the curve, and explains why NT Fisheries responded with a river closure in 1989. The statistical significance of the quadratic polynomial term, however, is determined by only one catch-effort point in a highly variable data set. Hence, this paradox is examined further in Section 4.3.3 through examination of the simultaneous relationships between commercial and recreational catch, effort and flow.



Figures 4.28 a - d Trends in (a) Trend (1983 – 2005) in commercial barramundi catch (t) and effort (100m net sets/day) between 1983 and 2005, (b) angler recreational 'Classic' barramundi catch (number fish) and effort (angler hrs) between 1985 and 2005, and (c) barramundi catch on recreational tours (number fish) and effort (line hrs) between 1994 and 2005. (d) Increasing trends in population size of barramundi in the Daly River between 1983 and 2005 as indexed by $\log_{10}$ CPUE from all three fisheries. The regression between tour CPUE and time is not significant (see text).

River flow, barramundi ecology & catch

The importance of river flow as a key driver of aquatic ecosystem dynamics is outlined in Section 4.1.1, particularly in relation to the 'Flood-Pulse Concept' (Bunn & Arthington 2002, Douglas et al 2005, Hamilton & Gehrke 2005). Flow is also critical for population level responses, such as the timing, intensity and direction of fish migration, and survival and fecundity related to habitat condition. Most likely flow interacts with other key environmental variables such as water temperature and light (Jonsson 1991).



Figures 4.29 a - d (a) Strong regression relationship between recreational angler ('Classic' competition) and commercial CPUE of barramundi in the Daly River (1985-2005). (b) Time trends in commercial catch (weight t, 1983-2005) and recreational Classic barramundi catch (umber fish caught, 1985-2005). (c) Lack of correlation between instantaneous total recreational catch [$\log_{10}$ (angler Classic+Tours)] and commercial catch ($\log_{10}$weight t) and (d) cross correlation correlogram between recreational and commercial catches showing no lagged effect. Dashed lines are 95% confidence intervals, and standard errors (3rd column left of figure) are white noise estimates.

The general positive correlation between environmental flow and commercial fisheries production is well known (eg see Griffin 1987, Sawynok 1998, & Staunton-Smith 2004 for barramundi; Glaister 1978 for prawns; Sutcliffe 1973 for American Lobster & Atlantic Halibut; Beamish et al 1994 for Pacific salmon & herring; & Loneragan & Bunn 1999 in general), although exact causal mechanisms are not (Robins et al 2005, & see Humphrey et al 2006 for negative flow relationships for fish in Magela Creek, Kakadu National Park). Lagged correlations between fisheries production (catch) and freshwater flows are often used to support the argument that flows affect the survival of fish during their first year of life and, hence, year-class strength (Stauton-Smith et al 2004). Robins et al (2005) undertook a comprehensive review of freshwater-flow requirements of estuarine fisheries in tropical

Australia, and suggested that a framework be adopted that integrates life-history knowledge with correlative flow-catch data to better understand and therefore manage freshwater-flow requirements of estuarine fisheries. There are many studies that demonstrate strong covariance between commercial marine and estuarine fisheries catch data and natural variations in freshwater flows (Lloret et al 2001, Quiñones & Montes 2001), often with time lags equalling the approximate age at which a species enters the fishery (Stauton-Smith et al 2004).

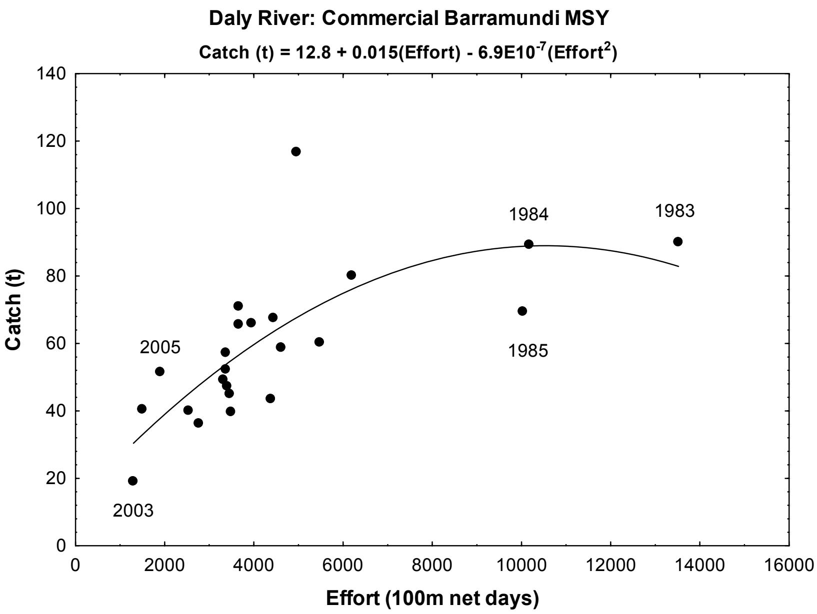
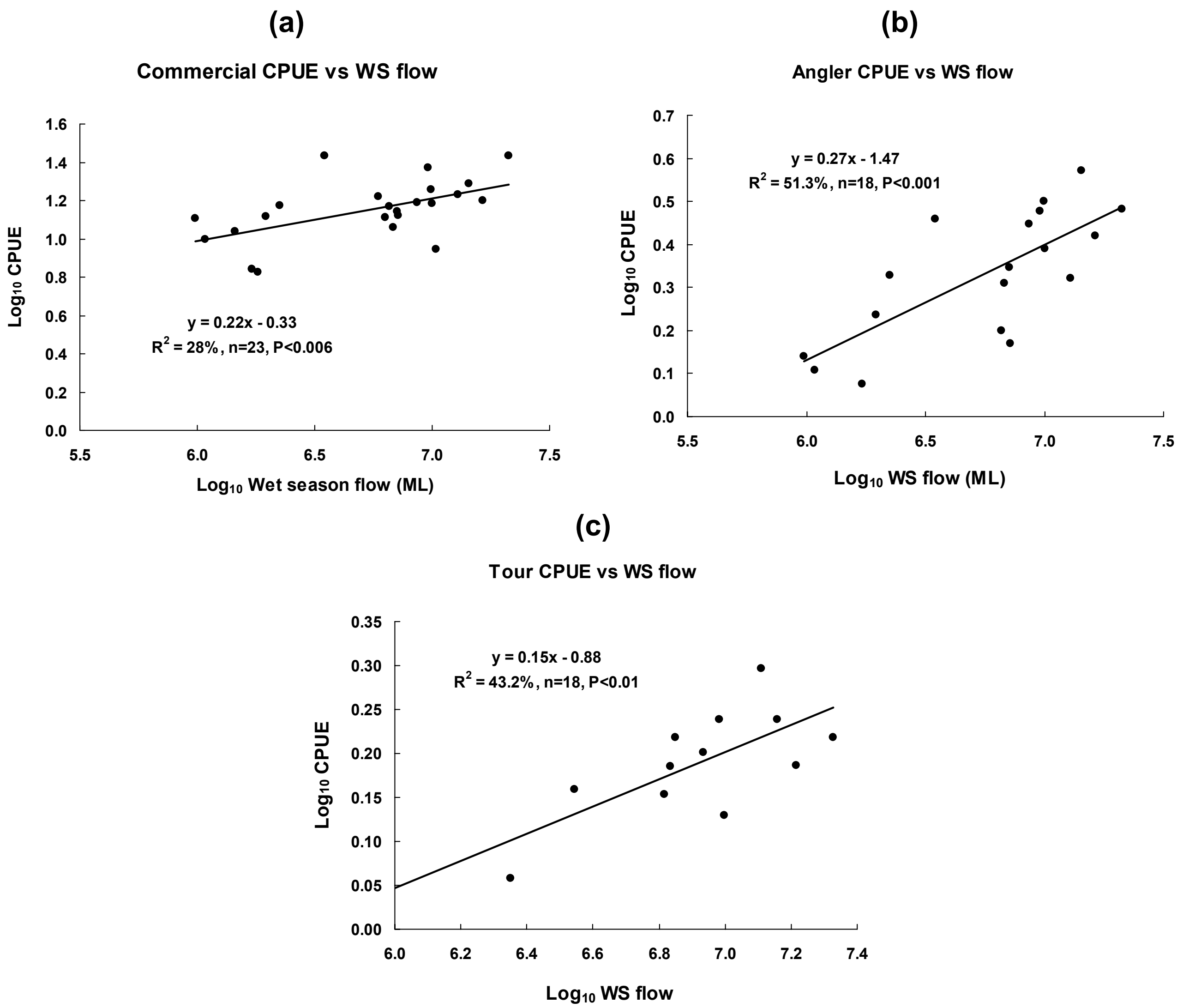


Figure 4.30 Catch (t) vs. effort (100m net days) plot used to estimate Maximum Sustained Yield (MSY) for the commercial barramundi fishery on the Daly River between 1983 and 2005. Years at MYS and above are indicated.

Stauton-Smith et al (2004) found positive correlations between the abundance of barramundi year classes and the amount of freshwater flowing in the Fiztroy River estuary in Queensland during spring and summer, when they spawn and young of the year recruit to nursery habitats. They used multiple regression analysis to explore relationships between year-class strength and environmental variables, and a similar approach is adopted here for catch and CPUE population indices. Additionally, Robins et al (2005) found that barramundi catch in the Fitzroy region in Queensland was correlated to freshwater flow.

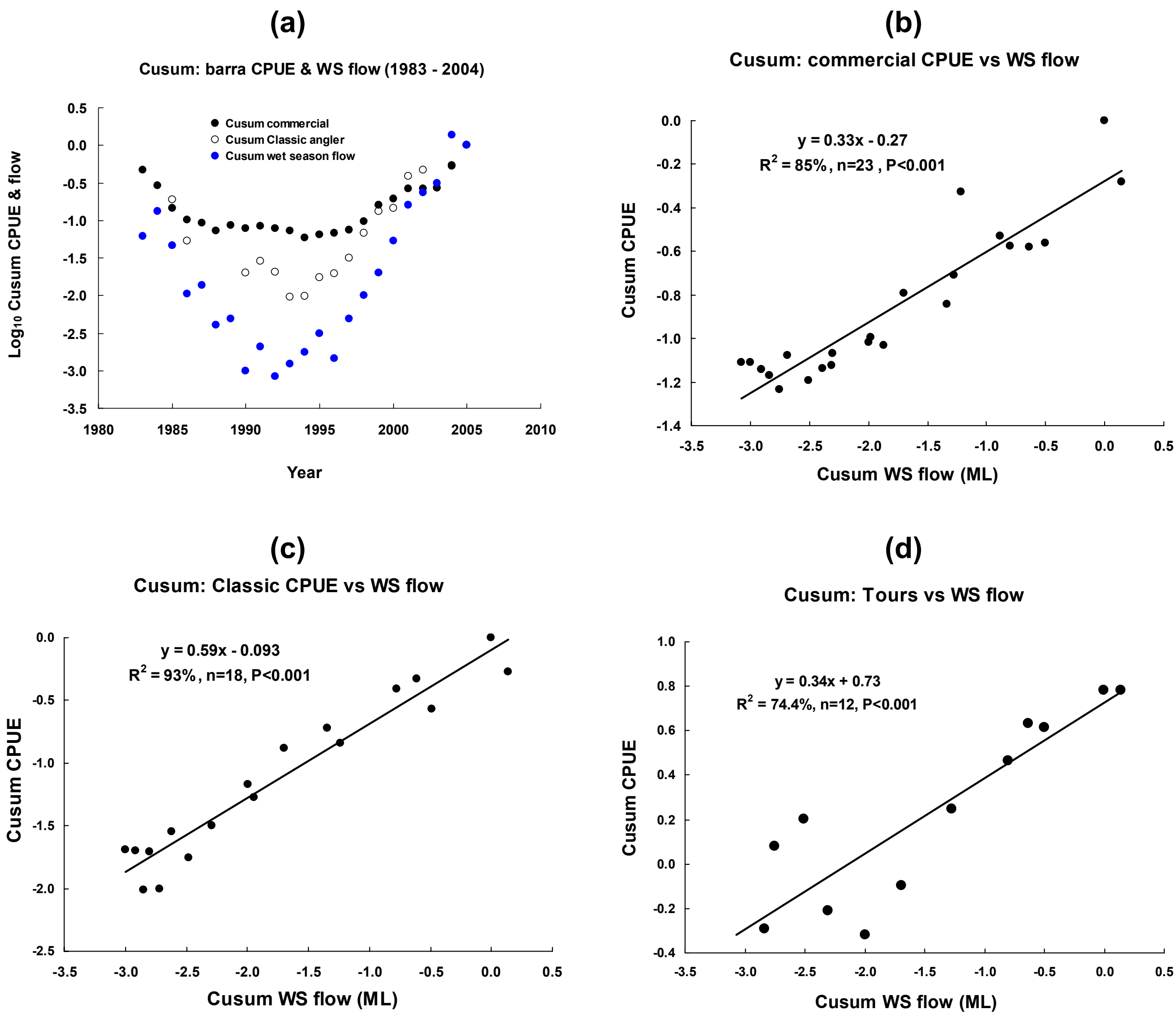
Regression analysis is used here to examine the relationship between barramundi abundance in the Daly River, as indexed by CPUE of the three fisheries between 1983 and 2005, in relation to natural wet season flow (October–April) at Mt Nancar gauging station, which is situated just above the tidal limit. In all analyses total annual flow (Sept. – Aug.) and dry season flow (May-Sept.) were examined also; however wet season flow always explained more variance. All barramundi CPUE population indices (Commercial, Tour & Classic) were positively correlated to wet season flow (Figure 4.31a-c, all data transformed to $\log_{10}$), possibly reflecting better survival and/or recruitment into the catchable size class as a function of present or previous flows enhancing habitat condition, and/or the fact that fish migrate more easily with increased flows.



Figures 4.31 a - c Regression relationship showing population size of barramundi (indexed by CPUE) increases with increasing wet season flow (ML), for the (a) commercial (1983-2005), (b) recreational angler 'Classic' (1985-2005) and (c) tour (1994-2005) fisheries. All data are transformed to $\log_{10}$.

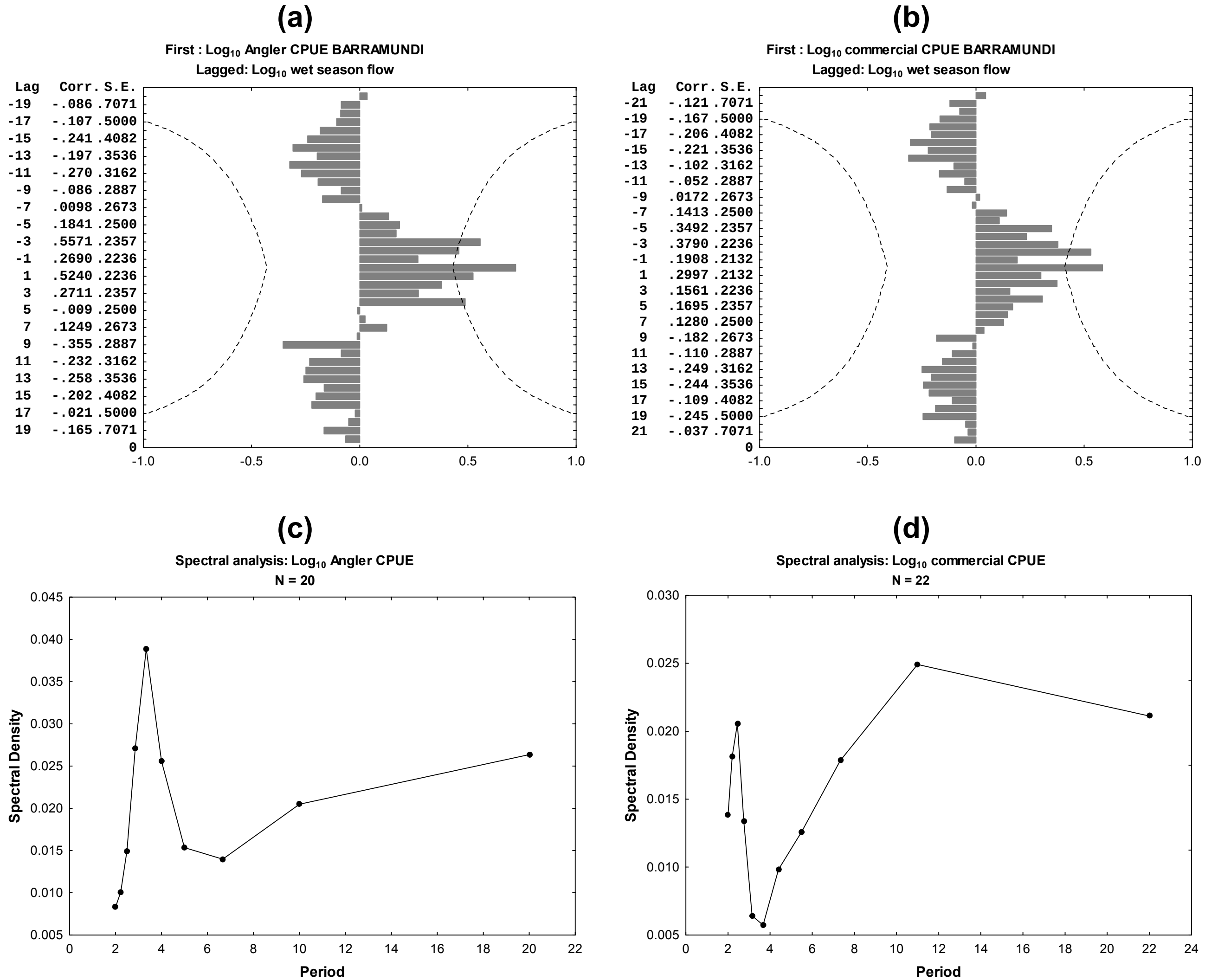
Cusum analysis showed that long-term flows for the Daly and Katherine rivers exhibited 22y periods (Section 4.1.1), and magpie goose populations in the NT were found to exhibit similar and concordant 22-23y periods (Section 4.2.1). Figure 4.32a plots concordant cusum values for Daly River flow and all three barramundi CPUE population indices between 1983 and 2005. Hence, changes in the abundance and/or catchability of barramundi faithfully track changes in flow, as demonstrated by the tight regression relationships between the cusum values of all CPUE population indices and cusum values in river flow (Figure 4.32 b-d, all data transformed to $\log_{10}$). On average, however, wet season flow ($\log_{10}$ ML) increased over the same time period (R^2=25.8%, n= 23, P<0.01), explaining the average increase in barramundi population abundance.

Nevertheless, use of CPUE as an index of population size to track relative trends assumes that the index is linearly related only to absolute abundance; that is, the underlying assumption is one of constant catchability. However, the catchability of barramundi in both commercial and recreational fisheries may increase with increased flow due to increased fish movements. A positive correlation between instantaneous catch and flow may encompass a significant



Figures 4.32 a - d (a) Concordant time trends in cusum values for Daly River flow (ML), commercial CPUE and recreational angler 'Classic' CPUE barramundi population indices. Strong regression relationships between cusum CPUE values and river flow for the (b) commercial, (c) angler 'Classic' recreational and (d) tour recreational fisheries. All data are $\log_{10}$ values.

component of increased catchability in addition to increased recruitment and/or survival effects. In contrast, a positive correlation at 2-3 year time lags suggest enhanced recruitment and/or survival of a cohort that has reached the size limit to enter the fishery (currently 55cm total length in commercial & recreational barramundi fisheries). Cross-correlation correlograms between recreational Classic barramundi CPUE and wet season flow, and between commercial barramundi CPUE and wet season flow (all values transformed to $\log_{10}$ & detrended only), show that significant positive correlations occur at 0, 2 and 3 year time lags (Figure 4.33a), and 0 and 2 year time lags (Figure 4.33b), respectively. A periodogram of spectral density of angler Classic CPUE versus period (Figure 4.33c) shows a sharp peak at 3 y, possibly reflecting the regular entry of 3y old fish into the fishery, followed by a gradual rise towards the 20 y period, possibly reflecting a tendency towards the 20 y flow period detected in longer (ie > 20 y) flow data. Similarly, a Periodogram of spectral density of commercial CPUE versus period (Figure 4.33d) shows a sharp peak in the interval 2-3 y,



Figures 4.33 a - d (a) Cross correlation correlogram between angler 'Classic' recreational barramundi CPUE ($\log_{10}$) and wet season flow ($\log_{10}$ ML) and (b) similarly for commercial CPUE. Both correlograms show significant positive correlations with flow with a zero and 3 year time lag. Periodograms of spectral density of barramundi population size as indexed by (c) angler 'Classic' CPUE and (d) commercial CPUE, both showing peaks at 2-3y and 10y and 20y, concordant with period trends in Daly River flow. Dashed lines are 95% confidence intervals, standard errors (3rd column left of figure) are white noise estimates.

possibly reflecting the regular entry of 2-3 year old fish into the fishery, a noticeable peak at 11y, and a high but slightly lower spectral density at 22y. As with angler CPUE, the latter period may reflect the 22y flow period. Currently there is no explanation for the 11y period trend in commercial barramundi catch, although other landscape and population processes also exhibit this period (eg fire extent, Bayliss et al in prep. c).

4.3.2 Conceptual model for in-stream health

The CPUE-flow analyses above suggests that barramundi catch *per s*e and flow are also strongly correlated, and this is supported by further analyses (see Section 4.3.3 below). We use the strong relationship between barramundi catch and flow to indirectly predict potential tradeoffs between reduced flow from extractions and lost fisheries value in terms of either revenue (commercial & recreational) or some intangible benefit (eg recreational catch). Benefit-cost analysis can therefore be used to determine optimal allocation of competing resources, such as flow, but requires knowledge on how the demand function shifts with changes in flow or flow-related variables such as fish catch (Loomis & Cooper 1990).

Two assessment endpoints for barramundi are used to assess in-stream health; catch and population size as indexed by CPUE. Whilst both endpoints are related they have two entirely different social contexts about river values. Barramundi catch, either commercial or recreational, is a key socio-economic endpoint that, at the end of the day, may contribute to river protection through allocation of environmental flows. However, fishing is also a threat if not sustainably managed and, hence, this caveat needs to be attached to any benefit-cost analysis. In contrast, the size of barramundi populations is an ecological assessment endpoint and would treat fishing as a threat in addition to other threats such as reduced river flow. Both assessment endpoints simply reflect different values being assessed, yet both would contribute to an overall assessment of in-stream health. There is strong interaction between the two values and, ideally, they should be combined into an overall assessment. However, they are treated as two independent assessments here because they both depend on the same data (catch & effort). This is currently not an issue as barramundi stocks in the Daly River are estimated to be about 15-20% less than unfished stocks, well above the stock level associated with estimates of maximum sustained-yield (MSY, NT Fisheries). However, recreational effort and barramundi catch in the Daly River are increasing on average (Figure 4.28 b & c), whilst flow and hence barramundi population size are predicted to decease over the next decade (Section 4.3.1). Therefore, the two assessments of in-stream health need to be combined in future assessments using population data independent of the fishery (ie of traditional catch-effort & CPUE statistics).

Two conceptual models were therefore constructed to guide assessment of Daly River in-stream health under different wet season flow extraction scenarios; the first uses catch as a socio-economic assessment endpoint (Figure 4.34a), and the second uses an index of population abundance as an ecological assessment endpoint (Figure 4.34b). Stochastic process models were developed from the conceptual models (Sections 4.3.3 & 4.3.4), which were then incorporated into Bayesian Networks (BN) to examine the tradeoffs between barramundi catch, population size and simulated flow extractions (Sections 4.3.5 to 4.3.7).

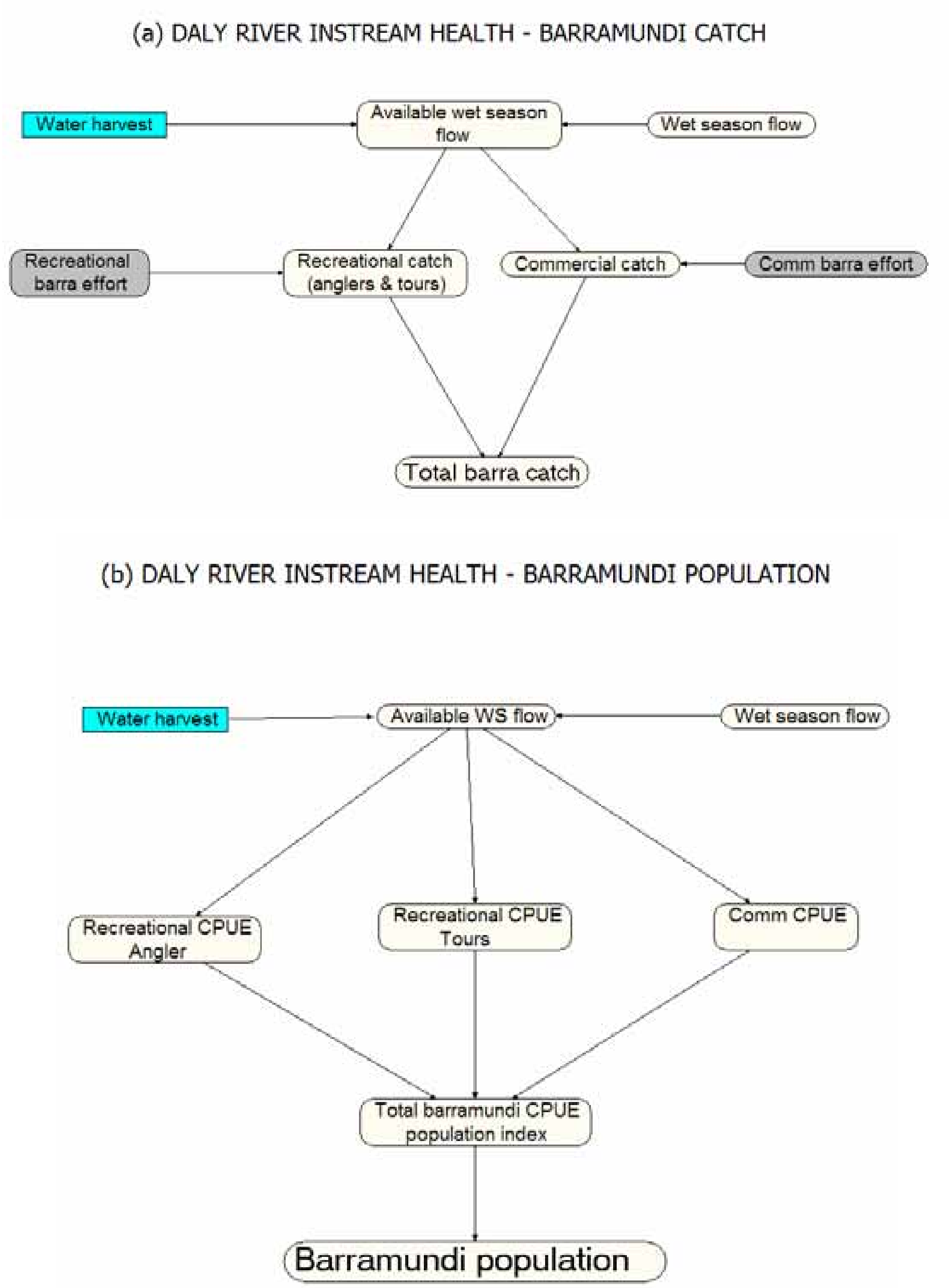


Figure 4.34 a & b Conceptual models used to construct Bayesian Networks to assess the in-stream health of the Daly Rive under different wet season flow extraction scenarios with respect to: (a) a social assessment endpoint (commercial & recreational barramundi catches); and (b) an ecological assessment endpoint (population size of barramundi as indexed by CPUE).

4.3.4 Ecological models – barramundi catch, population size and river flow

Relationships between catch, effort and river flow

Multiple regression analysis was used to examine the relationship between catch as a function of effort and flow. The effects of flow and effort in the regression analysis are assumed mostly independent because any intercorrelation would be 'partialled-out'. Commercial barramundi catch was related to both effort and flow (Table 4.14), and the combined regression model explained 67% of observed data. Beta coefficients indicate that effort had 3 times more influence on catch than flow. Recreational Classic barramundi catch was also strongly related to the combined effects of effort and flow (Table 4.15), explaining 78% of observed data. Beta coefficients indicate that effort had 1.5 times more influence on catch than flow, half that for commercial catch. The relationship between total recreational catch (numbers of fish caught) and total recreational effort and flow were examined by combining recreational Classic effort (angler hours) and Tour effort (line hours), which assumes that they are equivalent. This index of total recreational barramundi catch was highly related to effort and flow, with the combined regression model explaining 94% of observed data (Table 4.16). Beta coefficients of the regression model suggest that combined recreational effort had 2.5 times more influence that flow, similar to commercial catches.

Table 4.14 Summary of the multiple regression between commercial barramundi catch ($\log_{10}$ weight t) on effort ($\log_{10}$ CE 100m net sets/day) and wet season flow ($\log_{10}$ WSQ ML). Daly River (1983-2005), n = 23, see data summary in Appendix 8.3.1 and 8.3.2.

R= 0.8149, adjusted R^2 = 67%, n= 23, P< 0.001, SE regression = 0.099

Variable	Beta	SE Beta	B	SE B	P
Intercept			-1.256	0.608	=0.052
WSQ	0.308	0.140	0.127	0.058	=0.040
CE	0.881	0.141	0.597	0.095	<0.001

Table 4.15 Summary of the multiple regression between recreational Classic barramundi catch ($\log_{10}$ numbers of fish caught), effort ($\log_{10}$ ARE angler 'Classic' hours) and wet season flow ($\log_{10}$WSQ ML). Daly River (1985-2005), n = 18 (some years are missing), see data summary in Appendix 8.3.1 and 8.3.2.

R= 0.8821, adjusted R^2 = 78%, n= 18, P< 0.001, SE regression = 0.204

Variable	Beta	SE Beta	B	SE B	P
Intercept			-8.397	1.836	<0.001
WSQ	0.392	0.159	0.390	0.158	=0.026
Classic ARE	0.576	0.159	2.299	0.635	<0.001

Table 4.16 Summary of the multiple regression between total recreational barramundi catch (log_{10} numbers of fish caught) on effort (log_{10} TRE Tour line hours + angler Classic hours) and wet season flow (log_{10}WSQ ML). Daly River (1985-2005), n = 18 (some years are missing), see data summary in Appendix 8.3.1 and 8.3.2.

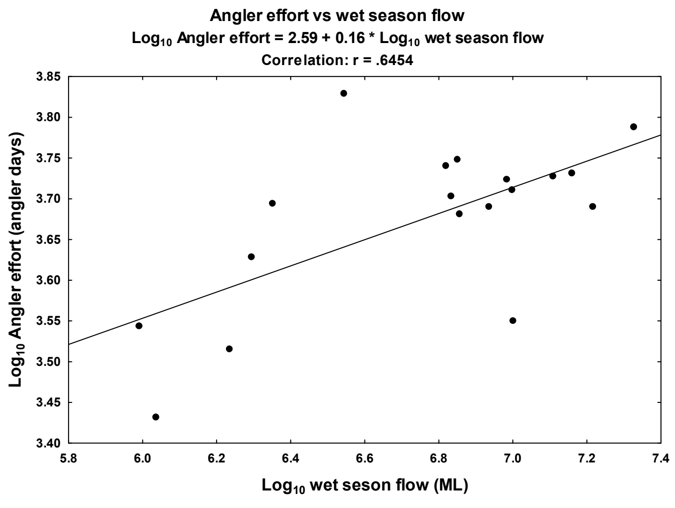
R= 0.9689, adjusted R^2 = 94%, n= 18, P< 0.001, SE regression = 0.167

Variable	Beta	SE Beta	B	SE B	P
Intercept			-8.538	0.772	<0.001
WSQ	0.309	0.080	0.484	0.126	=0.002
TRE	0.749	0.080	2.174	0.234	<0.001

Relationship between effort and flow

The regression models above assume that effort does not increase or decrease as a function of flow via easier or more difficult access, respectively. Any variation in catch is assumed to be simply a function of variation in fishing effort *per se* and variation in numbers of catchable barramundi in the legal size class. Commercial effort was negatively correlated to flow and, in contrast, recreational effort was positively correlated to flow (all variables transformed to log_{10}; commercial: R= -0.3827, n=23, P=0.073; recreational: R= +0.6454, n=18, P=0.004).

Effort in the commercial barramundi fishery decreased on average over time (see Figure 4.28a) because of reduced catches and management intervention in 1989. This trend, however, has been coincidental to increased trends in both wet season flow and recreational effort over the same period (see Figure 4.28 b & c) via the number of Classic anglers and angler hours fished, and line hours of Tours. However, a GLM model that incorporated an interaction term for effort and flow showed that it is redundant, explaining no additional variation above that captured by the independent effects of flow and effort. It is assumed that the positive correlation between angler Classic effort and flow is coincidental (Figure 4.35), and that the partial multiple regression analysis method used above factors out (or partials out) the intercorrelation between flow and effort, relegating possible interactions to the residual regression error term.



Figures 4.35 Regression relationship between angler 'Classic' effort (log_{10}angler hrs) and wet season flow (log_{10} ML).

Relationship between recreational and commercial catches

Analyses presented in Section 4.3.1 indicate that there is no interaction between recreational and commercial catches (see Figs. 4.29b-d), although the same data have been used to suggest otherwise (eg DBIRD 2003 for closure of Bynoe Harbour commercial barramundi fishing). Hence, further examination of data is warranted. A multiple regression model was used to examine the multiple working hypotheses that recreational angler catch is influenced by wet season flow and angler effort, as in the previous model, but in addition commercial barramundi effort and catch. The overall regression model explains 89% of observed data and all variables were significant entries into the equation (Table 4.17). As expected, angler catch increased with increasing flow. However, results show also that whilst recreational catch increased with decreasing commercial effort, recreational catch increased with increasing commercial catch. Taken together, the results suggest that river flow and the individual efforts of each fishery are the main drivers for their respective catches, and not any strong cross-over interaction in effort between the two fisheries.

Table 4.17 Summary of the multiple regression between recreational Classic barramundi catch ($\log_{10}$ numbers of fish caught) and wet season flow ($\log_{10}$WSQ ML), angler effort ($\log_{10}$ARE angler hrs), commercial effort ($\log_{10}$CE 100m net sets/day) and commercial catch ($\log_{10}$CC weight t). Daly River (1985-2005), n = 18 (some years are missing), see data summary in Appendix 8.3.1 and 8.3.2.

R= 0.9434, adjusted R^2 = 89%, n= 18, P< 0.001, SE regression = 0.154

Variable	Beta	SE Beta	B	SE B	P
Intercept			-8.380	2.606	=0.007
WSQ	0.298	0.123	0.296	0.122	=0.031
ARE	0.396	0.166	1.578	0.664	=0.033
CE	-0.482	0.215	-0.981	0.439	=0.043
CC	0.569	0.163	1.429	0.411	=0.004

Relationship between population abundance (CPUE) and river flow

All three CPUE indices of barramundi population size were highly correlated, hence only recreational Classic CPUE (number of fish caught/angler hr) data are analysed here and in subsequent analyses and modelling. Regression analysis shows that the CPUE index of barramundi population size was positively correlated to wet season flow explaining 51% of observed data (Table 4.18).

Table 4.18 Summary of the multiple regression between recreational Classic CPUE ($\log_{10}$ ACE barramundi caught/angler hr) and wet season flow ($\log_{10}$ WSQ ML). Daly River (1985-2005), n = 18 (some years are missing), see data summary in Appendix 8.3.1 and 8.3.2.

R= 0.8149, adjusted R^2 = 51%, n= 18, P< 0.001, SE regression = 0.104

Variable	B	SE B	P
Intercept	-1.471	0.415	=0.003
WSQ	0.267	0.061	<0.001

4.3.5 Water extraction simulation, model uncertainties & sensitivity analyses

Catch

The stochastic process model used to predict the impact of simulated wet season flow extractions (0–100%) on barramundi catch (see regression equations in Tables 4.14 – 4.16) is conceptually illustrated in Figure 4.36 and shows all model uncertainties. The frequency distribution of observed flow data ($\log_{10}$WSQ ML) during the fishing period and used in the regression model for commercial catches ($\log_{10}$Weight t) is best described by a Triangular probability density function (pdf, Figure 4.37a & b) and, that for commercial effort ($\log_{10}$ 100m net sets/day) a Logistic pdf (Figure 4.37c & d). The frequency distribution of observed recreational Classic effort (angler hrs) data during the fishing period and used in the regression model to predict Classic catch is best described by a Logistic pdf (Figure 4.38a & b) and, similarly, for total recreational effort (Classic angler hrs + Tour line hrs, Figure 4.38c & d).

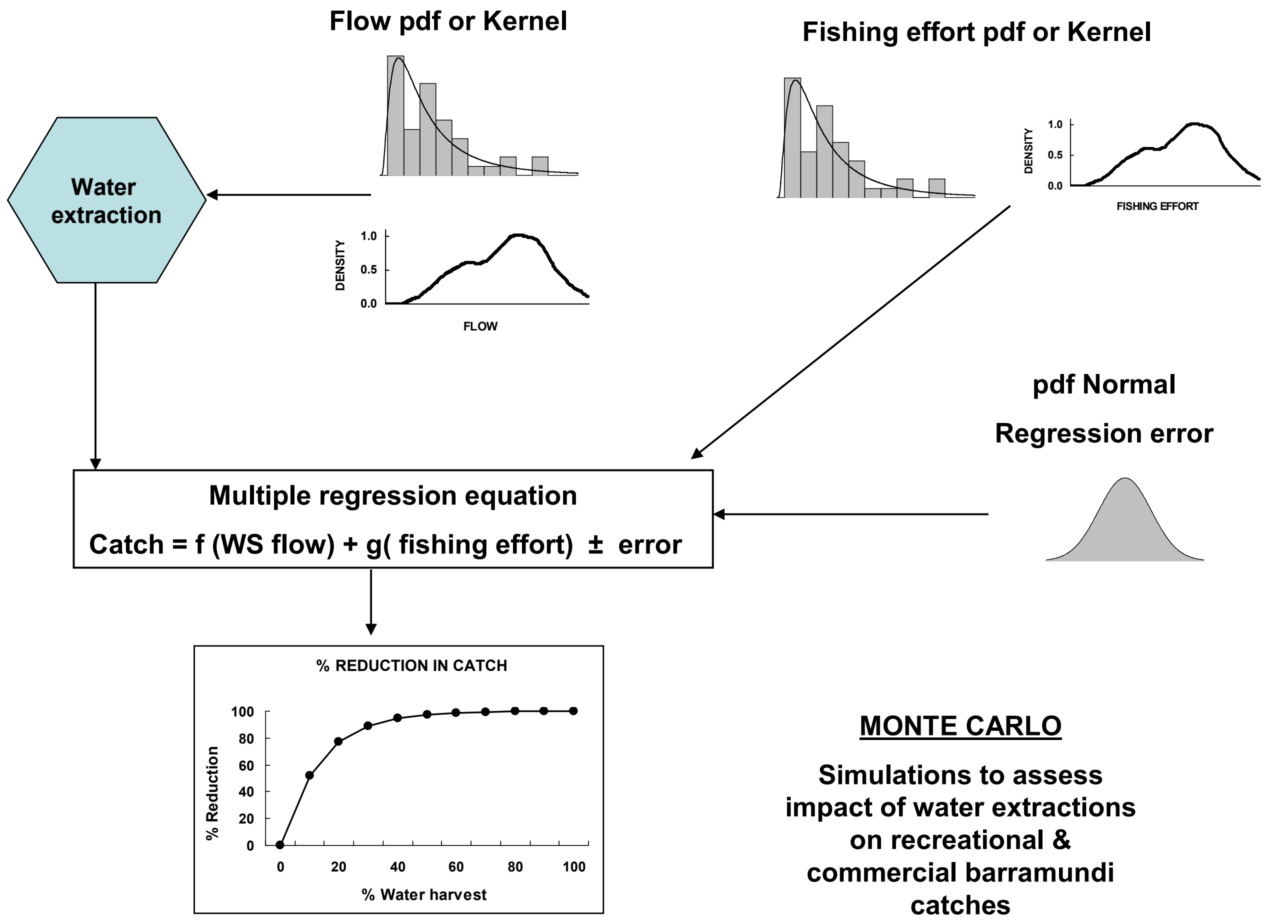
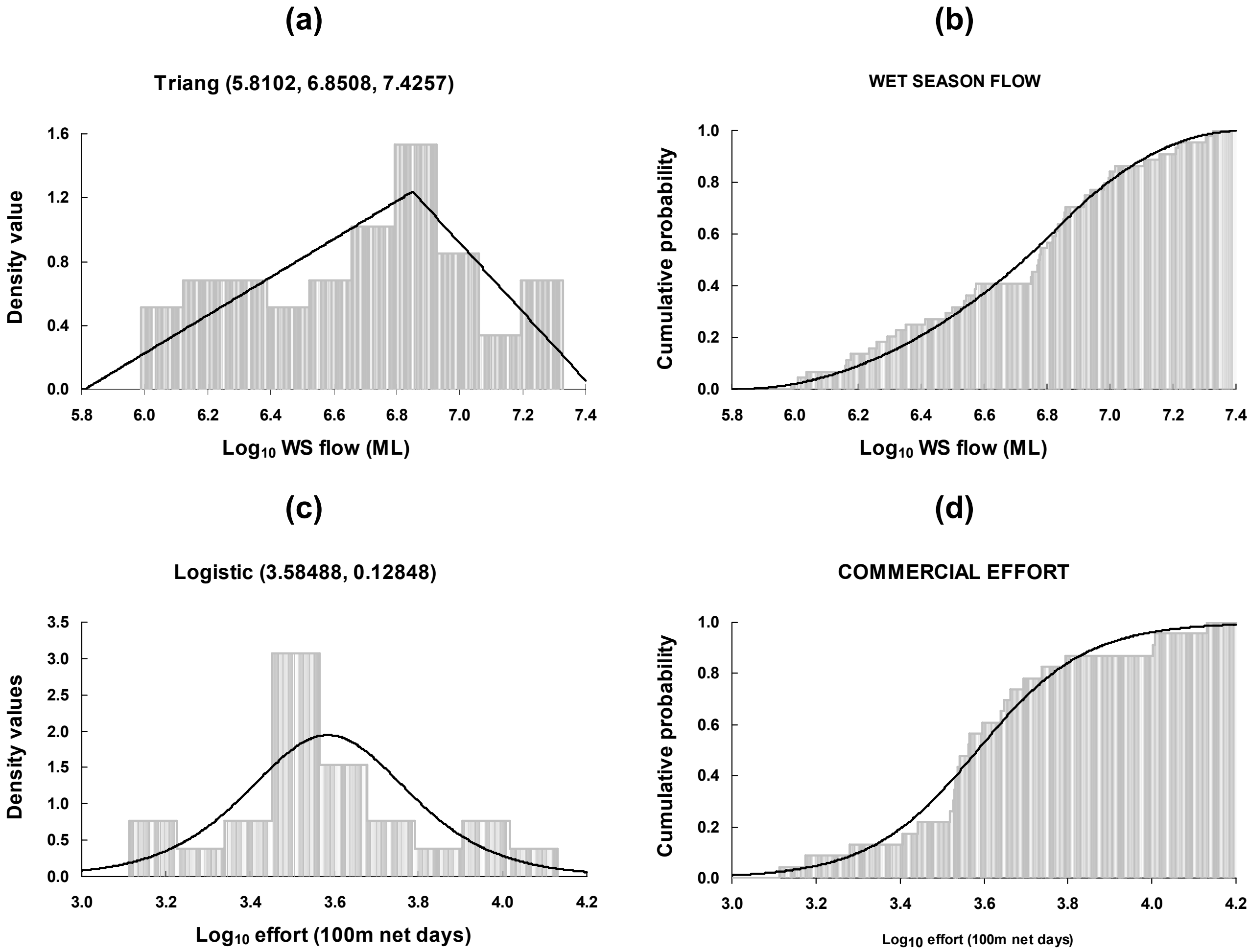


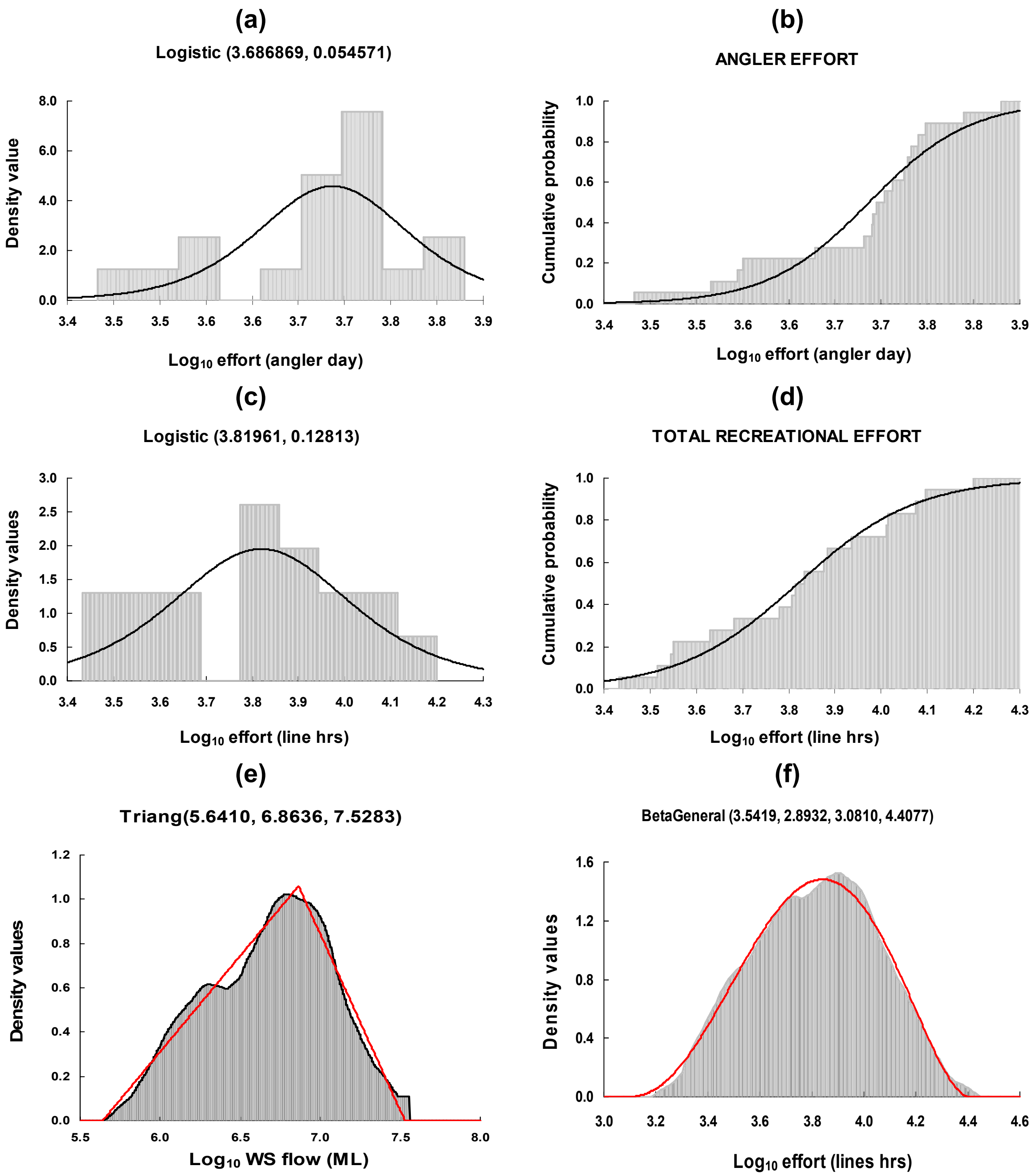
Figure 4.36 Stochastic process sub-model of the simulated impact of wet season flow extraction (0-100%) on commercial and recreational barramundi catch in the Daly River showing all model uncertainties. The statistical model was used to support the Bayesian Network for In-stream Health using a social endpoint. The multiple regression equation was developed from observed NT fisheries data to predict catch based on effort and current wet season flow.



Figures 4.37 a - d Statistical distributions fitted to observed data used in the regression equation (& subsequent Bayesian Network) to predict barramundi catch as a function of wet season flow and fishing effort over the fishing period. (a) Probability density function (pdf, Triangle) and (b) cumulative probability curve for wet season flow ($\log_{10}$ ML). (c) Probability density function (pdf, Logistic) and (d) cumulative probability curve for commercial barramundi fishing effort ($\log_{10}$ 100m net sets/day).

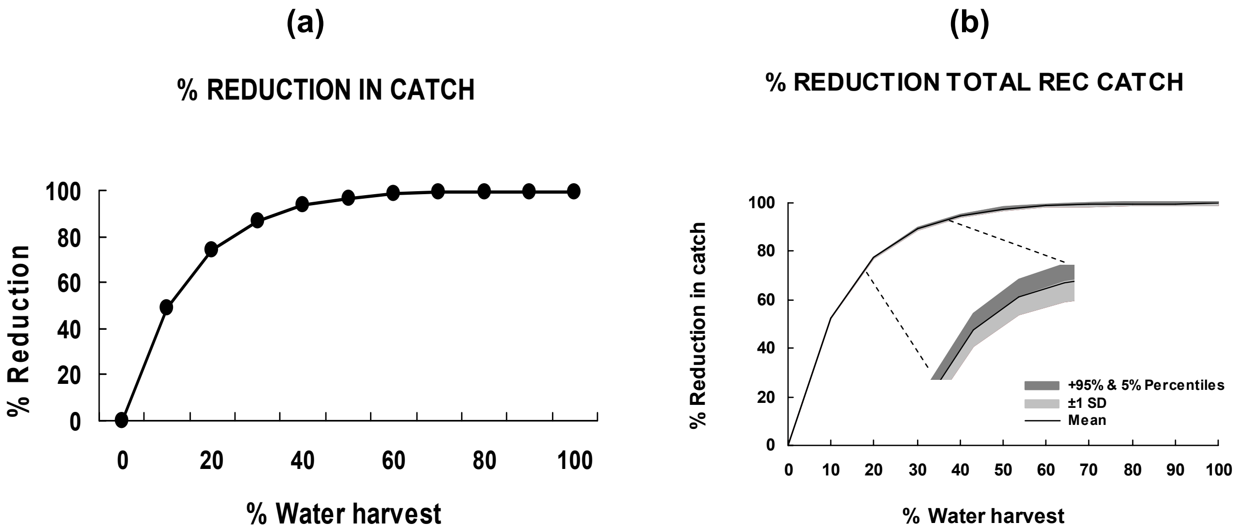
As outlined in Section 4.1.2, the method of Wu and Tsang (2004) was trialled for barramundi catch-effort data sets by replacing often poorly fitted pdfs with Kernel density smoothing functions, to compare with Monte Carlo simulation using pdfs fitted from a standard array of statistical distributions. Kernel distributions were fitted to observed data for flow ($\log_{10}$WSQ ML) and total recreational effort ($\log_{10}$ hrs fished), and show that standard statistical distribution models compare reasonably well (Figure 4.38e & f, respectively). Whilst the Logistic fit for total recreational effort was replaced with a Beta General distribution, Goodness of Fit tests (Pallisade 2002a) indicate that they are closely ranked. Monte Carlo simulation results using Kernel distributions or 'best fit' statistical distributions should, therefore, produce very similar results and is demonstrated below.

Mean values were derived by Monte Carlo (MC) simulation (10,000 iterations) using @Risk™ software (Pallisade 2002b), and uncertainty analysis incorporated both intrinsic variability in model parameters and overall model error as outlined in Figure 4.36. Simulation results for commercial, angler Classic and Tour catches are similar and, hence, only results for total recreational catch are illustrated here. Simulations predict that the percentage



Figures 4.38 a - f Statistical distributions fitted to observed data used in the regression equation (& subsequent Bayesian Network) to predict barramundi catch as a function of wet season flow and fishing effort over the fishing period. (a) Probability density function (pdf, Logistic) and (b) cumulative probability curve for angler 'Classic' effort ($\log_{10}$ angler hrs). (c) Probability density function (pdf, Logistic) and (d) cumulative probability curve for total recreational barramundi fishing effort ($\log_{10}$ angler hrs + line hrs). (e) Closely matched smoothed Kernel and Triangle density functions (see Silverman 1990) for wet season flow ($\log_{10}$ Q ML). (f) Similarly, a smoothed Kernel distribution function and a Beta General pdf are closely matched for total recreational effort ($\log_{10}$ line hrs).

reduction in mean total recreational catch will rapidly increase as the proportion of wet season flow extraction is increased, but will ameliorate and asymptote at 100% or zero catch (Figure 4.39a). The model predictions are highly certain because of low model error (Figure 4.39b). Sensitivity analysis (via the regression method) for commercial and angler Classic catches indicate that model outputs were more influenced by effort and model error than flow, whilst that for total recreational catch was more influenced by effort and flow than model error (Figure 4.40a-c Tornado graphs). These results are obvious in that they basically reflect the regression coefficients and the amount of explained variance of each regression model. Nevertheless, additional model outputs from the MC simulations indicate results are generally coherent with observed data. For example, the pdf of actual commercial barramundi catch is best described by a Gamma distribution (Figure 4.41a) and is similar to the distribution of predicted catches including model error (Figure 4.41b). Similarly, the pdf of actual angler Classic barramundi catch is best described by a Weibull distribution (Figure 4.41c) and is similar to the distribution of predicted catches including model error (Figure 4.41d).



Figures 4.39 a & b (a) Simulated reduction in mean total recreational barramundi catch as a function of wet season flow extraction (%) using the simple regression equation catch (number caught) vs. flow (ML) and effort (angler hrs) summarised in Table 4.16. Mean values were derived by Monte Carlo simulation (10,000 iterations) using @Risk software (Pallisade 2002b) and incorporated model uncertainty as outlined in Figure 4.36. (b) As for (a) but with small uncertainty levels barely registering. Insert is magnified x50 and illustrates one standard deviation (SD) about the mean trend and the + 95% and – 5% percentiles for the 90%-100% reduction in catch range.

Population abundance

The stochastic process model used to predict the impact of simulated wet season flow extractions (0-100%) on barramundi population size as indexed by recreational Classic CPUE (see regression equation in Table 4.18) is conceptually illustrated in Figure 4.42, and shows all model uncertainties. The frequency distribution of observed flow data ($\log_{10}$WSQ ML) during the fishing period used in the regression model has been described in the previous section on catch.

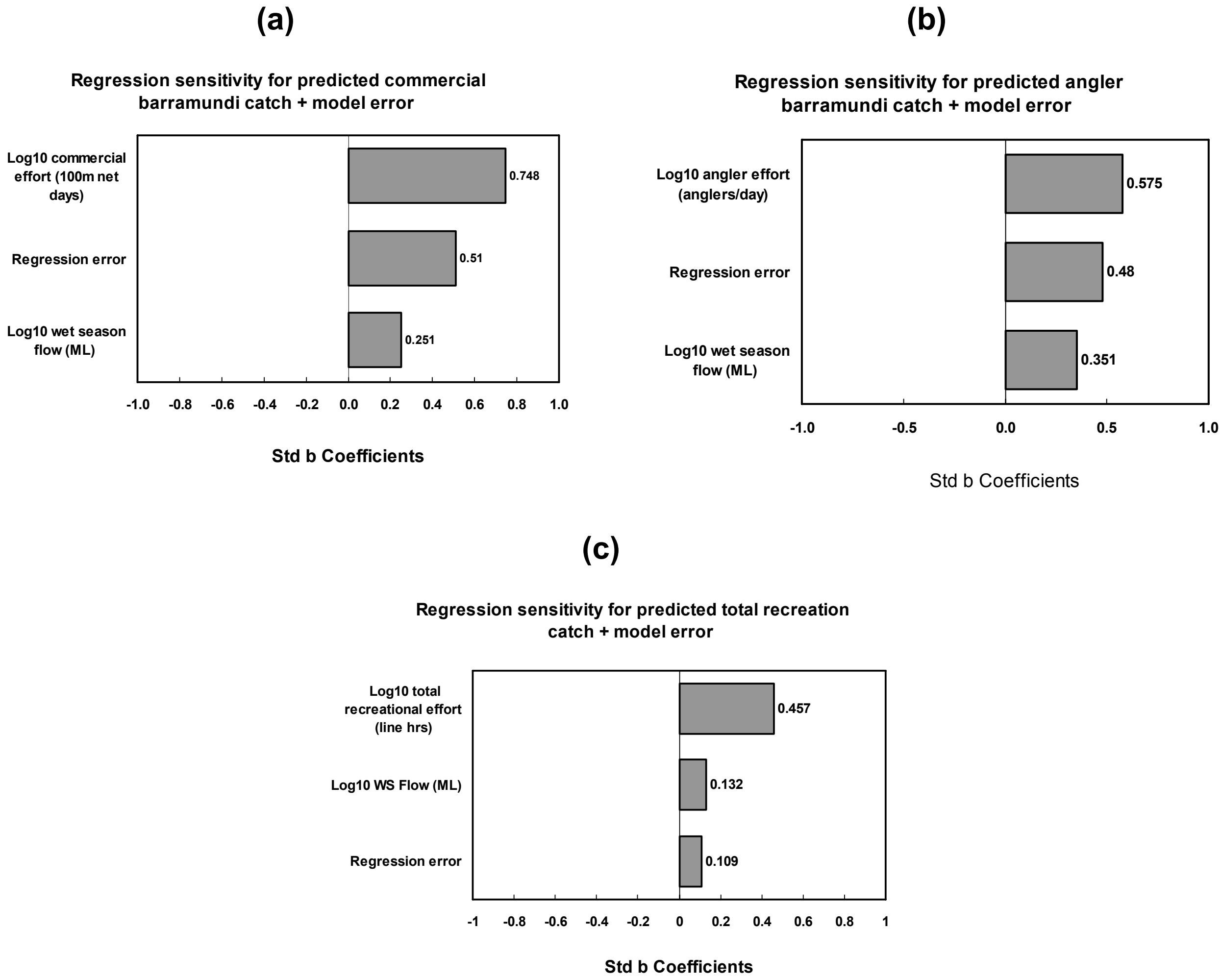
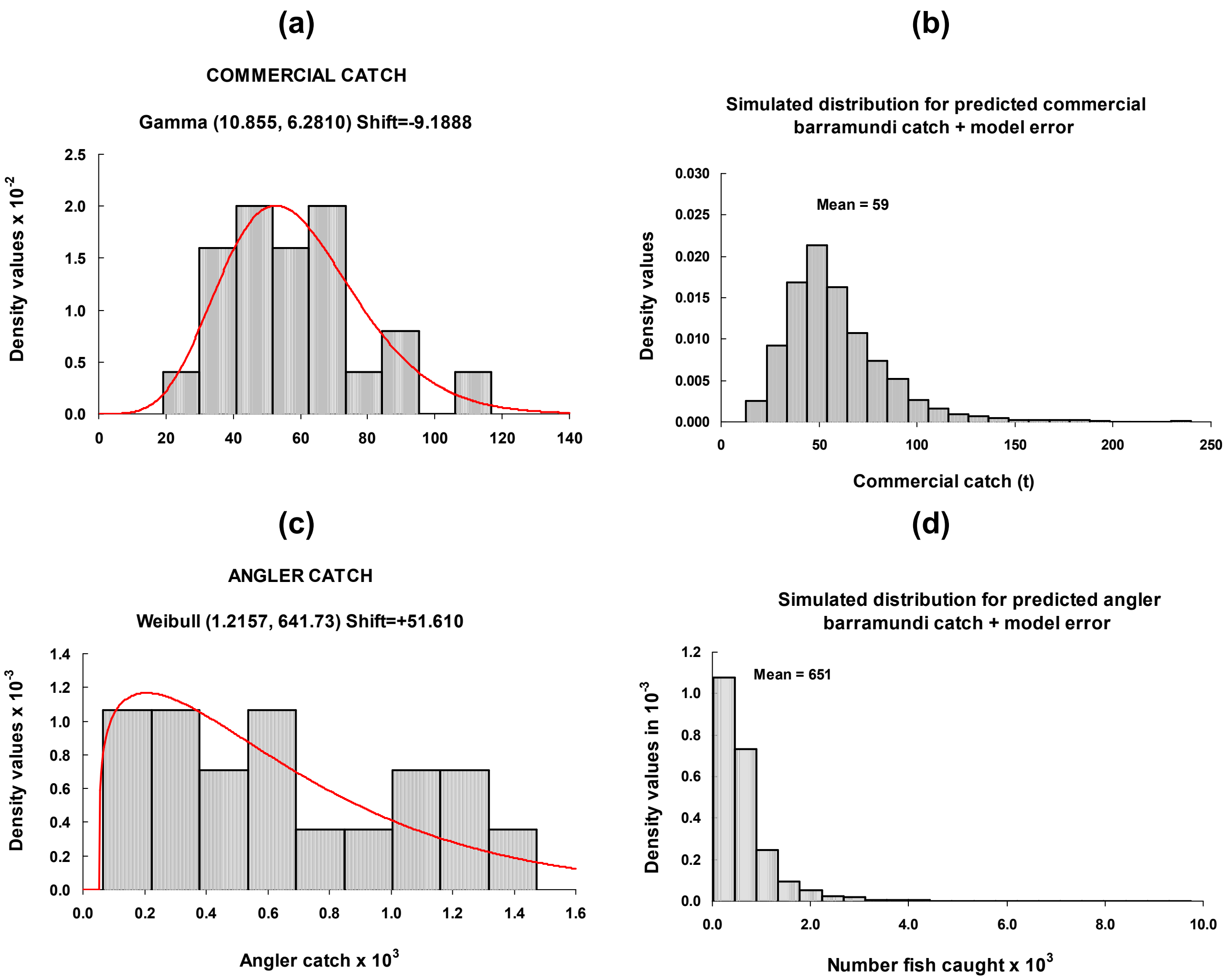


Figure 4.40 a – c Tornado graphs summarising sensitivity analyses of variable inputs into the regression equations predicting by Monte Carlo simulation barramundi catch on flow and effort, showing that (a) commercial effort contributed most to simulated model outputs, followed by regression error and then flow, (b) similarly for angler catch and (c) that for total recreational catch effort contributed most, followed by flow and then regression error.

Mean values were derived by Monte Carlo simulation (10,000 iterations) using @Risk™ software (Pallisade 2002b), and uncertainty analysis incorporated both intrinsic variability in wet season flow and overall model error as outlined in Figure 4.42. Mean simulation results predict that barramundi populations in the Daly River (here adjusted for a 15% reduction in stock due to current harvest levels) will linearly decrease as the percentage of wet season flow extraction is increased (0-100%, Figure 4.43a). However, the rate of decrease is not directly proportional to water extraction, being proportionally greater at smaller extraction levels. For example, a 20% simulated wet season flow extraction will reduce barramundi populations by 32%, and that for a 50% flow extraction by 58%. The model predictions are highly certain at low levels of percentage flow extractions and increase with the level of extraction (Figure 4.43b). The distribution of simulated barramundi population abundance indices for 20% and 50% wet season flow extraction scenarios are illustrated in Figure 4.43c.

In summary, both barramundi catch and population size appear highly sensitive to flow extraction; the greater the extraction rate the greater the negative impacts on both barramundi socio-economic and ecological assessment endpoints.



Figures 4.41 a - d Comparison of the observed distribution of (a) commercial catches with the (b) catches derived by Monte Carlo simulation and, similarly, for (c) observed distribution of angler 'Classic' catches and (d) simulated angler 'Classic' catches.

4.3.6 Bayesian Network for in-stream health based on barramundi catch

A Bayesian Network (BN) for in-stream health was constructed that incorporates commercial and total recreational barramundi catches as socio-economic assessment endpoints. An initial BN was constructed that used variable ranges for flow, effort and catch converted to state levels (Low, Medium & High), and follows procedures to develop Bayesian Networks recommended by Cain (2001). The associated probabilities of each state level entered in the Conditional Probability Tables (CPT) depended on the form of their pdf as determined by Best Fit™ (Pallisade 2002a) and an examination of natural breaks in the data (Table 4.19). However, this approach also proved unsatisfactory for similar reasons as for magpie geese. The identification of three state levels in the distribution of observed data was generally arbitrary, and this was accompanied by the necessity to populate large CPTs involving much guess work and creative invention. The approach adopted for magpie geese was therefore used here: large unwieldy CPTs of intersecting child nodes were avoided by replacing them

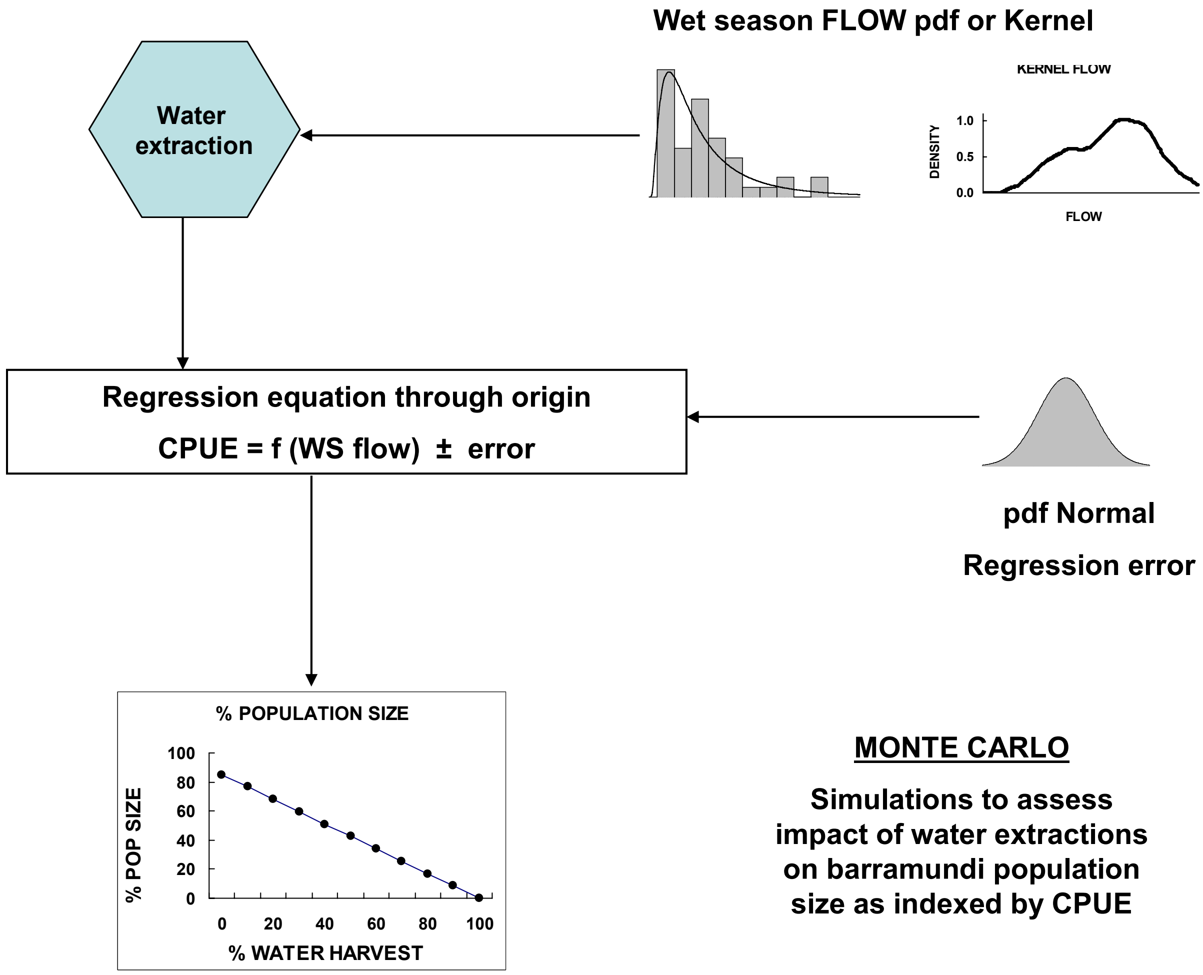


Figure 4.42 Stochastic process model (multiple regression equation) used to simulate the impact of wet season flow extractions (0-100%) on barramundi population size as indexed by CPUE (see text). The statistical model was used to support the Bayesian Network for In-stream Health using an ecological assessment endpoint.

with equations that used outputs (other equations, pdfs or constants) from parent nodes as input variables. Hence, the predictive stochastic models developed above to simulate the effects of flow extraction on both commercial and total recreational barramundi catches were combined into the one BN. Additionally, rates of water extraction and fishing efforts can be pinpointed to simulate different management scenarios, rather than assigned to a broad level.

Two simulation scenarios are illustrated here for the recreational barramundi fishery; no flow extraction and a 20% flow extraction. Commercial effort is set to the lowest level recorded in 2005 and, hence, their catch is set to 'Poor' in a socio-economic sense. Note that fish catch assessment endpoints were kept separate to reflect the fact that the commercial barramundi fishery in the Daly River reach is effectively closed. Recreational effort (angle hrs) and wet season flow (ML) are characterised by their pdfs, hence inputs into subsequent child nodes and their outputs will encompass the full range of effort and flow conditions during operation of the recreational fishery. State levels of 'Poor, Ok and Excellent' for total recreational and

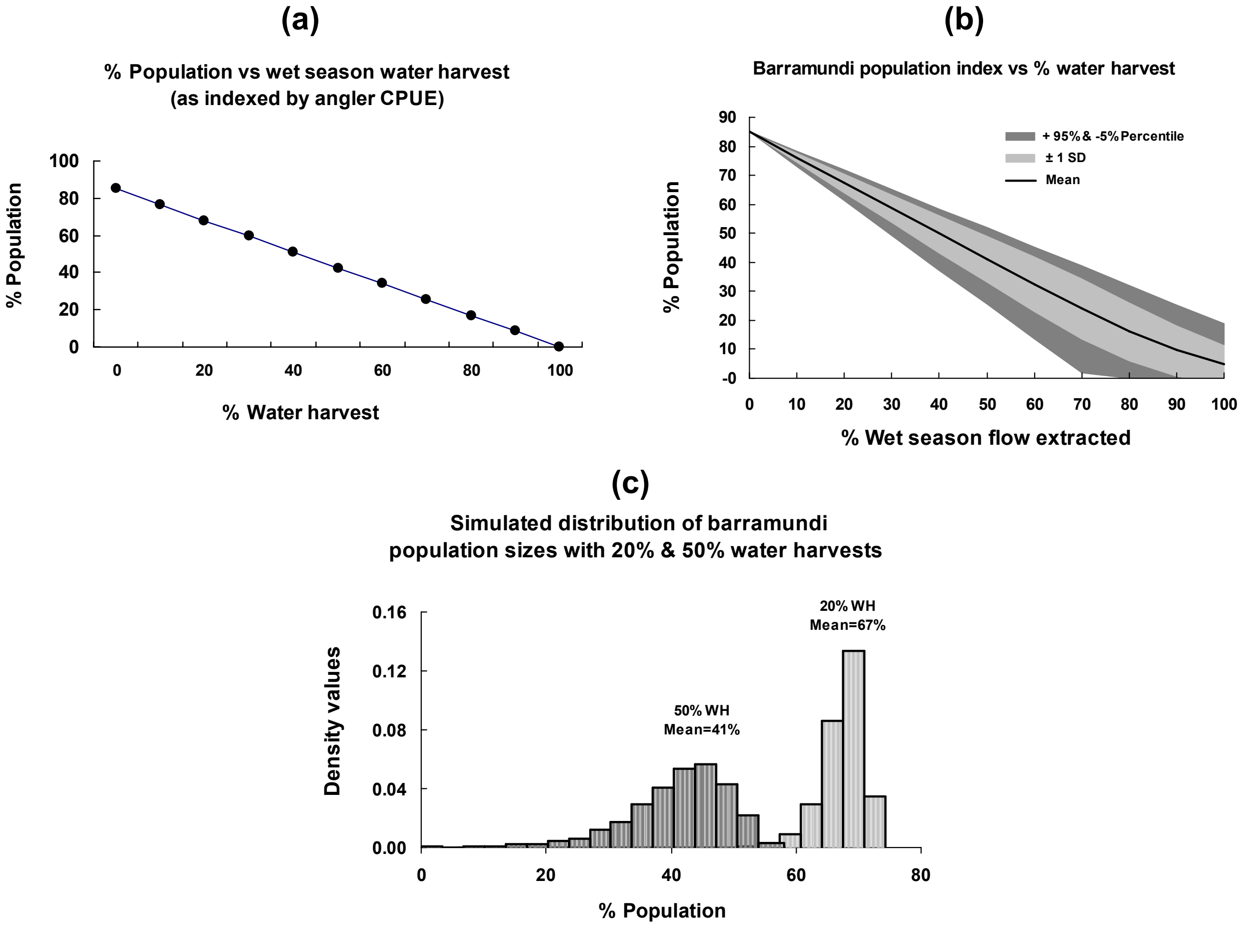


Figure 4.43 a-c (a) Simulated reduction in mean barramundi population size (angler 'Classic' CPUE) as a function of wet season flow extraction using the simple regression equation summarised in Table 4.18. Mean values were derived by Monte Carlo simulation (10,000 iterations) using @Risk software (Pallisade 2002b) and incorporated model uncertainty as outlined in Figure 5.39. (d) As for (c) but with uncertainty levels illustrated using one standard deviation (SD) about the mean trend and the + 95% and – 5% percentiles. (c) Simulated pdf distribution of barramundi population sizes (angler 'Classic' CPUE) for simulated wet season water harvests of 20% and 50%.

commercial catches are arbitrarily defined based on the distribution of observed catches (see Table 4.19). Needless to say, these subjective levels can be varied according to stakeholder input and/or better technical knowledge. Under these conditions Figure 4.44a shows that, with no flow extraction, 65% of the assessment of in-stream health is classified as 'Ok and Excellent', with the majority being 'Excellent' (48%). In contrast, Figure 4.44b shows that, with a 20% wet season flow extraction, only 17% of the assessment of in-stream health is classified as 'Ok and Excellent' with the majority being 'Poor' (83%).

Decision Tree for recreational & commercial catches and water extraction

The above BNs were converted into a Decision Tree to more clearly examine the effects of different water extraction and fishing effort policies on commercial and recreational barramundi catch in the Daly River. The assessment endpoint to determine the optimal policy for each fishery separately can be the number of barramundi caught for recreational fishing or the weight of catch for commercial fishing. However, to compare policy options between

Table 4.19 Variable ranges, state levels (Low, Medium & High) and associated probabilities used in an initial Bayesian Network to assess risk to in-stream health from simulated water extractions. Probability density functions (best fit statistical distributions) and cumulative probability distributions were examined in Best FitTM (Pallisade 2002a) for natural breaks in observed data. Current state levels are highlighted in bold, and simulations are limited to the min-max range of observed data. In-stream health is indexed by two barramundi endpoints, one is socio-economic (total recreational catch) and the other is ecological (population size as indexed by recreational Classic CPUE).

Variable	State Level	Range	Best fit distribution	Prob.
Wet season flow (ML x 10^6)	LOW MEDIUM HIGH	0.00 – 4.7 4.7 – 10.0 10.0 +	Exponential but tri-modal	0.50 0.30 0.20
Commercial effort (100m net sets/day)	LOW MEDIUM HIGH	0.00 – 3,076 (mode) 3,076 – 8,000 8,000 +	Log Logistic	0.33 0.59 0.08
Angler recreational effort (angler hrs)	LOW MEDIUM HIGH	0.00 – 4,000 4,000 – 5,400 3.74 +	Triangular	0.21 0.47 0.32
Total recreational effort (angler hrs + line hrs tours)	LOW MEDIUM HIGH	0.00 – 5,000 5,000 – 9,000 9,000 +	Logistic	0.29 0.45 0.26
Commercial catch (weight t)	LOW MEDIUM HIGH	0.00 – 41 41 – 71 71 +	Log Logistic	0.18 0.62 0.20
Angler recreational catch (number fish caught)	LOW MEDIUM HIGH	0 – 450 450 – 850 850 +	Weibull	0.43 0.30 0.27
Total recreational catch (number angler & tour fish caught)	LOW MEDIUM HIGH	0 – 1,400 1,400 – 4,500 4,500 +	Exponential	0.48 0.38 0.14
CPUE angler (fish caught/angler hr)	LOW MEDIUM HIGH	0.00 – 1.0 1.0 – 2.3 2.3 +	Extreme Value	0.24 0.48 0.28

fisheries, or to compare policy options encompassing the benefits of water extraction from agricultural production to the benefits generated by fishing, a monetary ($) value of the catch should be used as the assessment endpoint. A first-cut is attempted here but obviously requires more considered revenue data. The mean dollar value of a recreational barramundi caught is estimated at $21/fish using assumptions and data found in Coleman (2004). They could be inaccurate (either biased up or down) and, if so, the BN and Decision Tree approach would allow the uncertainty level associated with this estimate to be incorporated into the decision making process. They are used here merely as a starting point until better data become available. Table 4.20 summarises the payoff utility values (ie converting fish numbers or weight to $s), and state variable levels for water extraction (0, 20% & 50%), fishing effort (Low, Medium & High) and flow (Low, Medium & High) used as Decision Tree branches. Branch pathways for flow and effort are determined by Monte Carlo simulation of their probability distributions and so accounts for model uncertainty and variability. However, these state levels can be chosen *a priori* depending on which policy option, or combination of control variables (flow & effort), is being explored.

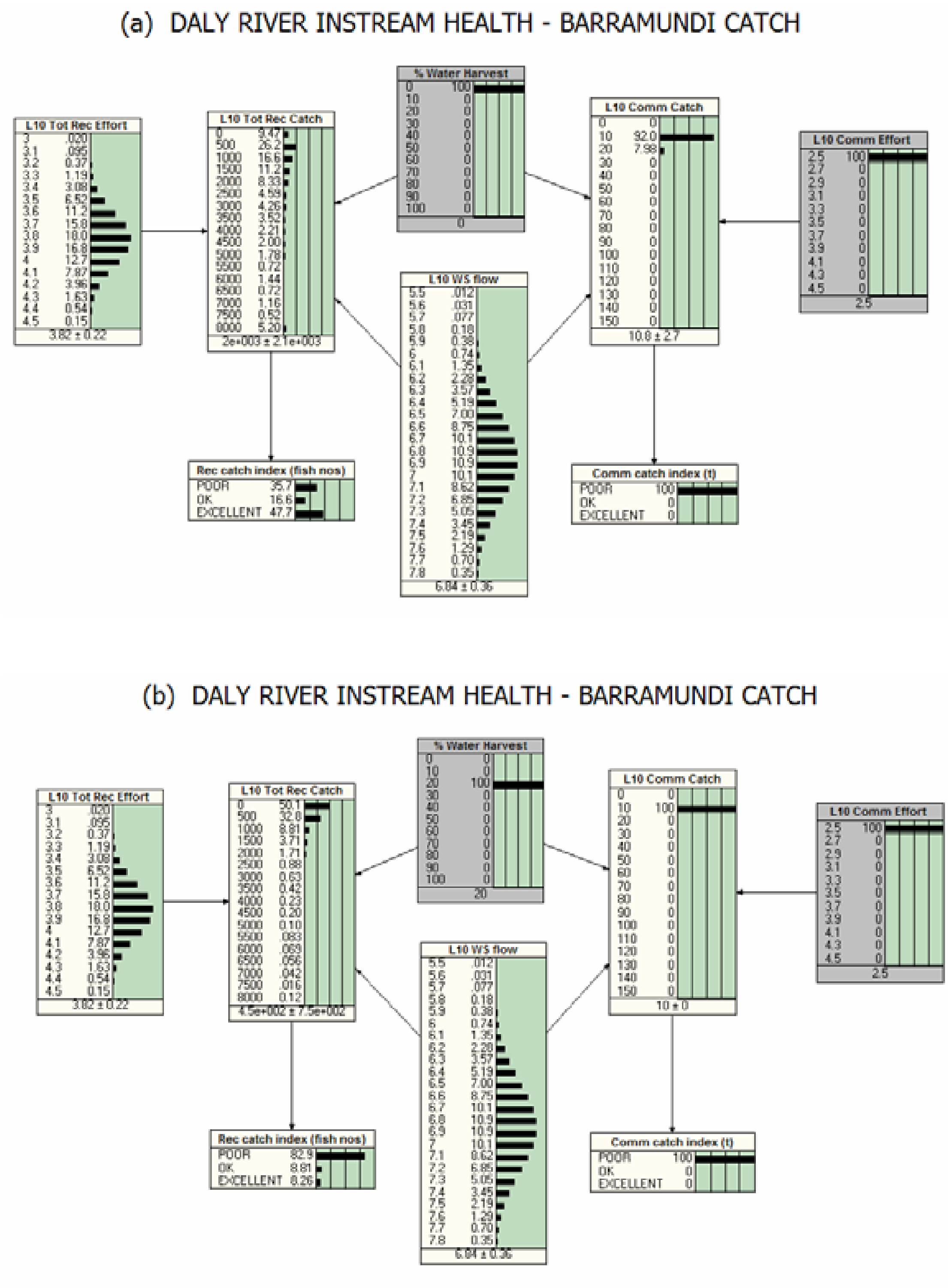


Figure 4.44 a & b Bayesian Network for Daly River In-stream Health as indexed by impact on barramundi fishery catches, for two wet season flow extraction scenarios: (a) no extraction and (b) maximum allowable 20% extraction. States reflect current conditions (High flow, High recreational effort & Low commercial effort). Overall potential catch is low because of low commercial effort, and a 20% wet season flow extraction has significantly reduced total recreational catch.

Table 4.20 Summary of Decision Tree analysis payoff utility functions ($ cost/recreational barramundi caught, $ cost/t for commercial barramundi), and state variable levels (Low, Medium, High) used to simulated effects of different water extraction policies on barramundi fish catch in the Daly River.

PAYOFFS		Fishery type (1=Rec 2 = Comm)	1	WATER HARVEST	H	Proportionate reduction
$/t barra	$18,000			No water harvest	0	1.0
		Water harvest policy None	1	Max allowable	20	0.8
		Max 20% wet season flow	2	High	50	0.5
		50% wet season flow	3			
				EFFORT (T REC)	% Catch reduction REC	
$ spent/fish caught/rec fisher	$21		1	Low (Min)	0	1.00
$cost/fish classics	$3.36		2	Medium (Mean)	20	0.23
Number anglers (2005 Classics)	135		3	High (Max)	50	0.02
Number tour fishers						
Total recreation	15,853 hrs			EFFORT (COMM)	% Catch reduction COMM	
Anglers	6,750 hrs		1	Low (Min)	0	1.00
Tours	9,103 hrs		2	Medium (Mean)	20	0.68
			3	High (Max)	50	0.38

Figure 4.45a shows the Decision Tree results for the recreational barramundi fishery in terms of both total catch and dollar value. As expected the optimal policy is no water harvest, high flow and high effort. For simplicity the branches for a 50% water harvest and for commercial fish catch are not shown. The recreational fishery decreases in value by 77% with a 20% water harvest ($129K c.f. $29K) under high flow and high effort states (present condition). Note that effort may increase in future without major impact on sustainability of the fishery, and that this trend, or even reduced trends in catch as a result of decadal declines in flow, would need to be accounted for in any decision analysis. Similarly, Figure 4.45b shows the Decision Tree for the commercial barramundi fishery in terms of both total catch in weight and dollar value. As expected, the optimal policy is no water harvest, high flow and high effort. The branches for a 50% water harvest and for recreational fish catch are not shown. The commercial fishery decreases in value by 67% with a 20% water harvest ($1.9 million c.f. $0.7 million p.a.) under high flow and effort states. Under high flow and low effort state (present condition), the decrease in value is 64% ($957K c.f. $343K p.a.).

4.3.7 Bayesian Network for in-stream health based on barramundi population size

A second Bayesian Network (BN) for in-stream health was constructed for the ecological assessment endpoint. Angler Classic CPUE (the measurement endpoint) was used to index barramundi population size (the ecological assessment endpoint). As with the approach adopted for previous BNs, the stochastic model developed above (see regression equation in Table 4.18) to predict barramundi CPUE from natural flow regimes was incorporated into the BN and used to simulate the influence on in-stream health of different wet season flow extraction scenarios (0-100%). Wet season flow is characterised by its probability distribution rather than state levels, hence inputs into the subsequent CPUE child node, and the assessment endpoint child node for in-stream health, will encompass the full range of flow conditions encountered between 1983 and 2005. The percentage population size of barramundi was adjusted for the fact that current stock biomass levels are thought to be about 15% of unharvested levels (B. Grace pers. comm., NT Fisheries).

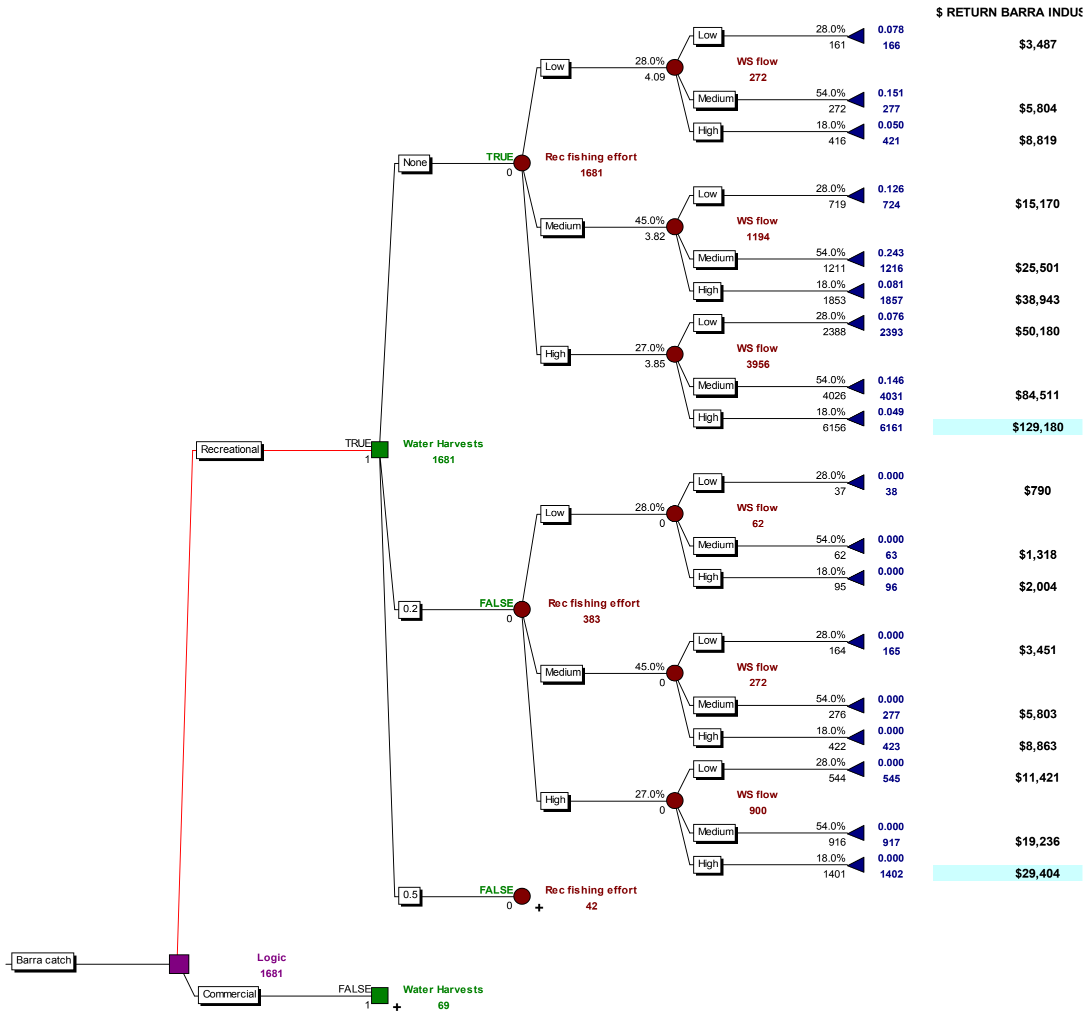


Figure 4.45a Decision Tree or the recreational barramundi fishery in the Daly River in relation to three (Low, Medium & High) effort and wet season flow regimes, and a two flow extraction policies (no water extraction, maximum 20% of wet season flow). The optimal policy with respect to barramundi catch can be measured using total number of fish caught or the dollar value of a recreational fish caught times all fish caught (see Appendix, derived from Coleman 2004). The branch for a 50% water harvest is not show as for the branch for commercial fish catches. The recreational fishery decreases in value by 77% with a 20% water harvest ($129K c.f. $29K) under high flow and high effort states (preset condition, although effort may increase in future without major impact on sustainability of the fishery).

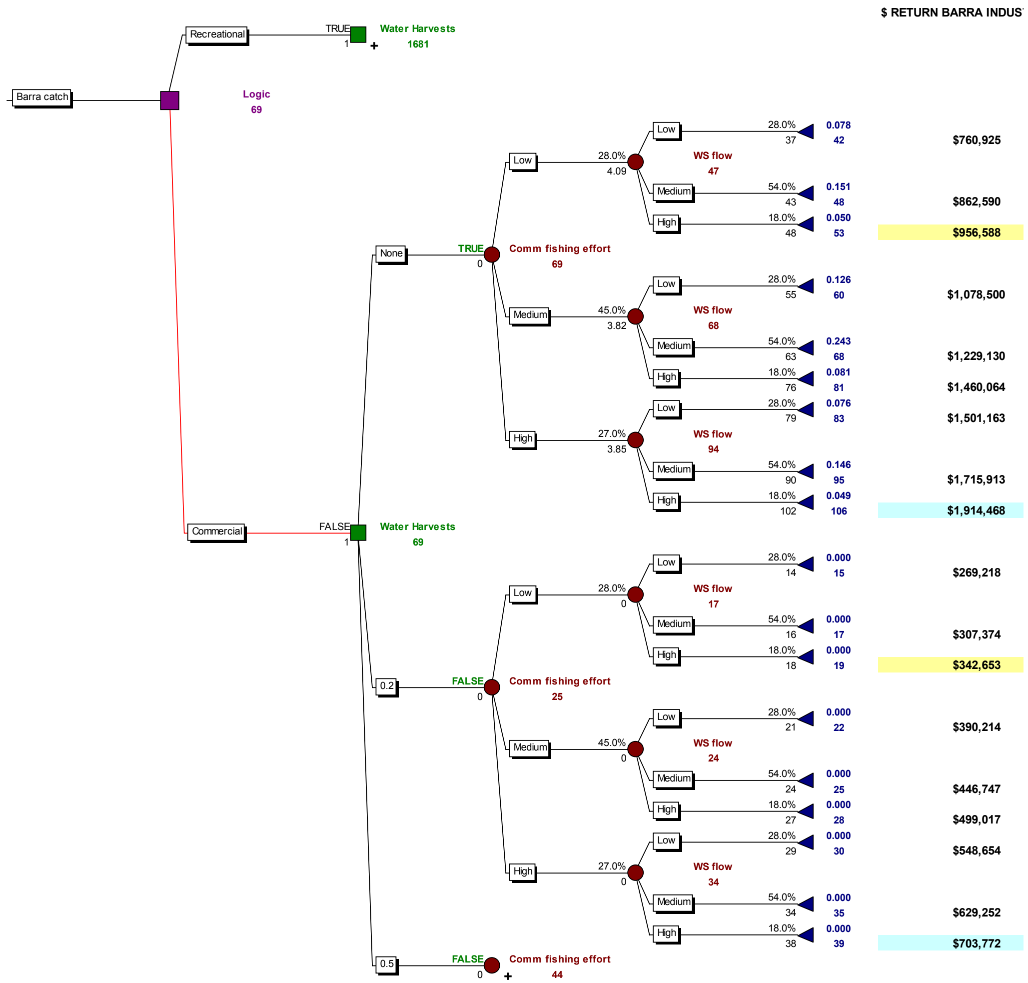


Figure 4.45b Decision Tree or the commercial barramundi fishery in the Daly River in relation to three (Low, Medium & High) effort and wet season flow regimes, and a three flow extraction policies (no water extraction, 20% & 50% extraction of wet season flow). The optimal policy with respect to barramundi catch can be measured using total weight of fish caught or the dollar value of the commercial catch. The branch for a 50% water harvest is not show as for the branch for recreational fish catches. The commercial fishery decreases in value by 67% with a 20% water harvest ($1.9 million c.f. $0.7 million p.a.) under high flow and effort states. Under high flow and low effort state (present condition) the decrease in value is 64% ($957K c.f. $343K p.a.)

Whilst state levels for in-stream health based on barramundi population size are arbitrary, they are nevertheless underpinned by basic harvesting dynamics theory and the precautionary principle. We assume that the population dynamics of barramundi is best approximated by an interactive consumer-resource model, implying the existence of negative density-dependent feedback loops that impose some form of regulation (eg via food supply, cannibalism, availability of habitat etc). Hence, if true, MSY at 50% of unharvested stock level as estimated by a logistic or surplus production model would be inappropriate. A more appropriate MSY (& accompanying 'safe' reduced population level) would be about 70% of unharvested stock level (see Bayliss 1989 for magpie geese). Even so, if Logistic population growth assumptions apply then the influence of environmental variability on key life history parameters (eg recruitment, survival & dispersal) may reduce SYs and, consequently, increase the harvested population level required to generate MSY somewhere between 50% and 100% (Bayliss 1989). Hence, 'Excellent' population levels are assumed to be between 90-100% of unharvested stock level, 'Ok' population levels > =70% and < 90% and 'Poor' population levels <70% but >=1%. A simulated population size <=1% is considered an extinct population, the worst case scenario.

Two simulation scenarios are illustrated here; no flow extraction and a 20% flow extraction. Under a scenario entailing natural flow conditions and no flow extraction, most (97%) of the barramundi population is classified as 'Ok' (Figure 4.46a). In contrast, 100% of the barramundi population is classified as being in 'Poor' condition under a 20% wet season extraction scenario (Figure 4.46b).

4.3.8 Other commercial fish

The commercial gillnet catch data for the Daly River included other commercial fish species besides barramundi, providing an opportunity to examine possible flow relationships for a range of life histories. The matrix of species catches between 1983 and 2005 is seriously unbalanced across time; hence sample sizes varied considerably limiting the scope of multi-species analysis. Catch data are available for 14 species of commercial fish including barramundi, and a record for mud crabs. Fish that weren't identified to species when caught had their catch weights combined into a 'Mixed fish' class. There were sufficient catch data for nine other fish species for analysis besides barramundi. Multiple regression analysis was used to examine relationships between catch in combination with effort and flow for each of the nine species. Linear regression analysis was used to examine trends in population size as indexed by CPUE over time (years). The following analyses are exploratory only, and results are summarised in Table 4.21.

Catch significantly increased with effort only for cod, jewfish and mackerel, and flow was positively correlated with catches of shark, mackerel and snapper, although these relationships are complex and not as direct as that for barramundi (Table 4.21a). Spectral analysis was undertaken on population size of all species (as indexed by their CPUE) to detect possible period trends in relation to flow or other factors. Periodograms of spectral density of shark and snapper population sizes both show 11 and 22 year periods, effectively an 11 year cycle, and is similar to the period signature for commercial barramundi CPUE. No fish species declined significantly between 1983 and 2005, and only snapper and mixed fish significantly increased on average over time (Table 4.21b).

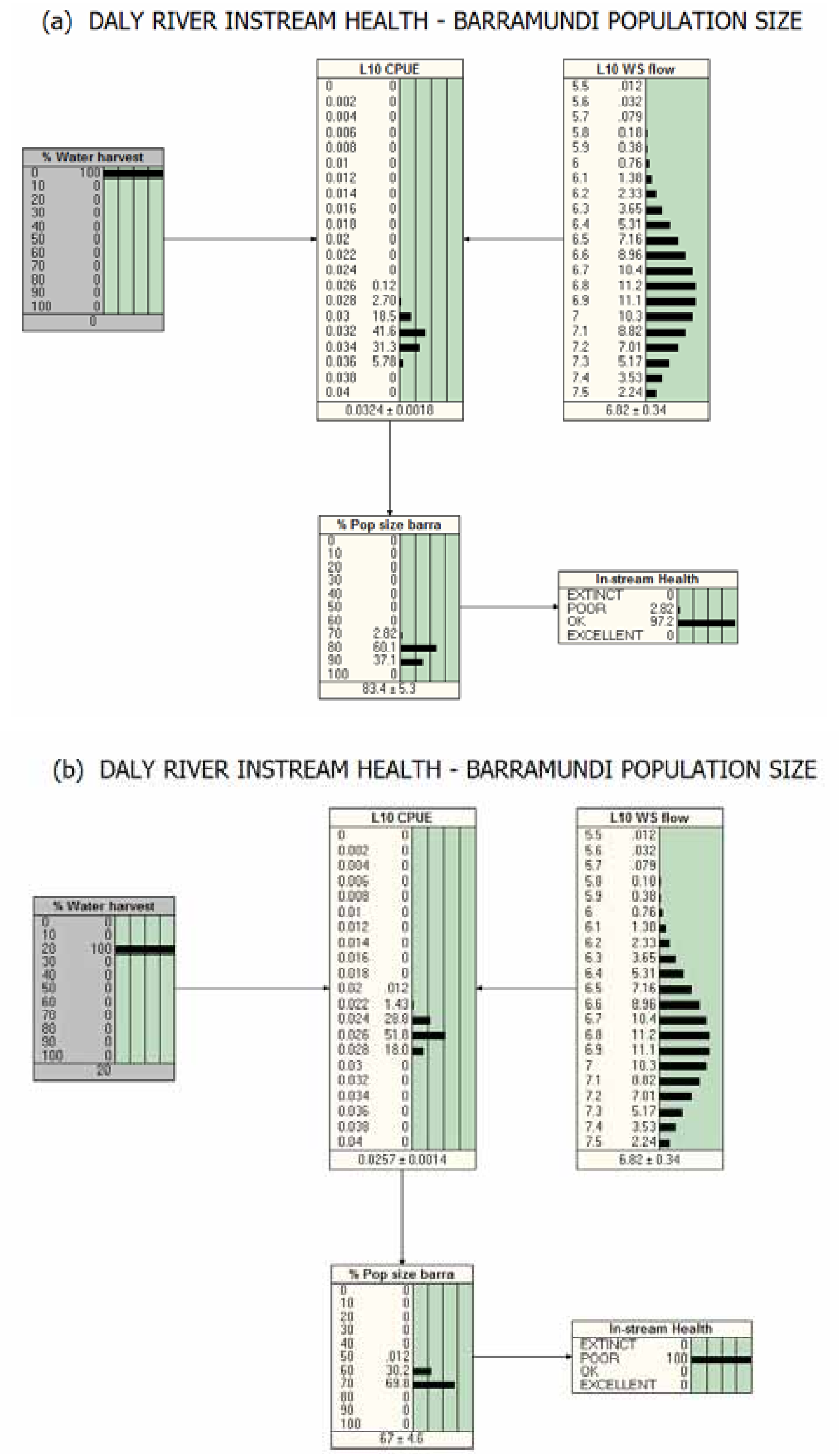


Figure 4.46 a & b Bayesian Network for Daly River in-stream health as indexed by impact on barramundi population size (via CPUE in the recreational fishery), for two wet season flow extraction scenarios: (a) no extraction and (b) maximum allowable 20% extraction.

Table 4.21a & b Summary of relationships between (a) other fish species caught (log_{10} Weight kg) in commercial catches in combination with river flow (log_{10} WSQ ML) and effort (log_{10}100m net sets/day), and (b) log_{10} CPUE (as an index of population size) and time (years). Daly River fishing zones 1329 and 1330 (Figure 4.27a, 1983 – 2005). See data summary in Appendix 8.3.1 and 8.3.2.

Species	(a) Catch vs. effort & flow in joint equation (NS = non-significant; sig = significant)	(b) CPUE vs. time (yrs) Regression stats
Cod	Flow NS, effort sig (P<0.05); (regress R^2=30%, P=0.04)	NS, n=16
Snapper	Wet season flow sig. (via cross correlation), effort NS	P<0.01, n=11, R^2= + 63%
Queenfish	Flow & effort NS	NS, n=18
Jewfish	Flow NS, effort sig (P<0.001); (regress R^2=57%, P<0.001)	NS, n=23
Threadfin salmon	Flow NS; effort sig.	NS, n=23
Blue salmon	Flow & effort NS	NS, n=13
Shark	Wet season flow sig (P=0.03) & with a 3-year time lag, effort NS; (regression R^2=27%, P=0.07)	NS, n=20
Mackerel	Wet season flow & effort sig; (regression R^2=78%, P=0.02)	NS, n=7
Mixed fish	Flow & effort NS	P<0.01, n=23, R^2= + 25%

Robins et al (2005) documented significant positive correlations with freshwater flow for barramundi, mud crabs, mullet, flathead and a number of prawn species (banana, school, king, tiger & greasy). These are species where flow relationships were reported as being investigated, however there are most likely a number of commercial species that have not been examined and a number of species that have been examined but not reported. A framework approach, as suggested by Robins et al (2005), is required to systematically screen all commercial species for freshwater-flow relationships.

4.3.9 Discussion & recommendations

Baran and Cain (2001) recommended coupling ecosystem models with a Bayesian Networks in order to address relationships between environmental modifications (eg water extractions) and natural fish production in the Mekong Basin, a tropical river-floodplain system. We adopted a similar approach for the Daly River; however we were not constrained by data on flow, fish catch and fishing effort as in the Mekong Basin.

Our scenario simulation results predict that wet season flow extraction, even at moderate levels (20%), will have a major impact on barramundi fish catch in the Daly River and, hence, one socio-economic value of the river (Table 4.22). The strong relationship between flow and barramundi catch was used to predict potential tradeoffs between reduced flow from simulated extractions and lost fisheries value in terms of revenue (commercial or recreational) or intangible benefit (eg recreational catch – number of fish caught). The Bayesian Network for barramundi catches and the Decision Tree analysis reported here sets the scene for more comprehensive benefit-cost analysis if required, to determine optimal allocation of competing resources such as flow and fish catch. This approach, however, requires knowledge on how the economic demand function shifts with changes in flow or flow related variables such as fish catch (Loomis & Cooper 1990), and part of this knowledge is provided here. Needless to say, more detailed knowledge is required on the benefits and costs of recreational fishing in the Daly River, including intangible benefits and opportunity costs.

Table 4.22 Summary of (a) simulation results predicting the percentage reduction of recreational and commercial barramundi catches in the Daly River from a 20% and 50% flow extraction scenario, using the simple multiple regression models summarised in Table 4.14 - 4.16 and, similarly, for (b) percentage reduction in population size as indexed by angler Classic CPUE.

Barramundi fishery	Percentage reduction	
	20% water extraction	**50% water extraction**
(a) Catch		
Commercial	32	62
Angler Classic	70	95
Total recreational (angler + tours)	75	97
(b) Population abundance		
Angler Classic CPUE	32	58

Resource economists generally use contingent valuation methods (CVM) and travel cost methods (TCM) to measure the value of environmental goods in stream flow studies (Loomis & Cooper 1990). The CVM is a market simulation approach that asks people their net willingness to pay for alternative stream flows, and can be used to value river users and the general publics' willingness to pay for river protection through reduced flow extractions. Loomis and Cooper (1990) used a travel cost demand equation that included the level of fish catch as the quality variable, which in turn was a function of river flow, and a variation of this method was used here. A key value of barramundi in the Daly River is now recreational fishing, and should not be underestimated in terms of generating economic revenue and non-monetary benefits (Griffin 1979, Coleman 2004). Additionally, whilst commercial barramundi fishing has been excluded from the Daly River reach, the value of their catch in adjacent coastal fishing zones still influenced by freshwater flows from the Daly River should also not be underestimated.

The lack of strong interaction found between the commercial and recreational barramundi fisheries in the Daly River between 1985 and 2005 may reflect a lack of competition for a shared and reducing fish stock because of early preventative management in 1989. However, an equally plausible hypothesis is that the combined fishing efforts and catches at the time of intervention were insufficient to cause competition. Additionally, barramundi harvesting models (B. Grace pers. comm., NT Fisheries) suggest that current fishing effort has reduced fish biomass by only 15-20%. Hence, flow appears to be the underlying driver of past and current barramundi catches and their population abundance. Both the Katherine and Daly rivers exhibited 20-year periods in average flow between 1971 and 2006 (see Section 4.1.1), and strong relationships were found between barramundi catch, effort and river flow in the Daly River fisheries between 1983 and 2005 (see Section 4.3.1). Hence, we predict that the positive trends in barramundi catch and CPUE population indices will reverse in the next decade. The occurrence of decadal trends in stream flow has been documented in many catchments across the globe (see Section 4.1.1), as have relationships between freshwater stream flow and fish landings or catches (Beamish et al 1994, Staunton-Smith et al 2004). In particular, Robins et al (2005) found that barramundi catch in the Fitzroy region of

Queensland was positively correlated to freshwater flow, and that changes in catch between 1945 and 2002 showed notable 15 to 20-year cycles in the data.

Relationships between changes in climate regime, marine ecosystems and fish catch are well documented globally (eg Francis & Hare 1994, Hare & Mantua 2001). However, in contrast and somewhat surprisingly, there are almost no documented studies for Australia. For example, Beamish et al (1999) used cusum analysis of climate indices (eg SOI, PDO & North Atlantic Oscillation Index or NAOI, among others) to show that trends in Pacific salmon production this century are linked to trends in climate in the Pacific, which are in turn associated with climate trends in the Northern Hemisphere. Beamish et al (1999) emphasised the importance of being able to differentiate between sudden changes in fish catches due to climate regime shift with those due to changes in fishing effort. Hare and Francis (1994) demonstrated that Alaskan salmonid catches alternated between high and low production periods, and suggested that this phenomenon was linked to North Pacific climate processes. Mantua et al (1997) linked dramatic shifts in salmon production regimes in the North Pacific Ocean with recurring patterns of ocean-atmosphere climate variability (eg via SOI & PDO) and, McFarlane et al (2000) demonstrated similar links for a range of North Pacific fisheries (eg hake, sardines, groundfish). The North Pacific climate pattern apparently also affected streamflow in major west coast river systems from Alaska to California (Mantua et al 1997). Additionally, the abundance and hence catches of sockeye salmon have been related to decadal-scale changes in climate and the ocean (Beamish et al 1997, Schindler et al 2005), and similarly for Chinook salmon (Scheuerell & Williams 2005) and Pacific halibut (Clark et al 1999).

Not surprisingly barramundi abundance was sensitive to simulated wet season flow extractions (Table 4.22), suggesting that impacts of future water developments in the Daly River catchment on aquatic biodiversity may be significant. Nevertheless, although a few other commercial fish species also exhibited catch-flow relationships, we have no knowledge of the importance of freshwater flow of the majority of fish species. Hence, we cannot comment with certainty on potential biodiversity impacts of future flow extractions, although there are many studies highlighting the importance of flow in linking fish life history traits with habitat condition. For example, Travnichek et al (1995) found that artificial fluctuations in stream flow cause by hydroelectric power dams can degrade fish habitat and reduce the abundance and diversity of riverine fish fauna. Turner et al (1994) found that larval and juvenile sunfish (*Lepomis* spp) were most abundant in habitats exhibiting low flow rates and high temperatures. And Scheidegger and Bain (1995) highlighted that, even though larval fish distributions and habitat requirements are often distinctly different from large and older fish, the ecology of larval fish is poorly known for many species in free-flowing and regulated rivers. They suggested that flow regulation and the associated degradation of nursery habitats is a major threat to the conservation of natural and diverse riverine faunas.

We emphasise that we have only simulated wet season flow extraction although future water development scenarios in the Daly River catchment are likely to comprise dry season flow extractions and/or groundwater extractions. The Daly River maintains a high base flow in the dry season because of groundwater inflow, and wet season flow extraction has the potential to reduce dry season flow on top of potential reductions from dry season groundwater use through bores. The effects of such changes on the ecology of freshwater fish are unknown and, hence, research is required on seasonal flow relationships for a wide range of fish species encompassing the diversity of life histories and functional community groups found in aquatic habitats of the Daly River-floodplain ecosystem. Such a study has already commenced for the Daly River and is funded through NHT (M. Douglas pers. comm.).

Finally, besides the direct effects on fish communities from reductions in flow, there are many other potential impacts associated with water extraction that need to be examined in future risk assessments. For example, land use associated with water use within a catchment may have a major influence on the terrestrial-aquatic interface *per se* and, hence, could affect fish populations and their community dynamics through the disruption of biophysical processes, particularly spatial heterogeneity and connectivity of physical habitats (Schlosser 1991). The effects of invasive species, from aquatic weeds and exotic fish in riverine and floodplain habitats, needs to be considered also in future risk assessments and their impacts on ecological assessment endpoints made explicit.

4.4 Risks of land clearing on surface water quality

Executive summary

Land clearing for land use has been identified as a key threat to the health of aquatic ecosystems in the Daly River catchment. The major land use is cattle grazing, followed by irrigated horticulture and cropping production. However, in 2005 only 4% of the catchment had been cleared for land use with 96% remaining in relatively 'pristine' condition. Nevertheless, agricultural developments are likely to expand in the near future, increasing risk to surface water quality and, hence, aquatic ecosystem health. A major constraint in assessing ecological risks of future land clearing and associated land use change is the paucity of ecologically relevant surface water quality data, especially for downstream reaches close to floodplain and estuarine environments. To overcome this knowledge gap regression models were developed using data from North Queensland catchments to predict exports of total sediment (TS t/y) and total phosphorus (TP t/y), and dissolved inorganic nitrogen concentration (DIN μM), from the percentage of the catchment remaining in pristine condition, to apply to the Daly River catchment. We assume that land use is a simple and direct predictor of sediment and nutrient loads because it integrates many disturbance-based environmental attributes that influence their export. Hence, the regression models are treated as surrogate physico-chemical process models. All regressions are significantly nonlinear because of an 80% threshold effects level for the percentage of land cleared, indicating the possible existence of irreversible binary states in catchment condition. The 80% threshold contrasts with a 50% threshold for land cleared as suggested by Harris (2001a) for Australian catchments in general.

Monte Carlo simulation was used to predict mean TS and TP exports from the Daly River catchment, and mean DIN concentration. As expected, results show that a 4% extent of land cleared would have a negligible effect on surface water quality. A Bayesian Network was constructed to establish a framework approach so that future land clearing scenarios at different threshold effect levels can be examined for the whole Daly River catchment or its sub-catchments. Scenario simulations show that, when the effects threshold is 50% of land cleared, the only sub-catchment predicted to have poor water quality is Green Ant. Sub-catchment Relative Risk Ranks underpinning the semi-qualitative risk assessment method in Chapter 3 were highly correlated to modelled losses of sediments and nutrients derived here, and so may be a more practical measurement endpoint for surface water quality when screening typically data poor northern catchments across regional scales. The broad physico-chemical endpoints used to assess surface water quality need to be refined to encompass more sensitive and so more useful ecological indicators, such as those that capture direct links between nutrients, sediments and the condition of biological communities in different trophic levels (eg some fish, macroinvertebrates, *Vallisneria nana*, algae & benthic diatoms).

The influence of land use type with respect to vegetation cover and agricultural nutrient inputs, and the effect of landscape-scale fires, on sediment and nutrient loads were not captured in the QERA and should be addressed in future. Riparian and estuarine ecosystems were not addressed also and, hence, should be included in future because of the connectivity between all ecosystems from catchment to coast to sea. To conclude, a major limitation of the QERA is that we excluded key flow values of local Indigenous people. However, we argue that Bayesian Networks could accommodate most Indigenous cultural values because they implicitly recognise the value of subjective knowledge and beliefs from other domains and, therefore, gives respect and weighting to them.

Technical summary

1. Water quality in catchments is a function of land use, hydrogeochemistry and interactions between in-stream, riparian and floodplain habitats. Harris (2001b) found that a sharp increase in the export of salinity, sediments and nutrients to the surface and groundwater pathways occurs when about 50% of land is cleared in a catchment, and is associated with a corresponding decline in water quality.
2. Land clearing and associated agricultural land use activities has been identified as a key threat to the health of Daly River aquatic ecosystems through its potential to change surface-water runoff characteristics, increase soil erosion and sediment delivery, reduce groundwater recharge and river baseflows through flow extraction for irrigation, and impact on aquatic ecosystems through changes in water quality. The current major land use in the Daly River catchment is cattle grazing, followed by irrigated horticulture and cropping.
3. In 2005 only 4% of the total catchment had been cleared for other land uses with 96% remaining in relatively 'pristine' condition, representing an average annual loss rate of pristine land between 1966 and 2005 of 0.1%. However, the extent of land cleared for land use had not been uniform between sub-catchments (0.1-59%) and this variation was investigated as part of the risk assessment. Tropical rivers discharge about 70% of Australia's annual freshwater runoff and, hence, their potential is now being investigated for expanded agricultural developments because of water supply issues in southern catchments that currently support most of Australia's agricultural production. Hence, agricultural developments in the Daly River are likely to expand in the near future, increasing the risks to surface water quality and aquatic ecosystem health from increased land clearing rates.
4. Concern over the threat to ecosystems of the Great Barrier Reef from pollution in terrestrial runoff has led to the development of a variety of physico-chemical process models to estimate sediment and nutrient (N & P) exports from North Queensland catchments. There are insufficient historical and contemporary nutrient and sediment concentration data collected near the Daly River mouth to adopt a similar approach, especially under high flow conditions. Hence, regression analysis was used to develop physico-chemical process models for Queensland tropical catchments to predict total sediment (TS t/y), total phosphorus (TP t/y) and dissolved inorganic nitrogen concentration (DIN μM) from the percentage of the catchment remaining in pristine condition. These relationships are applied to the Daly River catchment and its sub-catchments. The underlying, albeit convenient, assumption is that the percentage of land cleared for land use is a simple and direct predictor of nutrient loads because it integrates many disturbance-based environmental attributes that influence sediment and nutrient exports.

5. The relationship between modelled TS export and the percentage of pristine land remaining in North Queensland catchments was nonlinear and exhibited an extreme threshold effects level. A similar nonlinear threshold relationship was found by Brodie (2002) for DIN concentration and the percentage of the catchment remaining in pristine condition. North Queensland catchments apparently exist either in one of two states; one with low sediment and nutrient loads and low to moderately high levels of land clearing, or one with high sediment and nutrient loads and very high levels of land clearing. The switch between the two states is very abrupt, and the threshold between them occurs when about 80% of the catchment is cleared. This is in contrast to the 50% level found by Harris (2001b) for Australian catchments in general.
6. A conceptual model was constructed to guide our risk assessment of the health of Daly River 'surface water quality' under different land clearing scenarios. The following three physico-chemical assessment endpoints were used in combination to assess potential ecological impacts to aquatic ecosystems: (i) total sediment (TS) export; (ii) total phosphorus (TP) export; and (iii) dissolved inorganic nitrogen (DIN) concentration. Measurement endpoints were TS and TP exports (t/y), and DIN concentration (µM), as predicted from the nonlinear relationships between these variables and the percentage of North Queensland catchments remaining in pristine condition.
7. The stochastic process models above were then used to predict the effects of existing and simulated land clearing extent in the Daly River catchment on TS and P exports, and DIN concentration, for the different scenarios outlined above. Mean values were derived by Monte Carlo simulation. The extent of land cleared is entered as a constant in a single catchment or sub-catchment analysis, hence only model uncertainty is addressed in simulations.
8. As expected, simulated mean modelled TS and TP export from the Daly River catchment increased rapidly with increasing percentage of land cleared past the 80% threshold effects level and, similar results were obtained for DIN concentration. Uncertainty levels in predicted outputs are low because model error is low and inputs for the extent of cleared land is constant. Model outputs predict that by 2005, when 4% of the catchment had been cleared for other land uses, sediment and nutrient impacts to surface water quality and, hence, aquatic ecosystem health would have been negligible.
9. A Bayesian Network was constructed to establish a framework to examine future land clearing scenarios and the influence of different threshold effects levels on sediment and nutrient water quality parameters. This approach could be applied separately to the whole Daly River catchment or to its sub-catchments. Four scenarios were examined: (i) the 2005 4% extent of land cleared is applied to the whole Daly River catchment with an effects threshold set to 80% of land cleared, as suggested from North Queensland catchments; (ii) the 2005 59% extent of land cleared in the Green Ant sub-catchment is applied with the same 80% threshold value; and (iii & iv) the effects threshold for the extent of land cleared is lowered to 50% for both scenarios above (ie the level suggested by Harris 2001b). The combined water quality index for both scenarios with an 80% threshold is 'Good'. At a 50% threshold the Daly River catchment water quality index remains 'Good', but that for the Green Ant sub-catchment slides to 'Bad'. Whilst these results are obvious, the BN nevertheless provides a framework to encompass a range of more sensitive and perhaps more informative biological responses linked to the surface water quality assessment endpoints used here, and which may respond well before threshold levels and potentially irreversible regime shifts are reached.

10. The proportion of land cleared in a catchment or sub-catchment underpins all modelled outputs of sediments and nutrients used here to assess surface water quality. The area or proportion of land cleared when classified as a threat to ecosystem health also underpins the spatially-based Relative Risk Model (Chapter 3) applied to the whole catchment. We examined the concordance of modelled sediment and nutrient water quality parameters with the total Relative Risk Ranks derived for sub-catchments, and found strong positive linear correlations. Whilst this concordance doesn't validate either methodology because both are predicated on the same 'land cleared equals threat' paradigm, use of sub-catchment Relative Risk Ranks may be a more practical and efficient measurement endpoint for broad surface water quality and other ecological assessment endpoints.
11. The 80% threshold effects level suggests that most of the catchment would have to be cleared of native vegetation cover for alternative land uses before a measurable, and most likely irreversible, change is detected. Hence, models that predict sediment and nutrient exports loads in relation to land cover *per se* would be useless as early warning systems to change land use policy before aquatic ecosystem collapse occurs. Another conceptual problem with our 'export out of the catchment' model, although relevant to coastal and offshore impacts (& highly relevant to the Great Barrier Reef), is the centennial-scale retention times of sediments in catchments. This may also apply to some nutrients, although phosphorus is generally associated with the fine suspended clay fraction that is highly mobile. Hence, equal importance should be placed on potential water quality impacts on aquatic ecosystems within catchments that may trap sediments and some nutrients, such as freshwater wetlands and the riparian zone. The broad physico-chemical endpoints used to assess surface water quality here could be refined to encompass more sensitive and so more useful ecological indicators, such as those that capture the direct links between nutrients, sediments and the condition of biological communities in different trophic levels (eg some fish species, macroinvertebrates, *Vallisneria nana*, algae & benthic diatoms).
12. Other key ecological processes in catchments that may influence water quality and not addressed in this risk assessment are: the influence of land use type on water quality; the influence of land clearing and land use on ground water recharge rates and, hence, ground water dependent ecosystems such as riparian habitats; and the effects of landscape-scale fires on water quality in floodplain, in-stream and riparian habitats.
13. The cause of the elevated levels of nitrate found in groundwater in the Douglas River sub-catchment needs to be identified because it has the potential to impact on future water quality in both the Douglas and the Daly rivers. The source may be natural (eg as a result of decadal shifts in rainfall & associated groundwater & surface water flows), or anthropogenic (eg agricultural fertiliser inputs &/or constant release of soil nitrates after land is cleared). With respect to possible anthropogenic sources further work is required to determine whether or not the increased nitrate level is due to the septic discharge or fertiliser used in land use (P. Jolly pers. comm., NT NRETA). It may be beneficial, therefore, for comparative purposes, to sample groundwater nitrate concentrations in adjacent sub-catchments influenced by the Tindall Limestone aquifer that have relatively pristine and heavily cleared land covers (eg Green Ant). Additionally, N-isotopic ratio studies could be conducted at the same time to identify nitrate sources. In conclusion, more work is required on what happens to nitrate after application and as it travels through the groundwater and through the hyporrheic zone into the river (P. Jolly pers. comm., NT NRETA).

4.4.1 Introduction

Land clearing and land use change – global, national & local issues

Land clearing has far reaching environmental impacts on catchments and waterways, in addition to habitat loss and fragmentation, and is a national and global issue. In association with land use change it can cause deleterious and cumulative changes to soil properties, vegetation cover and ground and surface water quality. A major compounding issue is that surface runoff and seepage to groundwater is increased because reduced vegetation cover is less effective at impeding water flow and retaining nutrients and particulates (Harris 2001b, Fierer & Gabet 2002).

In southern Australia the clearing of forested land and regulation and extraction of river flows for agriculture and urban developments has led to eutrophication of inland and coastal waters (Harris 2001a) and, consequently, to an increase in the frequency and severity of algal blooms (Harris 1994a, Caraco 1995, Young et al 1996). Ganff and Rea (2007) reported that NT rivers (including the Daly River) have the potential to experience algal blooms with nutrient enrichment; they found that NT rivers had low nutrient status and a viable inoculum of blue-green, brown and green algal communities. The combination of flow regulation and wetland loss has also had major impacts on the ecology of many Australian rivers. Tropical rivers discharge about 70% of Australia's annual freshwater runoff and, hence, are now being targeted for similar agricultural developments (Hamilton & Gehrke 2005).

Water quality in catchments is a function of land use, hydrogeochemistry and interactions between in-stream, riparian and floodplain habitats. Harris (2001b) examined explicit links between land use and water quality in Australian catchments and found that, at about 50% clearance, there is a sharp increase in the export of salinity, suspended solids (in surface waters) and nutrients to the surface and groundwater pathways, with a corresponding decline in water quality. Schueler (1997) argued that suspended and deposited sediments have major impacts on aquatic biota and recreational values.

Harris (2001a, b) reviewed the factors influencing Nitrogen (N) and phosphorus (P) exports from Australian catchments and found that pristine forested catchments exported little N and P, and that the predominant form of N is dissolved organic nitrogen (DON). In contrast, as catchments are cleared of native vegetation nutrient exports increase and the predominant form of N changes from DON to dissolved inorganic nitrogen (DIN). Harris (2001b) found also that as catchments are cleared the DIN: SRP (soluble reactive phosphate) export ratios increase sharply and DIN becomes an increasing proportion of total N. He therefore argued that the ratios of N: P and DIN: SRP in rivers reflect land use and residence times of the waters. Harris (2001a) suggested also that DIN and dissolved inorganic phosphorus (DIP) are better nutrient indicators than total nitrogen (TN) and total phosphorus (TP), and that the ratio of DIN to DIP is a good predictor of algal blooms. Harris (2001a) concluded, however, that whilst both N and P are biologically relevant indicators of water quality and, hence, good predictors of aquatic ecosystem health, their biogeochemistry are not well understood. Downing et al (1999) summarised knowledge on the potential impact of land use change on the nitrogen biogeochemistry of tropical aquatic ecosystems, and traced the N-cycle from pre-disturbance (pristine) conditions through the phases of disturbance. They found that tropical freshwaters are more frequently N-limited than temperate waters, while tropical marine systems are more frequently P-limited. They concluded that disturbances to pristine tropical catchments will lead to greatly increased primary production in freshwaters triggering substantial changes in biological communities, and significant impacts in fragile mangrove and reef ecosystems because of the switch from P to N-limitation in calcareous marine environments.

Drewry et al (2006) reviewed current knowledge of nitrogen and phosphorus generation from land use and exports to waterways in Australia and provided a link between current and future modelling requirements as a context for catchment models for use by catchment managers. They differentiated between empirical (ie data-based), conceptual (process-based; hypotheses to test) and physics-based models derived from small plot experiments, and concluded that catchment models need to represent the importance of event-based loads, type of land use management and associated forms of nutrients (eg intensively farmed, grazing etc). They argued that such representation is needed in order not to underestimate nutrient losses and overestimate the effectiveness of riparian buffers. North Queensland rivers generally have highly irregular flow regimes, and transport of materials through waterways (eg dissolved & particulate forms of suspended sediments & nutrients) occurs almost completely during major flow conditions (Brodie 2002; Moss et al 1992).

In global terms, however, Australia has low total export of nutrients from catchments because of low average rainfall, relief, soil nutrients and population pressure with accompanying fertiliser use. Atmospheric deposition of nutrients is also not a major issue as it is for Northern Hemisphere counties (Carpenter et al 1998, Harris 2001a). Harris (2001b) argued that freshwater aquatic ecosystems exist in two states; either clear and macrophyte dominated or turbid and plankton dominated, characterising relatively pristine and heavily impacted catchments, respectively. He argued also that the switch between the two states can often be abrupt because of complex, nonlinear responses to perturbations, which show strong resistance to switching back because of 'critical loads' or points of 'no return'. Many Australian rivers have changed from being clear and macrophyte dominated to being turbid and plankton dominated. Harris (2001b) suggested that, although the costs of river and wetland rehabilitation are too great to contemplate, estuaries and coastal lagoons are worth saving because they have not yet reached their critical load points.

Terrestrial and aquatic ecosystems are strongly linked because of the downstream transfer of materials such as nitrogen, phosphorus, carbon and sediment. The catch-22 situation is, therefore, that to protect downstream coastal environments we basically need to carefully manage the upstream catchments. Land clearing and associated land use changes can have large impacts on this linkage by altering material flows. Caraco and Cole (1999) found that the export of NO_3 and PO_4 was mostly influenced by land use activities and, in contrast, there was little detectable influence for dissolved organic carbon. They found that, for sediments, land use impact was detectable at regional scales despite the strong mitigating influence of physical features of the catchment.

Hence, as with nutrients, there is increasing concern over the downstream influence that sediments and erosion have on in-stream and estuarine ecosystems (Berry et al 2003, Syvitski et al 2003, Crowe & Hay 2004). Wasson et al (1996) found that changes in medium-coarse sediment supply in Australia within the last 200 years have not been accompanied by concomitant increases in river sediment yields, indicating that residence times of sediments in rivers is long. Increased storage of sediment can significantly alter river physical form, chemical processes and, ultimately, the health of aquatic ecosystems over decadal time scales (Prosser et al 2001a). Prosser et al (2001a, b) examined the patterns of sediment transport in some Australian rivers in terms of source, transport and deposition throughout the river network. They showed that erosion of hill slopes and stream banks greatly increased in historical times supplying vast quantities of sediments to rivers, with much of it still stored within the river system with the potential to affect aquatic ecosystems for decades to come.

Young et al (1996) suggested that land use could be used as a simple and convenient predictor of nutrient loads because it is a known integrator of many environmental attributes that influence nutrient export, and this approach is adopted here. Additionally, land use impacts in catchments may involve a suite of unmeasured accumulated effects over centennial timescales and, in the absence of detailed historical data on soil, water and vegetation condition, the proportion of the catchment left in a naturally vegetated state may be the only indicator available to assess condition at landscape and regional scales. Another advantage of this approach is that it allows rapid 'first pass' benchmarking of impacted and relatively pristine catchments across large slabs of northern Australia that characteristically lack biophysical data to assess current condition of their aquatic ecosystems for conservation and/or production purposes. Land use and land clearing data are generally readily available, even for remote areas, because of rapid advances in cost-effective remote sensing and GIS technologies (see Hosking 2002 for the NT). For example, Johnson et al (2000) used remote sensing and GIS technologies to quantitatively assess both spatial and temporal changes in land cover in the lower Herbert River catchment in north-east Queensland since European settlement in the mid 19th Century, and demonstrated that there has been substantial reductions in the areas of melaleuca, rainforest and eucalyptus dominated land cover patterns for intensive agriculture and grazing. Brizga et al (2002) outlined the 'Benchmarking Methodology' developed for Queensland aquatic ecosystems that encompasses the use of biophysical 'reference' and 'impacted' sites, ecological flow responses and risk assessment procedures, and is similar to the overall approach adopted in this report.

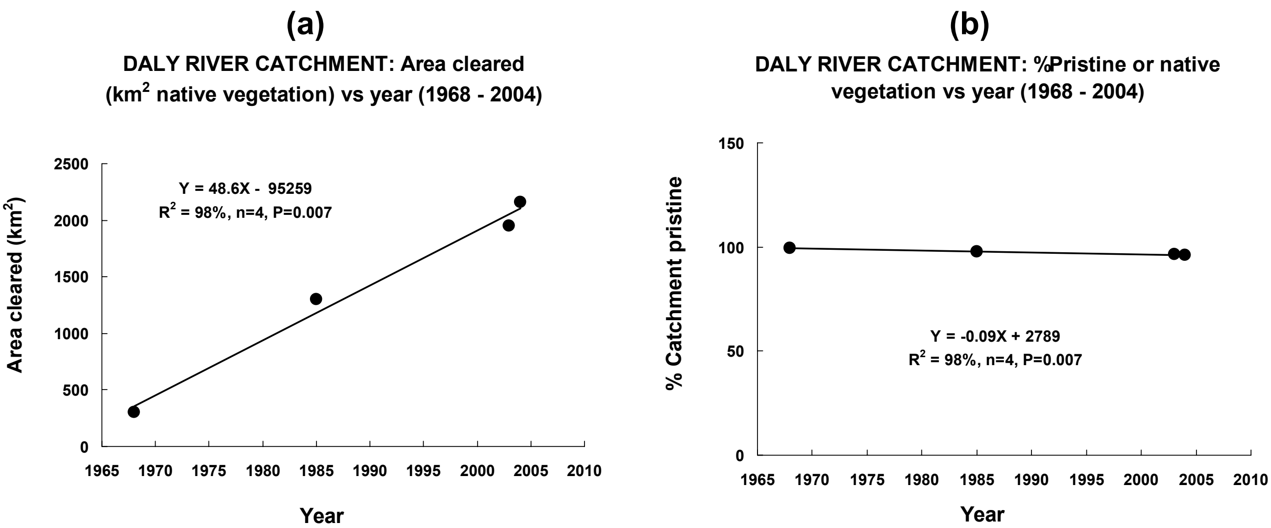
In contrast, however, Allan (2004) argued that empirical associations between land use and stream response has had limited success in implicating pathways of influence because of the following reasons: (i) covariation of anthropogenic and natural gradients in the landscape; (ii) the existence of multiple, scale-dependent mechanisms; (iii) nonlinear responses; and (iv) the difficulties of separating present-day influences from historical influences. He argued further that research examining land use responses under different management strategies should employ response variables that have greater diagnostic value than currently used aggregated measures, such as the extent of cleared land. We agree with his first argument because the reasons outlined above support our recognised need for greater understanding and knowledge of catchment-scale biophysical processes. However, we disagree with his second argument because the long times and considerable financial resources needed to acquire comprehensive predictive knowledge about individual processes may exclusively delay required intervention measures. Our overall approach, in the absence of such comprehensive predictive knowledge, uses aggregated indices of complex biophysical landscape processes such as the extent of land cleared (or conversely the extent of land remaining in pristine condition), within a risk assessment framework.

4.4.2 Land clearing and water quality in the Daly River Catchment

The major land use in the Daly River catchment is cattle grazing, followed by agricultural production such as horticulture and cropping, concentrated near Katherine township and the middle reaches of the Daly River in the vicinity of Oolloo Crossing and the confluence of the Douglas and Daly Rivers (Webster et al 2005; see Chapter 3). Land clearing and associated agricultural land use activities have been identified as a key threat to the health of Daly River aquatic ecosystems through its potential to change surface-water runoff characteristics, increase soil erosion and sediment delivery (see Furlonger 2004), reduce groundwater recharge and river baseflows through flow extraction for irrigation, and impact on aquatic ecosystems through changes in water quality (Kennedy 2004, Webster et al 2005 from Erskine et al 2003).

Land clearing for agricultural land use activities can directly affect water quality in receiving waters through loss of P and N fertilisers, in addition to sediments through soil erosion. Dilshad et al (1996) found that during 1984-89 in the Daly River Basin, conventionally tilled catchments produced 1.5-2.0 times more runoff and lost 1.5-6.0 times more soil than untilled catchments (up to 8.1 t/ha/y). Additionally, removal of native vegetation can dramatically change catchment water balance and, hence, recharge rates to streams. The potential effects of land clearing on recharge rates in the Daly River catchment has received much attention (Wilson et al 2006a,b; Knapton 2006), particularly with respect to tree water use by riparian vegetation (O'Grady et al 2002a,b; Lamontagne et al 2005).

Data on land use and land clearing in the Daly River catchment were obtained from a variety of sources (Blanch et al 2005, Wilson et al 2006b, Wygralak 2006), including the NT land use maps of Owen and Meakin (2003) and NT land clearing maps and statistics from Hosking (2002). Total catchment size is estimated at 53,322 km^2 (Begg et al 2001 & various). In 1968 Scott Creek station cleared 300km^2 for improved pastures and crops (sorghum, maize & millet) and an extra 100km^2 was cleared elsewhere. In 1985 Tipperary Station and Scott Creek combined cleared about 1,300km^2. In 2003 and 2004 about 1,950km^2 and 2,158 km^2 of land, respectively, were cleared for mixed farming in the Daly bioregion. Analysis of the extent of land cleared in the Daly River catchment between 1996 and 2005 (Figure 4.47a) shows, however, that on average only 39km^2 has been cleared annually over 36 years. Hence, by 2005 only 4% of the catchment had been cleared for other uses with 96% remaining in relatively 'pristine' condition, representing an average annual loss rate of pristine land between 1966 and 2005 of 0.1%. However, clearing rates have not been uniform between sub-catchments (see Table 4.33) and this is examined further in the risk assessment. By 2010 an extra 2,000km^2 of land is predicted to be at risk from clearing (Blanch et al 2005), and this estimate was made before the North Australian Taskforce was recently established to examine agricultural potential in northern Australia.



Figures 4.47 a & b (a) Regression between area of Daly River catchment cleared for land use and time between 1966 and 2005 (on average 39km2 p.a. over 36 years). (b) The percentage of the Daly River catchment that is pristine (ie not cleared for land use) between 1966-2005 (on average 0.1% loss p.a. over 36 years, a total of 4% of land cleared in 2005).

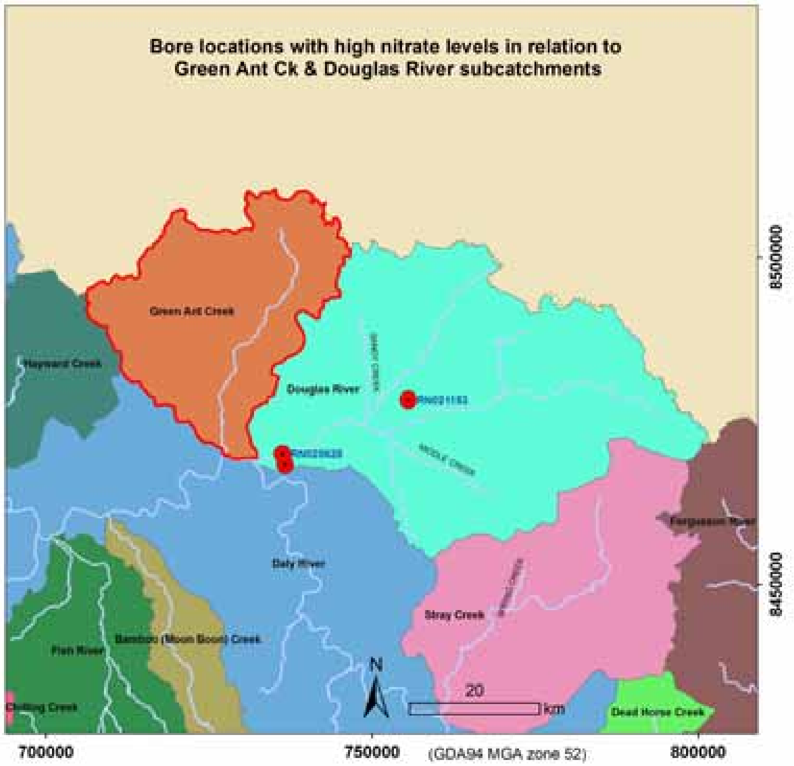
Padovan et al (1999) reviewed the water quality data in the HYDSYS™ database for the Daly River Basin and concluded that, overall, there is a paucity of ecologically significant data such as nutrients, suspended sediments and chlorophyll. Hence, determination of seasonal, inter-annual and decadal trends in water quality in relation to changes in climatic drivers such as rainfall, and/or anthropogenic change such as land clearing and associated land use, is not possible. They found that the most frequently sampled sites were for the Katherine River close to Katherine Township because of public health considerations.

Hence, a major constraint in assessing ecological risks of land clearing and associated changes in land use in the Daly River catchment is the paucity of ecologically relevant water quality data, especially for downstream reaches close to floodplain and estuarine environments. However, since the report by Padovan a number of studies have been initiated to address knowledge gaps in stream ecology and water quality as summarised by Erskine et al (2003). For example, Rea et al (2002) examined the environmental requirements of *Vallisneria nana* in the middle reaches of the Daly River and found very low nutrient levels (eg nitrite 0.001–0.01 mg/L, nitrate 0.004-0.04 mg/L, reactive phosphorus <0.005 mg/L) and concluded that the river would be extremely susceptible to the effects of enrichment. Webster et al (2005) examined primary production in the same river reach in order to better understand some of the key ecological drivers. Townsend et al (2002) examined periphyton (diatoms or microscopic algae) and phytoplankton responses to reduced dry season flows and concluded that they would be impacted by flow extraction and changes in water quality. Townsend & Padovan (2005) examined seasonal growth and biomass of benthic algae (*Spirogyra*), and Townsend & Gell (2005) examined the role of substrate type on benthic diatom assemblages.

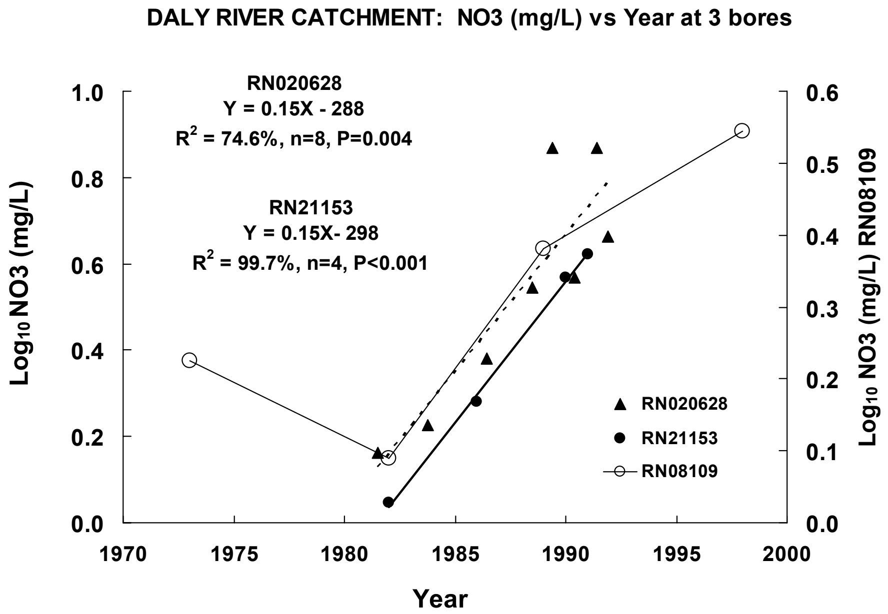
Despite very low nitrate concentrations found in the middle reaches of the Daly River by Rea et al (2002), Schult and Metcalfe (2006) examined the occurrence of elevated nitrate concentrations in the lower reaches of Douglas River during dry season surveys by Townsend et al (2002), which apparently persisted into 2004. Nitrate levels were 10-20 times higher than in the Daly River and other waterways in the area. Although Townsend et al (2002) did not detect any impact of these high nitrate levels on the water quality of the Daly River 62 km downstream (eg via increase phytoplankton concentration), the nutrient enrichment of the Douglas River may be an impact in itself and has the potential to impact on Daly River water quality (Schult & Metcalfe 2006) depending on mixing conditions at the confluence of the two rivers. Hence, it is examined below in further detail.

Nitrate concentrations measured in three bores in the Douglas River sub-catchment (Table 7 in Schult & Metcalfe 2006; Figure 4.48a) have not always been elevated and show a rapidly increasing trend between the early 1980's and 1999 (Figure 4.48b). This coincides with a 4-fold increase in the extent of land cleared from Scott Creek and Tipperary stations. By 2005 the Douglas River sub-catchment had 15% of land cleared, and the adjacent Green Ant Creek sub-catchment, 60%. However, despite the circumstantial association with high land clearing extents, the source of the elevated nitrate concentrations is currently unknown. Possible explanations offered by stakeholders include: naturally high nitrate levels in the Tindal Limestone aquifer; pollution from septic tank leakage; nutrients originating from animal faeces such as bat droppings; seepage of nitrogen into the groundwater from current or past application of fertilisers; and growing of nitrogen-fixing legumes in agriculture (Schult & Metcalfe 2006). Note that the first sampling point in the Douglas River that detected increased nitrate levels was located downstream of the large septic disposal system in the caravan park located on the river up-gradient of where some large springs discharge into the

(a)



(b)



Figures 4.48 a & b (a) Location of the water quality sample bores and sub-catchments, and (b) increasing time trends in nitrate concentration (NO3, log_{10} mg/L) at three bores in the Douglas River sub-catchment. These high nitrate concentrations still persisted into 2004 (Schult & Metcalf 2006).

Douglas River (P. Jolly pers. comm., NT NRETA). Hence, further work is required to determine whether or not the increased nitrate is due to the septic discharge or fertiliser used in land use (P. Jolly pers. comm., NT NRETA).

The impact that fertilisers have on nitrate, sulphate, calcium and fluoride concentrations in groundwater at the Douglas-Daly Experimental Farm has been examined by NT NRETA (P. Jolly pers. comm.), and they conclude that the magnitude of the impact on nitrate levels appears to be strongly and positively correlated with the previous wet season recharge (ie increase in the groundwater levels). Potential seepage into the groundwater across the Douglas-Daly area in concentrations like what has happened (or is happening) at the Experimental Farm is a concern, however it is interesting to note that similar concentrations of fertiliser chemicals have not been detected in springs or in the river. In conclusion, more work is required on what happens to the nitrate after the farmer applies it and as it travels through the groundwater and through the hyporrheic zone into the river (P. Jolly pers. comm., NT NRETA).

4.4.3 Catchment-based sediment and nutrient export models in relation to land clearing

The North Queensland experience

Agricultural land use along the east coast of Queensland has led to increased soil erosion and nutrient loss from catchments adjacent to the Great Barrier Reef (Brodie & Mitchell 2005, 2006). The export of fine sediment, nitrogen and phosphorus in coastal rivers has increased several-fold as catchments have been cleared of native vegetation for intensive grazing and crop production (Johnson et al 2000, Brodie & Mitchell 2006). Concern over the threat to ecosystems of the Great Barrier Reef from pollution in terrestrial runoff has spurned a large number of studies that used a variety of methods and models to estimate sediment and nutrient (N & P) exports from tropical catchments (Moss et al 1992, Brodie 2002, Bartley et al 2003, Brodie et al 2003, Brodie & Mitchell 2006). These estimates have been reported in a number of National Land and Water Audit reports (Norris et al 2001, Prosser et al 2001b), Commonwealth Sate of the Environment reports (Hamblin 2001) and at national water quality workshops (see Davis et al 1998 for eutrophication & phosphorus).

The sediment and nutrient export models and data that support them have grown in sophistication and reliability through time (Brodie & Mitchell 2006). Moss et al (1992) used a 'desk top' approach to estimate sediment, nitrogen and phosphorus exports from catchments using simple models of runoff, land use and sediment delivery. An established process model was calibrated with available, albeit often limited, data (eg via frequent wet-season sampling of nutrient & sediment loads), and estimates of sediment and nutrient exports based on flow-weighted discharge-export relationships for rivers characteristic of each region were derived. In contrast, Prosser et al (2001a, b) used a sediment generation, delivery and trapping model (SedNet) to estimate sediment discharge from Queensland rivers and, similarly, Kinsey-Henderson et al (2005) used SedNet to model sources of sediment at sub-catchment scales for the Burdekin catchment in North Queensland.

Harris (2001a) searched the published and grey literature sources for water quality and nutrient budget data to compare them with global patterns and found none for WA or the NT. Additionally, Young et al (1996) reviewed the exports of TN and TP from Australian catchments to compare with Northern Hemisphere catchments and noted the paucity of Australian data in general. They therefore suggested that land use could be used as a simple and convenient predictor of nutrient loads because it is a known integrator of many environmental attributes that influence nutrient export, and this approach is adopted here by coupling land use with land clearing. A major advantage of this approach is that it allows benchmarking of impacted and relatively pristine catchments across large regions of northern Australia that characteristically lack biophysical data to assess the condition of their aquatic ecosystems.

Water quality is generally only monitored on a regular basis at a small number of sites in a catchment, one of which is usually located at the catchment outlet because it integrates the effects of all point and diffuse sources throughout the catchment (Grayson et al 1997). Although a number of studies have recently been completed in the Daly River in the last several years that have sampled water chemistry parameters, most of these have been confined to the middle reaches of the Daly River or to the reaches of the Katherine River downstream of Katherine Township. Hence, there are insufficient historical and contemporary nutrient and sediment concentration data collected near the Daly River catchment outlet (see Section 4.4.2) to adopt the process modelling approach used for North Queensland catchments. Instead regression models were developed for Queensland tropical catchments to predict sediment and nutrient exports (or concentration) from the proportion of the catchment remaining in pristine condition (or left uncleared for intensive land use), and applied generically to the Daly River catchment.

Regression models linking sediment & nutrient exports to land clearing in North Queensland

Suspended sediment and phosphorus export or load data have been sourced from Moss et al (1992) for 12 North Queensland catchments (Table 4.24), which are reported as total annual exports (t/y) or total annual exports per unit area (t/y/ha). Harris (2001a) found that DIN became an increasing proportion of total N when catchments were cleared for land use. Hence, DIN concentration (μM) data from Brodie (2002) are used here to index potential export of N.

Annual rainfall and runoff in northern Australia varies greatly under the influence of the summer monsoon, the occurrence of El Niño events and the unpredictable occurrence of cyclones. Most of the runoff of water, sediments and nutrients to the Great Barrier Reef occurs during short-lived flood events (Brodie et al 2003). Young et al (1996) suggested that approximate adjustments for rainfall intensity and runoff volume may need to be made to estimates of sediment and nutrient exports based on land use. However, no adjustment was made by including them in our regression models as independent variables because the dependent response variables are modelled mean estimates of sediment and nutrient exports based on flow-weighted discharge-export relationships (Moss et al 1992); they would therefore not be independent.

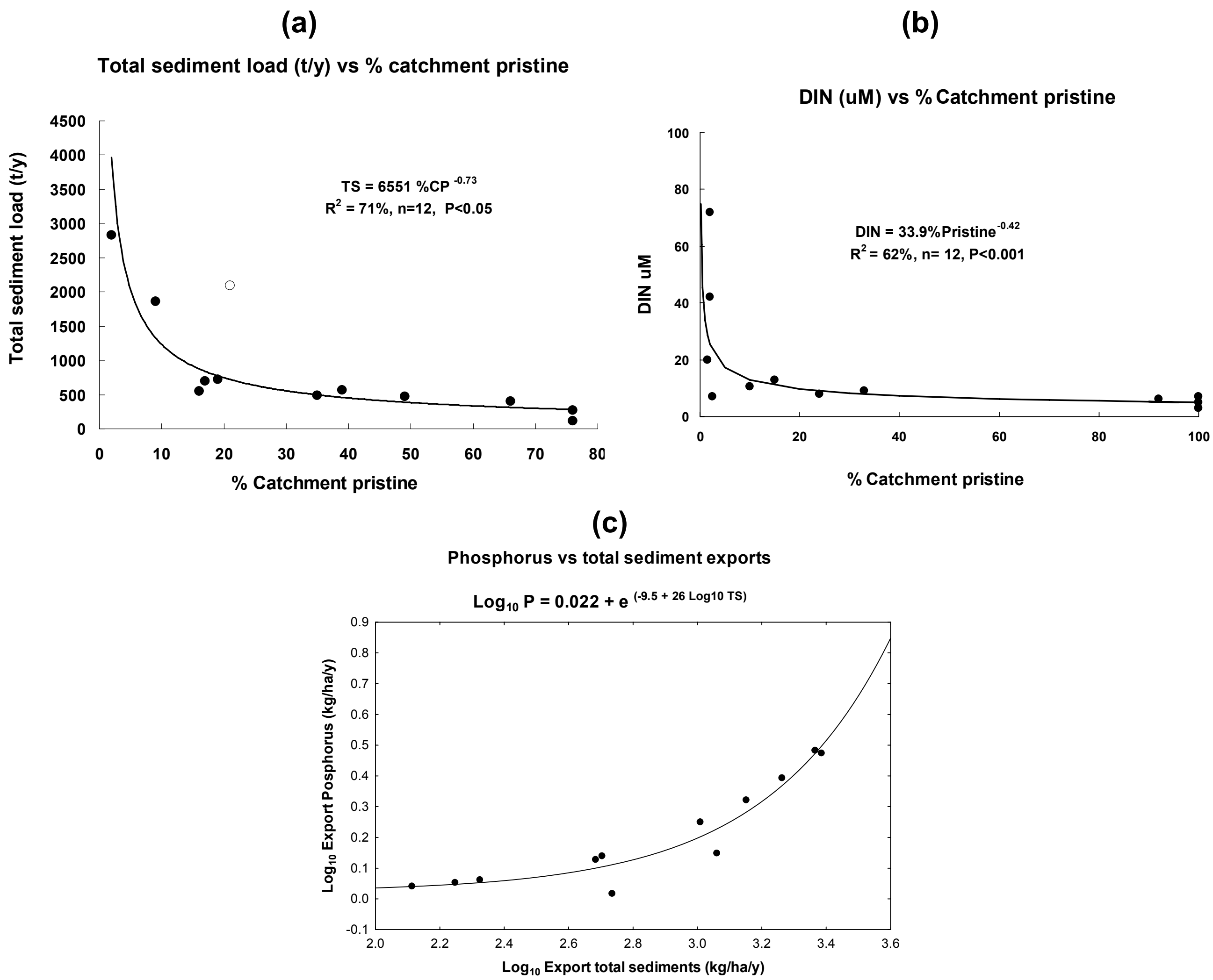
Total sediment load (TS t/y) increases exponentially as the percentage of the catchment in pristine condition (%CP) decreases below about 20% (Figure 4.49a), and the relationship is best described as a power function (Table 4.25a) explaining 71% of the data. NE Cape York appears an outlier and, when excluded, the explained variance of the model increases to 90% (Table 4.25b). Moss et al (1992) classified NE Cape York as an outlier in their analysis because 61% of the catchment was being used for cattle grazing and so could not be considered semi-pristine. The 20% level may be a threshold value, above which there is little effect of land clearing in catchments on TS yield, and below which TS yield increases almost vertically, perhaps exhibiting an extreme hysteresis effect.

Table 4.24 Catchment area, mean annual flow and estimated total sediment (TS t/y) and total phosphorus (TP t/y) exports for 12 North Queensland catchments. Data sourced from Moss et al (1992) and used by Davis et al (1998) and Hamblin (2001). Total annual phosphorus export was estimated from total annual phosphorus exports expressed in kg/ha.

Catchment	Catchment area km^2	% Catchment pristine	Mean annual flow ML x 10^3	TS export kg/ha/y	P export kg/ha/y	TS export t/y	TP export t/y
Mary	9,595	35	300	506	0.38	486	365
Burnett-Kolan	39,470	17	2900	177	0.13	698	513
Fitzroy	142,646	9	7100	130	0.10	1861	1426
Pioneer-O'Connell	3,925	19	2650	1838	1.47	720	577
Burdekin-Haughton	133,510	2	10850	212	0.15	2829	2003
Herbert	10,130	16	5000	543	0.04	550	41
Tulley-Murray	2,825	66	5300	1422	1.10	401	311
Johnstone	2,300	39	4700	2436	1.98	567	455
Mulgrave-Russell	2,020	49	4200	2328	2.04	471	412
Barron	2,175	76	4000	1150	0.41	114	89
Mossman-Daintree	2,615	76	4250	1024	0.78	268	204
NE Cape York	43,300	21	19100	484	0.34	2096	1472

Data for dissolved inorganic nitrogen (DIN µM; nitrate + ammonia) concentration data, and the corresponding percentage of the catchment in pristine condition, were extrapolated from Figure 2 in Brodie (2002, Table 4.26) and used to fit a nonlinear regression model (Table 4.27, Figure 4.49b). As with TS, DIN increases exponentially as the percentage of the catchment in pristine condition decreases below about 20% (Figure 4.49b). This relationship is best described as a power function (Table 4.27) explaining 62% of the data. The 20% level may also be a threshold value, above which there is little effect of land clearing in catchments on DIN concentration, and below which there is an almost vertical increase in DIN concentration.

Note that Brodie (2002) used µM as a unit of concentration and a correction factor to convert to mg/L would be 14 (D. Jones pers. comm., ***eriss***). Hence, the threshold DIN concentration value (for 20% of the catchment in pristine condition) of about 10 µM converts to 0.14 mg/L. The maximum concentration of nitrate from all three bores in the Douglas River sub-catchment of the Daly River (see Section 4.4.2) was 6.4 mg/L or 457 µM, about 46 times the threshold for surface water in North Queensland catchments and equivalent to a land clearing rate close to a 100%. However, comparison of groundwater and surface water nutrient concentrations may not be valid. A concentration of 0.55 mg/L of nitrate as NO3 was found at Crystal Falls on the Douglas River in July 2006 (NRETA, unpublished data), or 39 µM, about 4 times the threshold value for 20% of North Queensland catchments being in pristine condition.



Figures 4.49 a - c (a) Exponentially increasing total sediment export (t/y) for North Queensland catchments with decreasing percentage of the catchment in pristine condition (note threshold at 20% pristine level). Open circle data point is NE Cape York Peninsula catchment outlier (see text). (b) Exponentially (here a power curve) increasing DIN concentration (µM) with decreasing percentage of catchment in pristine condition (note similar 20% threshold as for total sediment exports). Data are from Figure 2 in Brodie (2002). (c) Exponentially increasing relationship between total phosphorus exports per unit area (log_{10}TP kg/ha) and total sediment export (log_{10}TS kg/ha). Data sourced from Moss et al 1992, Davis et al 1998 and Hamblin 2001.

Table 4.25 Nonlinear power relationship between estimated total sediment export (TS t/y) for North Queensland catchments in Table 4.2.4, and percentage of the catchment classified as pristine (%CP) with (a) and without (b) northeast Cape York Peninsula. Parameters were estimated by Maximum Likelihood (Gause-Newton method) and all were significant at P<0.001. In both equations the level of confidence is 95% (alpha = 0.05).

Equation	N	% R^2
(a) TS = 4228 %CP $^{-0.52}$	12	71
(b) TS = 4404 %CP $^{-0..59}$	11	90

Table 4.26 Percentage of the catchment in pristine condition (%CP) and corresponding estimates of dissolved inorganic nitrogen concentration (DIN µM) for selected North Queensland catchments (from Figure 2 in Brodie 2002).

Catchment	% Catchment pristine	DIN conc (µM)
Burnett-Kolan	2	72
Fitzroy	2	42
Pioneer-O'Connell	1.5	20
Burdekin-Haughton	2.5	7
Herbert	10	10.5
Tulley-Murray	15	13
Johnstone	24	8
Mulgrave-Russell	33	9
Barron	92	6
Mossman-Daintree	100	7
North-east Cape York	100	5

Table 4.27 Nonlinear power relationship between estimated dissolved inorganic nitrogen concentration (DIN µM) in selected North Queensland catchments and the percentage of the catchment in pristine condition (%CP). Parameters were estimated by Maximum Likelihood (Gause-Newton method) and all were significant at P<0.001. Level of confidence is 95% (alpha = 0.05).

Equation	N	% R^2
DIN conc = 1.60 %CP $^{-0.173}$	12	62
Final loss = 0.064		

Total phosphorus export (t/y) did not show a similar nonlinear relationship with %CP as for TS. However, because all variables were non-normal they were transformed ($\log_{10}$TP, arcsine √p, where p= %CP/100) and re-examined by linear regression analysis. Results show that whilst %CP just missed out being significant (Table 4.28a) catchment area (CA km^2) was significant (Table 4.28b), albeit with poor explanatory power (R^2=33%). The regression of $\log_{10}$TP export on $\log_{10}$TS exports expressed per unit area (kg/ha/y), however, was highly significant with greater predictive power (Table 4.29, R^2=76%).

The best model fit between TP and TS exports per unit area was a nonlinear exponential relationship (Table 4.30), explaining 93% of the data (Figure 4.49c). The transport of phosphorus from diffuse sources in catchments can occur in both dissolved and particulate form, with particulate phosphorus being carried by overland flow resulting from run-off and erosion (Davis et al 1998). Harris (2001a) suggested that land use change and reduction of forest cover also affects P exports because it is largely associated with the particulate load. Hence, the strong correlation between TP and TS exports is not surprising, but does not explain the weak correlation with the percentage of the catchment remaining in pristine condition.

Table 4.28a & b Regression summary of total phosphorus export ($\log_{10}$ TP t/y) for 12 North Queensland catchments and (a) percentage of the catchment in pristine condition (arcsine transformation, %CP_AS) and (b) catchment area ($\log_{10}$ CA). All variables entered into the joint multiple regression equation were non-significant indicating redundancy.

(a) R= 0.5423, adjusted R^2 = 22%, n= 12, P=0.068, SE regression = 0.434
(b) R= 0.6263, adjusted R^2 = 33%, n= 12, P=0.029, SE regression = 0.403

Variable	**B**	**SE B**	**P**
(a) Intercept	3.157	0.297	<0.001
%CP_AS	-0.895	0.438	=0.068
(b) Intercept	0.863	0.696	NS
$\log_{10}$ CA	0.436	0.171	=0.029

Table 4.29 Summary of linear regression of total phosphorus export per unit area ($\log_{10}$TP kg/ha/y) and total sediment export per unit area ($\log_{10}$TS kg/ha/y) for 12 North Queensland catchments.

R= 0.8837, adjusted R^2 = 76%, n= 12, P<0.001, SE regression = 0.084

Variable	**B**	**SE B**	**P**
Intercept	-0.763	0.165	<0.001
$\log_{10}$ TS export	0.343	0.057	<0.001

Table 4.30 Nonlinear exponential relationship between estimated total phosphorus export per unit area ($\log_{10}$ TP kg/ha/y) and total sediment export per unit area ($\log_{10}$TS kg/ha/y) for North Queensland catchments in Table 4.24. Parameters were estimated by Maximum Likelihood (Quasi-Newton method) and all were significant at P<0.001.

Equation	**N**	**% R^2**
$\log_{10}$ TP export = 0.022 + $e^{(-9.48 + 2.58 \log10 TS)}$	12	93
Final loss = 0.022		

The significant regression between TP export and catchment area may not be surprising given the association between total suspended sediment and phosphorus, especially fine suspended sediments. Wasson (1994) found that suspended sediment yields in Australian catchments (>0.01km^2) are positively and strongly correlated to catchment area, and a similar relationship was found here (see Figure 4.50b). Prosser et al (2001b) and Norris et al (2001) used the relationship found by Wasson (1994) as a basis for predicting suspended sediment budgets in order to examine regional variations in catchment sediment load. This is in contrast to the approach adopted here, whereby %CP is used to predict sediment and nutrient budgets for catchments. Unfortunately, %CP is negatively and significantly correlated to catchment area

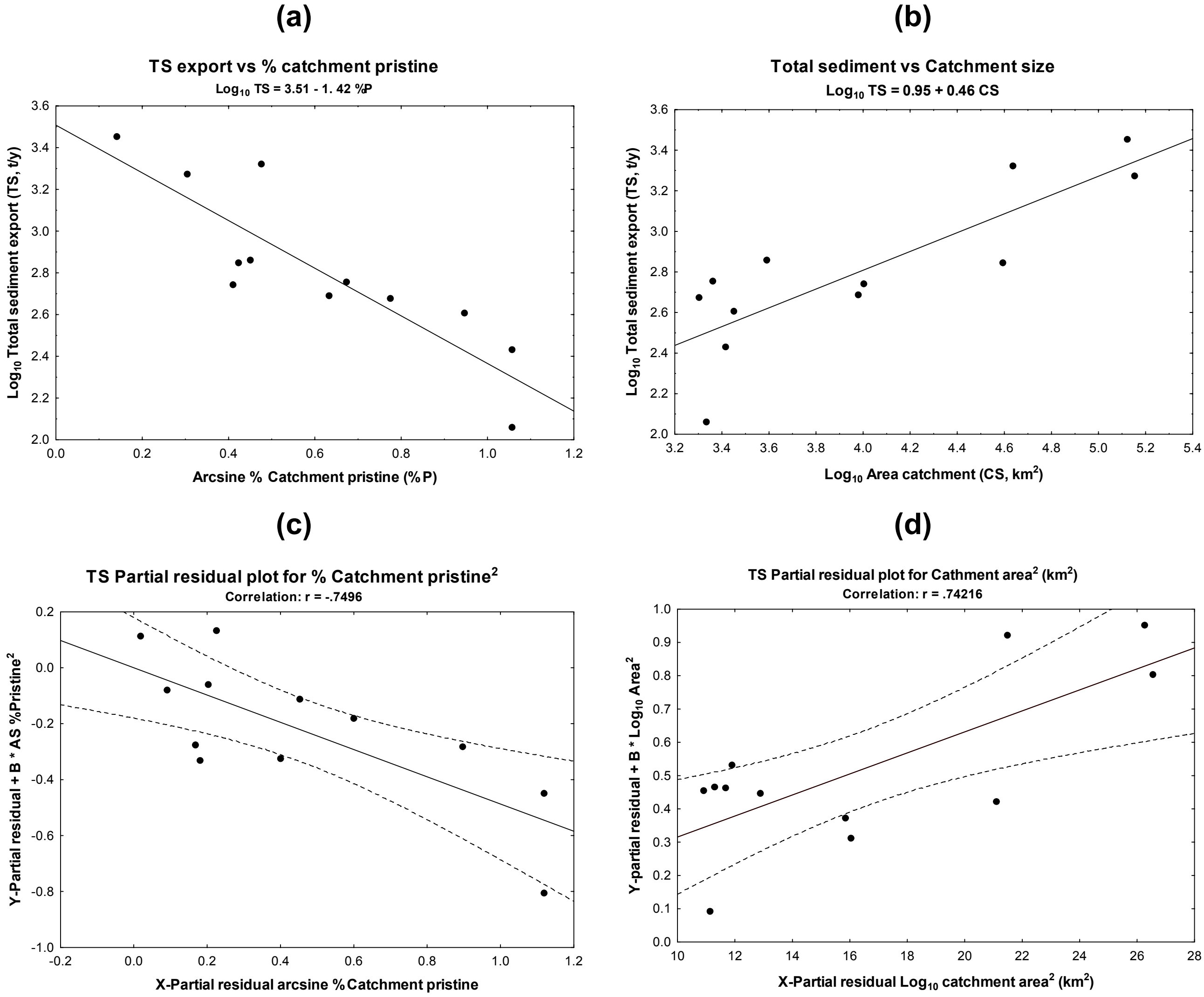
(R^2=65%, n=12, P<0.001; $\log_{10}$ & arcsine transformation), suggesting the two variables are confounded; larger catchments close to population centres may be more prone to land clearing pressures because they physically have more agricultural production capacity. Hence, additional regression analysis was undertaken in an attempt to 'factor-out' the confounding intercorrelation in order to select the best subset model. Model P-values were adjusted for Type I error using a Bonferroni correction (Zar 1984). Using transformed variables, multiple linear regression analysis shows that TS is significantly and negatively related to %CP (Table 4.31a, Figure 4.50a), and significantly and positively related to catchment area (Table 4.31b, Figure 4.50b), independently explaining large amounts of variability in the data (R^2=70% & 67%, respectively). However, when both variables are included in a combined multiple regression model both are significant entries, and the overall explained variance increases to 77% (Table 4.31c). This suggests that the intercorrelation between the two 'independent' predictor variables are 'partialled-out', meaning that one variable cannot completely substitute for the other. The partial residual plots supports the no redundancy argument, and it is noted that the spread and fit about the line of the residual data for %CP are better than that for catchment area (Figure 4.50 c & d, respectively).

The best subset model is the nonlinear power relationship predicting total sediment export from both the percentage of the catchment in pristine condition and catchment area (Table 4.32).

Table 4.31 Regression summary of total sediment export ($\log_{10}$ TS t/y) for 12 North Queensland catchments and (a) percentage of the catchment in pristine condition (arcsine transformation, %CP_AS), (b) catchment area ($\log_{10}$ CA) and (c) both variables in the equation. Nonlinear models were tested for significance by incorporating combinations of linear and quadratic polynomials of each independent variable in a step-down stepwise multiple regression. Regression critical P-values were adjusted for Type I error using a Bonferonni correction for the number of models tested.

(a) R= 0.8702, adjusted R^2 = 70%, n= 12, P<0.001, SE regression = 0.202
(b) R= 0.8369, adjusted R^2 = 67%, n= 12, P<0.001, SE regression = 0.225
(c) R= 0.9018, adjusted R^2 = 77%, n= 12, P<0.01*, SE regression = 0.187
* = Bonferonni adjusted regression P critical value for 0.001 = 0.001/4 = 0.00025. Adjusted regression P-value = P<0.01

Variable	Beta	SE Beta	B	SE B	P
(a) Intercept			3.508	0.139	<0.001
%CP_AS			-1.142	0.204	<0.001
(b) Intercept			0.954	0.389	=0.033
$\log_{10}$ CA			0.464	0.096	<0.001
(c) Intercept			2.511	0.313	<0.001
$\log_{10}$ CA^2	0.479	0.212	0.031	0.014	<0.05
%CP_AS^2	-0.490	0.212	-0.487	0.211	<0.05



Figures 4.50 a -d (a) Regression showing increasing total sediment export ($\log_{10}$TS t/y) with decreasing percentage of North Queensland catchments catchment in pristine condition (arcsine %CP). (b) Regression showing increasing total sediment export (TS, $\log_{10}$ t/y) with increasing catchment area ($\log_{10}$ CA km^2), after Wasson (1994). Partial regression residual plots for total sediment export ($\log_{10}$ TS t/y) and (c) percentage of the catchment in pristine condition (arcsine %CP) and (d) catchment area ($\log_{10}$ CA km^2). See table 5.23 for multiple regression equation. The intercorrelation between catchment area and %CP has been statistically factored out (or partialled out).

Table 4.32 Nonlinear power relationship between estimated total sediment export ($\log_{10}$TS t/y) and the percentage of the catchment in pristine condition (arcsine transformation, %CP_AS) and catchment area (CA) for North Queensland catchments (see Table 4.24). Parameters were estimated by Maximum Likelihood (Gause-Newton method) and all were significant at P<0.001. The solution was degenerate so the level of confidence is uncertain.

Equation	N	% R^2
$\log_{10}$ TS = 2.484 %CP_AS $^{-0.147}$ + 2 x10^{-6} CA	12	83
Regression residual mean square = 0.028		

4.4.4 Conceptual model – land clearing & water quality

Harris (1997) proposed a theoretical conceptual framework for modelling complex nonlinear events in aquatic ecosystems as coupled processes in catchments, water columns and sediments, and is based on paradigms for complex adaptive systems. He argued that, irrespective of complexity, it is possible to use similar models for each subsystem and that there are analogous processes in each differing only in scale (& see Harris 1998). Furthermore, Harris (1999a) argued that aquatic ecosystems may be a lot simpler than we think because higher order predictable behaviour most likely underpins complex systems behaviour at multiple spatial and temporal scales. Nevertheless, with respect to detailed properties of biological communities such as species composition, Harris (1994b) argued that prediction will necessarily be probabilistic because of ecological interactions, both horizontally between species and vertically within food chains. In contrast, Reckhow (1999) argued that nature is too complex to mimic with simple models and that, at some point, additional details will exceed our ability to simulate and predict within reasonable error levels. He further argued that, with respect to surface water quality assessments and predictions, an attractive alternative is to express complex behaviour probabilistically and recommended Bayesian probability networks. We adopt an approach that combines both views because it allows us to use the best available data and knowledge at hand to assess risks to surface water quality from the effects of land clearing.

Conceptual models developed for an initial ecological risk assessment of the Daly River catchment (DRCRG 2004, Begg et al 2001) identified land clearing as a potential key threat to the condition of riparian habitats, in-stream water quality, in-stream and floodplain environmental flows and, hence, the ‘condition’ of associated biotic habitats.

A new conceptual model was constructed (Figure 4.51) to guide the QERA quantitative risk assessment of the health of Daly River ‘surface water quality’ under different land clearing scenarios. It’s a ‘first cut’ model only and needs to be tested or validated in other tropical catchments against clearly defined performance criteria (see Rykiel 1996). The following three ecological assessment endpoints were used to assess potential ecological impacts in aquatic ecosystems: (i) total sediment export (TS); (ii) total phosphorus export (TP); and (iii) dissolved inorganic nitrogen (DIN) concentration. Measurement endpoints were TS and TP exports (t/y), and DIN concentration (µM), as predicted from the nonlinear relationship between these variables and the percentage of the catchment remaining in pristine condition (%CP) developed in Section 4.4.3 for North Queensland catchments. The model has the following characteristics and underlying assumptions:

1. For simplicity the different impacts of land clearing and land use activity on catchment water balance and, hence, recharge rates to streams and potential ‘knock on’ effects to ground water dependent ecosystems is ignored, although they can be incorporated into subsequent risk assessments. However, there would be a strong connection between this effect and water quality as indexed by agricultural soil and nutrient losses.

2. We assumed that the proportion of land cleared in a catchment integrates differences in types of land use activities that may affect water quality, and that the predictive models developed with Queensland data under their catchment conditions are generic across the tropics (ie high rainfall & flow conditions, similar grazing & cropping land uses). However, Drewry et al (2006) argued that the types of land use management and associated forms of nutrients needs to be specifically accounted in catchment-based empirical and process models.

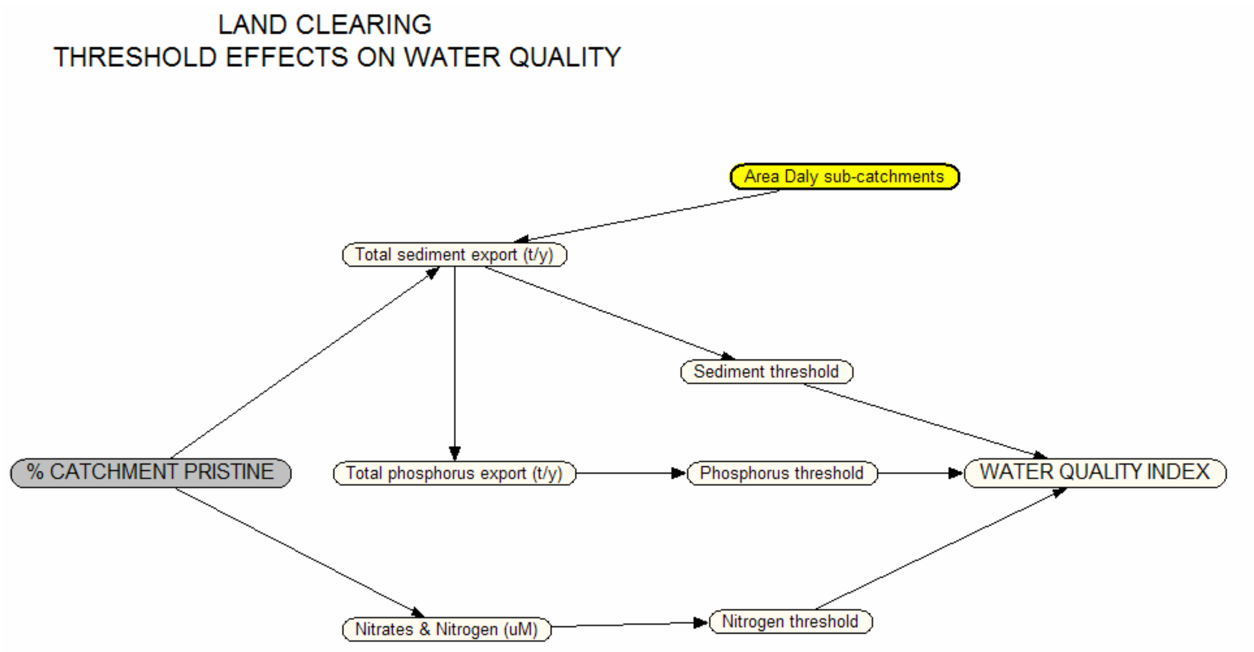


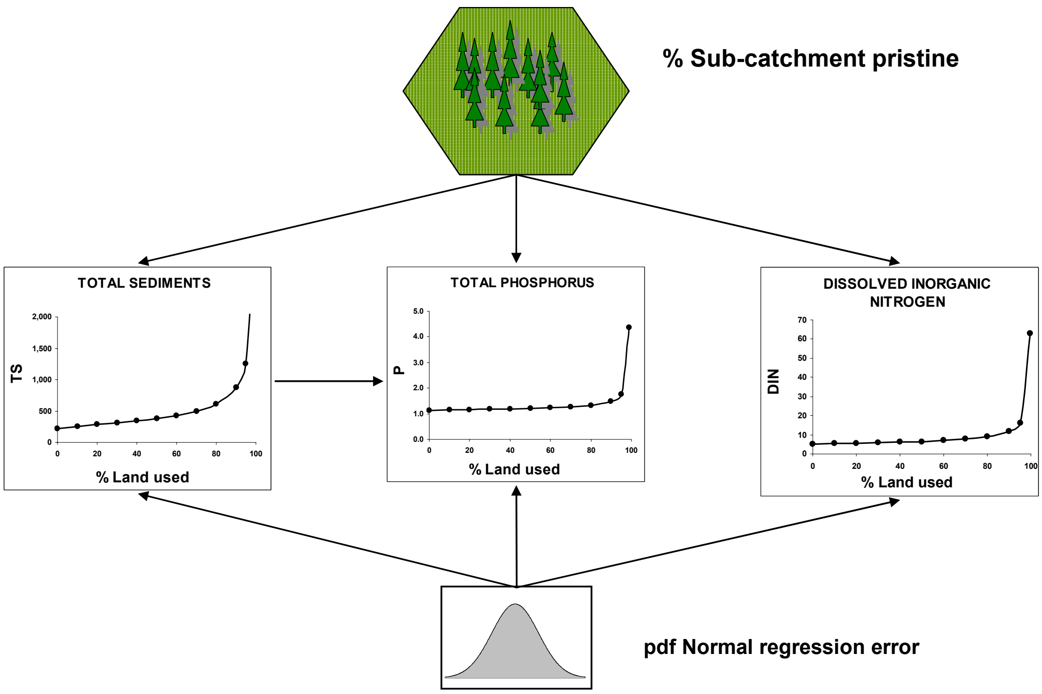
Figure 4.51 Conceptual model used to construct a Bayesian Network to assess the potential impact of land clearing in Daly River sub-catchments on In-stream health as indexed by surface water quality. Ecological (& social) endpoints are total sediment and phosphorus exports (t/y), and the concentration of Dissolved Inorganic Nitrogen (DIN, uM). Published North Queensland data were used to develop generic catchment-wide 'threshold' relationships between the amount of sediment and nutrients exported and the percentage of a catchment or sub-catchment remaining pristine (unused or uncleared).

3. We assumed that the catchment-based models developed for Queensland can be applied to sub-catchments of the Daly River because of the scaling relationship found by Wasson (1994) between total sediment export and catchment size. Additionally, Kinsey-Henderson et al (2005) used SedNet to model sources of sediment at sub-catchment scales for the Burdekin catchment in North Queensland, and dividing large catchments into sub-catchments has been used to varying degrees in Queensland export models.
4. We assumed that the 20% threshold effects levels for the percentage of catchments in pristine condition ascertained for TS and DIN in North Queensland catchments also applies to the Daly River catchment and its sub-catchments.
5. Although only 4% of the Daly River catchment was cleared by 2005, we assume that there is sufficient variation in the extent of land cleared in sub-catchments (0-59%, Table 4.33) to apply this assessment approach on a trial basis. Needless to say, model predictions of sediment and nutrient exports would need to be validated at some stage by comprehensive monitoring data at sub-catchment surface water exit ways.

The next stage of the risk assessment involves developing stochastic process models for each ecological assessment endpoint in the conceptual model, using the nonlinear regression models developed in Section 4.4.3. Models were then used to ultimately underpin the Bayesian Network (BN) developed to help assess surface water quality under existing and future land clearing scenarios.

4.4.5 Land clearing simulation based on sub-catchments of the Daly River catchment; model uncertainties & sensitivity analyses

The stochastic process models used to predict the effects of existing and simulated land clearing extents in the Daly River catchment on the export of total sediments (TS t/y) and phosphorus (TP t/y), and the concentration of dissolved inorganic nitrogen (DIN µM), are conceptually illustrated in Figure 4.52 and shows all model uncertainties and the link between TS to predict TP. To better conceptualise the impact of land clearing, the nonlinear regression equations developed in Section 4.4.3 are here converted to the percentage of land cleared in the catchment or sub-catchment for other forms of land use, rather than the percentage of land in pristine condition (ie 100% - %CP). Hence, the threshold 'land cleared' effects for TS and DIN is 80%. The extent of land cleared is entered as a constant in a single catchment or sub-catchment analysis, hence only model uncertainty is addressed in simulations.



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Simulations to assess impact of land clearing on export of sediments & phosphorus, & DIN concentration

Figure 4.52 Stochastic process models used to predict existing and simulated future effects of land clearing (% Land cleared) on export from sub-catchments in the Daly River of total sediments (TS, t/y) and total phosphorus (TP, t/y), and the concentration of Dissolved inorganic nitrogen (DIN µM/L). An 80% threshold effects valued is applied (see text). Nonlinear regression error terms were used to account for model uncertainty and TS is used to predict TP. The three statistical prediction models were used to support the Bayesian Network for surface water health, using a combined ecological and sociological endpoint (ie water quality with respect to ecosystem & human health).

Mean values were derived by Monte Carlo (MC) simulation (10,000 iterations) using @Risk™ software (Pallisade 2002b) and incorporating model uncertainty as outlined in Figure 4.52. Estimates of the proportion of land cleared for other uses are assumed to be error free. However, this is not true. Land use classification maps produced by the NT government do have associated errors. An evaluation of the magnitude of these errors has not yet been completed, but future risk modeling would need to incorporate them into the stochastic simulations as parameter variability.

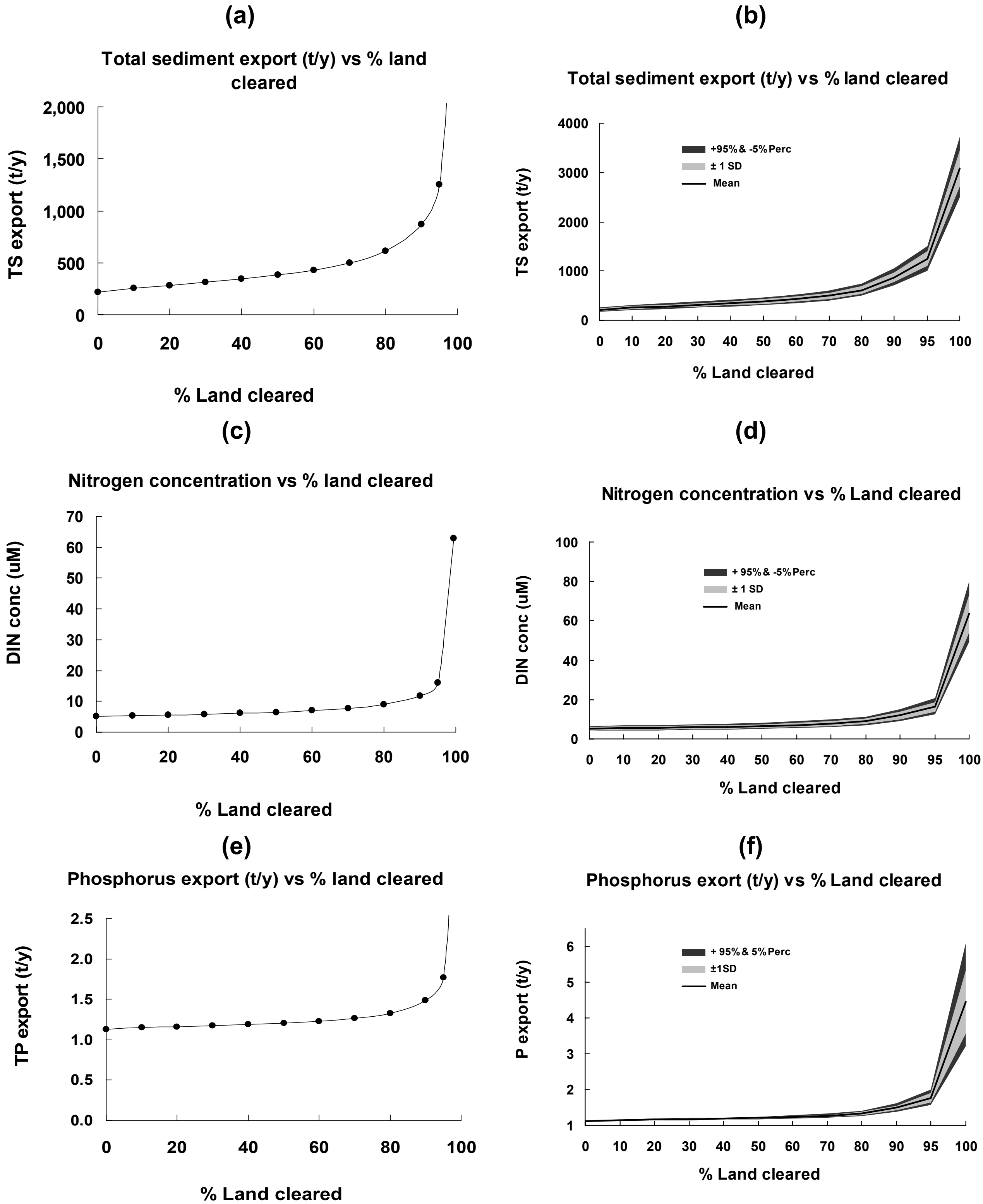
Simulated mean modelled TS export (t/y) from the Daly River catchment increased rapidly with increasing percentage of land cleared past the 80% level (Figure 4.53a). Uncertainty levels in predicted outputs are low because model error is low (Figure 4.53b). Once the catchment or sub-catchment size is selected the variable becomes a constant. Similar simulation results and low uncertainty levels were obtained for modelled DIN concentrations (Figure 4.53 c & d) and total phosphorus exports (Figure 4.53e & f). The results predict that by 2005, when 4% of the catchment had been cleared for other land uses, impacts to surface water quality and so aquatic ecosystem health would have been negligible.

4.4.6 Bayesian Network for land clearing in sub-catchments & water quality

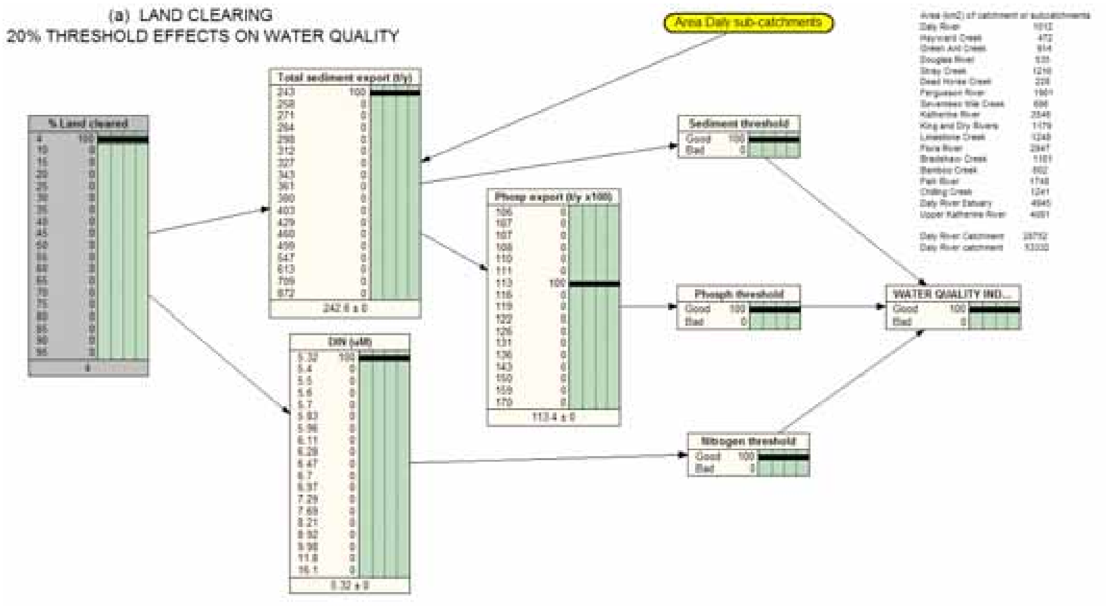
A Bayesian Network (BN) was constructed to establish a framework to examine future land clearing scenarios and different threshold effects levels on surface water quality, as indexed by modelled sediment and nutrient exports (Figure 4.54a&b & 4.55a&b). The BN can be applied separately to each sub-catchment or to the catchment as a whole, and is implemented with the node for area (km^2) used to calculate TS export and, hence, TP export. Threshold effects levels can be changed in the nodes for sediment and phosphorus indices of health, and DIN concentration. The following four scenarios were examined: (i) the 2005 4% extent of land cleared is applied to the whole Daly River catchment with an effects threshold set to 80% of land cleared, as suggested from North Queensland catchments; (ii) the 2005 59% extent of land cleared in the Green Ant sub-catchment is applied with the same 80% threshold; and (iii & iv) the effects threshold for land clearing is then lowered to 50% for both scenarios, and is the average threshold suggested by Harris (2001b) for Australian catchments. The overall water quality index for scenarios (i) and (ii) is ‘Good’ (Figure 4.54a & b, respectively), that for scenario (iii) remains ‘Good’ and, in contrast, scenario (iv) for the Green Ant sub-catchment slides to ‘Bad’ (Figure 4.55a & b, respectively). Whilst these results are obvious, the BN nevertheless provides the beginnings of a framework to encompass a range of more informative ecosystem cause-effect relationships, hopefully providing more responsive ecological assessment endpoints that can be used as an early warning system of catchment health.

4.4.7 Concordance of model predictions with risk ranks for sub-catchments derived from the spatial Relative Risk Model

The proportion of land cleared in a catchment or sub-catchment underpins all modelled outputs of sediments and nutrients used to assess in-stream water quality here and, hence, one component of the ecosystem health of surface waters. The area or proportion of land cleared also underpins the spatially-based Relative Risk Model (Chapter 3) applied to the whole catchment if classified as a threat to ecosystem health. Hence, there should be a strong link between the semi-qualitative risk approach adopted in the previous section and the more detailed QERA approach used here that is dependent on more data and, hence, has more associated levels of uncertainty. We therefore examine the concordance of modelled sediment



Figures 4.53 a - f (a) Simulated increase in mean modelled total sediment export (t/y) from the Daly River catchment with increasing percentage of land cleared for land use using the equation in Table 5.31, clearly showing the 80% threshold effects level. (b) As for (a) but with uncertainty levels illustrated using one standard deviation (SD) about the mean trend and the + 95% and – 5% percentiles. (c) Simulated increase in mean modelled DIN concentration (µM) from the Daly River catchment with increasing percentage of land cleared using the equation in Table 5.26, clearly showing the 80% threshold effects level. (d) As for (c) but with uncertainty levels illustrated using one standard deviation (SD) about the mean trend and the + 95% and – 5% percentiles.



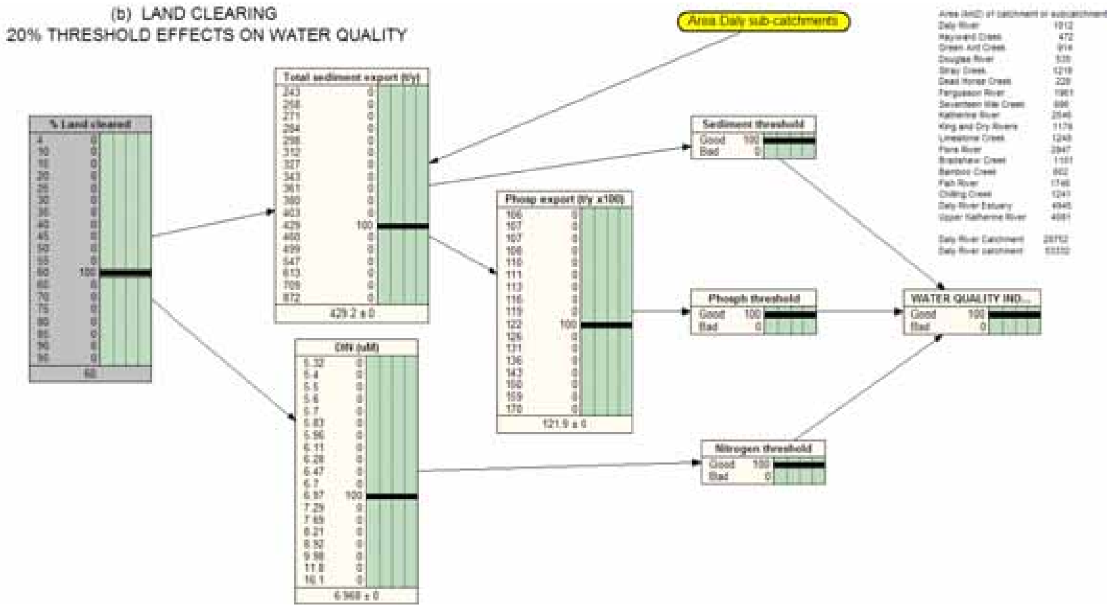


Figure 4.54a & b Bayesian Network for the potential effects of land clearing in the Daly River catchment, or sub-catchments, on water quality and hence 'In-stream Health' as indexed by modelled sediment and phosphorus exports, and nitrogen concentration (DIN). An 80% threshold effects level for land clearing is applied to (a) the whole Daly catchment with a 4% clearing rate (2005) and (b) to the Green Ant sub-catchment with a 60% clearing rate (2005). The water quality index for both scenarios is 'Good'.

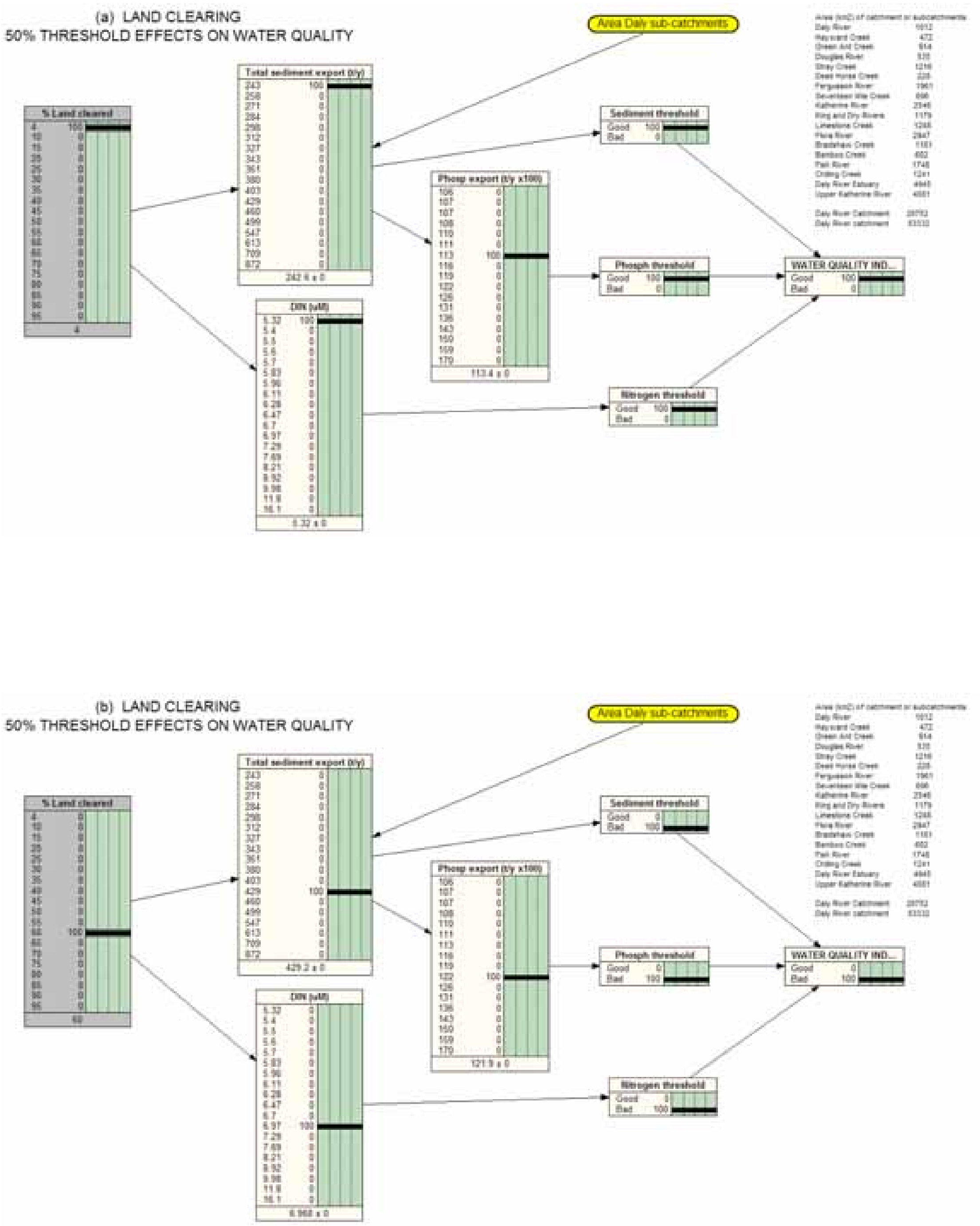


Figure 4.55a & b Bayesian Network for the potential effects of land clearing in the Daly River catchment, or sub-catchments, on water quality and hence 'In-stream Health' as indexed by modelled sediment and phosphorus exports, and nitrogen concentration (DIN). A 50% threshold effects level for land clearing is applied (as per Harris 2001b) to (a) the whole Daly catchment with a 4% clearing rate (2005) and (b) to the Green Ant sub-catchment with a 50% clearing rate (2005). The water quality index for scenario (a) is 'Good' and that for (b) 'Bad'.

and nutrient water quality parameters with the total relative risk ranks derived for sub-catchments. Table 4.33 summarises all comparative statistics for the two methodologies. The Green Ant sub-catchment is highlighted in red bold because it had the highest extent of land cleared in 2005 at 59%. All key variables (ie % of the sub-catchment remaining in pristine condition, TS, TP & DIN) used or modelled in the quantitative risk assessment for land clearing reported here are strongly correlated (Figure 4.56a-d) with the semi-qualitative Relative Risk Ranks derived for each sub-catchment. Sub-catchment risk ranks derived from spatial estimates of land cleared for other land uses are therefore concordant with modelled losses of sediments and nutrients.

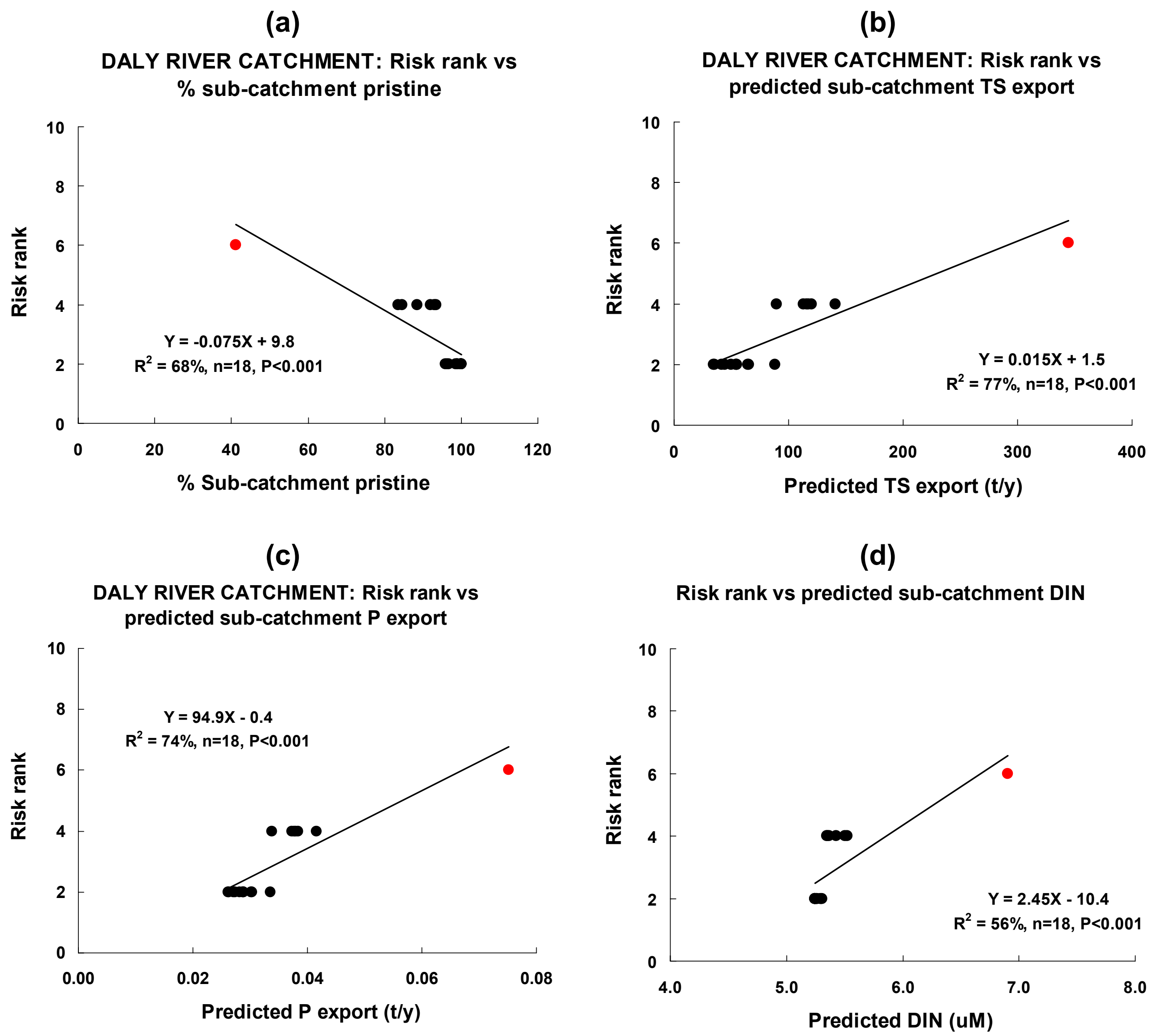


Figure 4.56 a-d Comparison of the spatially-derived relative risk ranks for sub-catchments in the Daly River using the Relative Risk Model (Bartolo & van Dam 2006) and modelled water quality indicators used here. Relative risk rank is plotted against: (a) percentage of the sub-catchment in pristine condition; (b) predicted total sediment export (TS t/y); (c) predicted total phosphorus export (TP t/y); and predicted DIN concentration (μM). Sub-catchment risk ranks derived from spatial estimates of land cleared are strongly related to complex modelled losses of sediments and nutrients.

Table 4.33 Summary of Daly River catchment sub-catchment statistics (area, % land used, % land pristine, risk region & rank) and modelled estimates of total sediment and total phosphorus exports (t/y) as predicted by sub-catchment area and the percentage of the catchment in pristine condition. The Green Ant sub-catchment is highlighted in bold and had 59% of land used by 2005.

Catchment	Catchment ID	Sub-catchment	Risk Region	Sub-catchment area (Km^2)	% Land used	% Pristine	Risk Rank RRM	Model TS(t/y)	Model DIN (µM)	Model P (t/y)
8	21	Daly River	1	1012	6.9	93.1	4	89	5.35	0.034
6	22	Hayward Creek	2	472	4.0	96.0	2	65	5.30	0.030
4	23	Green Ant Creek	3	914	58.9	41.1	6	344	6.91	0.075
5	28	Douglas River	4	535	15.4	84.6	4	117	5.50	0.038
12	33	Stray Creek	5	1216	11.5	88.5	4	116	5.43	0.038
22	30	Dead Horse Creek	6	228	3.2	96.8	2	54	5.29	0.029
9	2	Fergusson River	7	1961	1.5	98.5	2	65	5.26	0.030
18	1	Seventeen Mile Creek	8	696	0.0	100.0	2	35	5.24	0.026
24	8	Katherine River	9	2546	8.0	92.0	4	113	5.37	0.037
25	9	King and Dry Rivers	10	1179	1.0	99.0	2	55	5.26	0.029
31	6	Limestone Creek	11	1248	16.5	83.5	4	141	5.52	0.042
28	26	Flora River	12	2947	3.6	96.4	2	88	5.30	0.034
26	24	Bradshaw Creek	13	1181	1.0	99.0	2	55	5.26	0.029
16	18	Bamboo Creek	14	602	0.0	100.0	2	35	5.24	0.026
17	19	Fish River	15	1748	0.0	100.0	2	42	5.24	0.027
14	35	Chilling Creek	16	1241	0.1	99.9	2	44	5.24	0.027
3	17	Daly River Estuary	17	4945	6.5	93.5	4	120	5.34	0.038
2	11	Upper Katherine River	18	4081	0.0	100.0	2	50	5.24	0.028

4.4.8 Discussion & recommendations

A key feature of our results is that our chosen assessment endpoints for surface water quality are sediment and nutrient export loads that exhibit extreme threshold effects level with the extent of land cleared for land use. North Queensland catchments apparently exist either in one of two states; one with low sediment and nutrient export loads and low to moderately high levels of land clearing, or one with high sediment and nutrient export loads and very high levels of land clearing. The switch between the two states is very abrupt, and the threshold between them occurs when about 80% of the catchment is modified for land use. This binary condition is analogous to the two-state classification that Harris (2001b) used to describe degraded and relatively pristine aquatic ecosystems in Australia. In drawing an analogy between ecological and socio-economic systems, Kinzig et al (2006) suggested that most accounts of thresholds between alternate regimes involved a single dominant shift defined by one slowly changing variable in an ecosystem, and aptly describes land use within catchments. Nevertheless, given the exorbitant rehabilitation costs at landscape scales, and long lead times required, the effects of extensive catchment modifications are essentially irreversible. Additionally, most of the catchment would have to be cleared of native vegetation cover before a significantly measurable, and for all intents and purposes irreversible, change is detected. Hence, models that predict sediment and nutrient loads in relation to land cover *per se* would be useless as early warning systems in order to change land use policy before aquatic ecosystems collapse. Another conceptual problem with our 'export out of the catchment' model, although relevant to coastal and offshore impacts (& highly relevant to the Great Barrier Reef), is the centennial-scale retention times of sediments (& possibly some nutrients) in catchments, as highlighted by the study by Wasson (1994). In terms of regional impact therefore, equal importance should be placed on impacts on aquatic ecosystems within catchments that may trap sediments and nutrients. For example, freshwater wetlands retain sediments and nutrients and, hence, may increase surface water quality to the benefit of downstream estuarine and marine ecosystems and their users (Johnston 1991). Johnston et al (1990) examined the cumulative effect of wetlands on stream water quality and quantity and found that wetlands were more effective in removing suspended solids, total phosphorus and ammonia during high flow periods, but were more effective at removing nitrates during low flow periods. In addition to their intrinsic value, the critical role of the Daly River floodplain and associated wetlands in influencing downstream water quality and, hence, in-stream, riparian and estuarine ecosystems, needs to be understood. The Daly River floodplain is under serious threat from wetland weeds (Section 4.2), and wetlands are generally the first habitats to disappear under developmental pressures.

The surrogate physico-chemical endpoints used here to assess surface water quality, therefore, should be extended to encompass a wide range of more useful biological indicators given that some knowledge has already accumulated since 2000, and will continue to accumulate with focussed investment through TRaCK and other research efforts. More direct links between nutrients and sediments to the condition of biological communities, as represented by a range of trophic levels, is required. For example, the condition of: *Vallisneria nana* (Rea et al 2002); periphyton and phytoplankton abundance (Townsend et al 2002), benthic algae, *Spirogyra* (Townsend and Padovan 2005); benthic diatom assemblages (Townsend & Gell 2005); and of course populations of the iconic pig nose turtle (Georges et al 2002, 2003).

Additionally, there are obviously other key ecological processes in catchments that may influence water quality not captured in our risk assessment, and a few are mentioned here. The direction and magnitude of all conceptualised (hypothesised) land use affects on aquatic ecosystems will depend strongly on land use type. For example, cleared native vegetation may be replaced with annual pastures, horticulture crops or commercial forests. The non-native vegetation cover classes may either ameliorate or accelerate negative land clearing effects, and this needs to be teased out by careful study. Fire is a key ecological driver of landscape processes in both woodland and floodplain ecosystems in the tropics. Townsend and Douglas (2000) examined the effect of different fire regimes on stream water quality, water yield and export coefficients of some nutrients, sediments and metals at a site in the South Alligator River catchment, Kakadu National Park. They found that only suspended sediments increased with the extent of late dry season fires. Moliere et al (2004) also found that dry season fires increased mud load and, hence, turbidity, in Ngarradj catchment, Kakadu National Park, because of the loss of ground cover. However, the role of floodplain fires on floodplain and in-stream water quality, and needless to say landscape processes in general, needs to be elucidated.

In the context of the role of wetlands as nitrogen traps mentioned above, the results for elevate nitrate concentration in groundwater in the Douglas River sub-catchment, and in surface water in the lower reaches of the Douglas River, are surprising in that they are very high. Hence, they are a potential future water quality issue for the downstream Douglas and Daly rivers as flagged by Schult and Metcalfe (2006). The maximum concentration of nitrate from all three bores in the Douglas River sub-catchment was 6.4 mg/L or 457 µM, about 46 times the threshold for surface water in North Queensland catchments and equivalent to a land clearing rate close to a 100%. However, comparison of groundwater and surface water nutrient concentrations may not be valid. Nevertheless, a concentration of 0.55 mg/L or 39 µM of nitrate as NO3 was found at Crystal Falls on the Douglas River in July 2006 (NRETA, unpublished data), about 4 times the threshold value for 20% of North Queensland catchments remaining in pristine condition. The maximum value reported in Brodie (2002) for North Queensland catchments was 72 µM for the Burnett-Kolan catchment, followed by the Fitzroy with 42 µM, both with 98% of land cleared. Nitrate contamination of aquifers is a global agricultural problem with adverse environmental, economic and health effects. Of all agricultural contaminants it is the most widespread in exceeding drinking water standards in groundwater (Wassenaar et al 2006). The cause of the elevated levels of nitrates in the surface and groundwater of the Douglas River sub-catchment needs to be identified because it has the potential to impact of future water quality *in situ* and downstream in the Daly rivers. The source may be natural (eg bat droppings in limestone caves), or anthropogenic (eg agricultural fertiliser inputs &/or slow release of soil nitrates after land is cleared). It may beneficial, therefore, for comparative purposes to sample groundwater nitrate concentrations in adjacent sub-catchments influenced by the Tindall Limestone aquifer and with relatively pristine and heavily cleared land covers (eg Green Ant). Additionally, N-isotopic ratio studies could be conducted at the same time to identify nitrate sources.

4.5 Overall conclusions, summary & recommendations

The first step in an ecological risk assessment is to develop a conceptual model with stakeholders that capture the multiple threats and their pathways to multiple assets. The second step is to prioritise or rank them all based on qualitative and/or quantitative risk analysis. The most important step in ranking multiple risks occur between the conceptual model and qualitative assessments that filter lesser or even trivial risks in order to focus time consuming quantitative effort on more significant risks. A critical component of quantitative risk analysis is to develop, where data allow, predictive ecological process models that link measurement endpoints to the level of threat being assessed. Ecological models are, however, a gross simplification of a complex reality, and the conundrum between model complexity, predictability and utility has been around a long time. In Section 4.4.4 we highlighted two very different views about the feasibility and, hence, utility of developing predictive ecosystem models for complex, nonlinear and non-equilibrium systems such as aquatic ecosystems. One view is that better knowledge is needed for better management and that, despite the complexities of aquatic ecosystems, it is possible to develop simple predictive models because of common underlying physical processes that characterise complex adaptive processes independent of scale. In contrast, the other view is that ecosystems are far too complex to model and predict with any degree of certainty, and that a probabilistic approach to water quality management, such as Bayesian Belief Networks, would be more useful. However, whilst the views of Harris (1997) and Reckhow (1999) seem to represent two opposite points of a continuum amongst aquatic ecologists, they may not be as dichotomous as first appears. Harris argues for greater knowledge for better prediction, and the Bayesian approach of Reckhow is dependent on *a priori* probabilities or knowledge, suggesting that both approaches should converge.

We adopted an approach that combines both views because it allows us to use the best available data at hand to assess ecological risk. This is a key part of our selection process to choose the best quantitative approach for the issue at hand, even if rigorous uncertainty analysis at the end of the day suggests that the predictive ecological or physico-chemical model should be discarded in favour of simple frequency statistics of exposure and effects, or even a qualitative risk rank. For example, where data allow we develop ecological process models to predict the likely impact of a threat to the chosen measurement endpoint. However, there are huge calibration and validation problems associated with modelling complex systems from time series data (Young 1998), and so we use a simplistic approach that appears suited to Bayesian Networks (BNs) and the convergence of the two modelling views expressed above. We develop predictive statistical models, such as multiple linear regression or nonlinear regression equations, rather than complex mechanistic models. This data-based statistical approach allows development of stochastic process models that: quantifies cause-effect hypotheses identified in the conceptual model; accounts for both model uncertainty and intrinsic variability in key driving variables used to predict outcomes; and incorporates feedback loops, interactions between variables and the relative magnitude and direction of effects. However, complex ecosystem models that couple many interacting nonlinear functions together are generally highly sensitive to uncertainty in both conceptual and process model structures, and with respect to the latter, variability in model parameters. Hence, complex ecosystem models more often than not produce unstable and uncertain predictions and, if so, should be discarded and the conceptual model re-assessed. Methods to assess the effect of alternative conceptual model structures on ecological prediction, however, are in their infancy and need further development. Whilst Bayesian Networks based on expert opinion and probabilities are often used to fill empirical information gaps, they have severe practical limitations in themselves because of the need to populate (& often invent) large Conditional Probability Tables (CPTs). Whilst we attempted to avoid large unwieldy CPTs of intersecting child nodes by replacing them with equations that use outputs (eg other equations, pdfs or constants) from parent nodes as input variables, the constraint often still persisted. This problem is confounded by the need to incorporate feedback cycles within dynamic biophysical and decision making systems. Dambacher et al (2003) proposed an innovative methodology to screen conceptual ecosystem models whereby a qualitative analysis of sign directed graphs is embedded into the probabilistic framework of a Bayesian Network. The approach incorporates the effects of feedback on the models' response to a sustained change in one or more of its parameters, and may provide an efficient means to explore the effect of alternative model structures. In addition, the method is amenable to stakeholder input and may mitigate the cognitive bias in expert opinion. Other BN advocates suggest alternative pathways to utility. For example, Thomas et al (2005) argued that there is a clear need to simplify and focus BN modelling tasks, and that the best way to accomplish this daunting task is to minimise the number of factors representing threats that require parameterisation by using a bottom-up prioritisation process rather than the usual top-down process employed by scientists. They concluded that comprehensive expert input was vital in the conceptual model simplification process, and we agree.

Irrespective of the issues outlined above, where data do not allow development of ecosystem or population level models we use frequency statistics to estimate probabilities of exposures and effects, and then incorporate these into the BNs. Where there are no local data sufficient for either approach, we resort to use of knowledge in other locations (eg North Queensland) or the subjective belief of local experts, which essentially is the *a priori* Bayesian approach.

In summary, our quantitative ecological risk assessment is underpinned by our stochastic process models that attempt to link assessment endpoints to identified threats and, which account for uncertainty and variability in predictions. These process models were then embedded into BNs to guide decision making in the face of explicit and transparent uncertainties. The underlying premise of our approach is that better scientific knowledge of ecological processes will ultimately increase predictive power of the models and, hopefully, reduce uncertainty in risk assessments. The approach marries future knowledge aspirations with the present day reality of needing to manage ecological risk. It caters also to most natural resource management situations at hand, whereby if essential knowledge or understanding is lacking then the BN offers default options in the form of risk probabilities and/or subjective belief. The default values can be updated when new or increased knowledge manifests, and this approach underpins the adaptive management approach (Walters 1986).

Nevertheless, whilst we account for uncertainty and variability in our model predictions, the models themselves are not validated for the Daly River or other local NT rivers. Oreskes et al (1994) argued that it is impossible to validate numerical models in the earth sciences and, similarly, Hilborn and Mangel (1997) argued that models cannot be validated because alternative models are just options with varying degrees of belief. Rykiel (1996) states that a model is validated when it is acceptable for its intended use because it meets specified performance criteria. Hence, validation of a model is not that it is 'true', but that it has some form of utility (Levins 1966).

The short time series of data used for the magpie goose nest model, and the barramundi catch and population models, preclude model validation using boot-strapping techniques (ie re-sampling data with replacement, see Hilborn & Mangel 1997). Similarly, the limited number of spatial replicates (n=12 tropical catchments in North Queensland) used in the sediment and nutrient export models precludes validation through random selection of sub-sets of the total data set. Our preference, however, is to treat model validation as a separate future exercise by testing the application of the prediction equations in a number of independent catchments in the NT and elsewhere in the northern tropics. This would be true validation. For example, the Elizabeth and Darwin rivers are ideal validation candidates because of their long water quality time series. The validation process would allow improvements also to both conceptual and empirical models.

Key ecosystem types in the Daly River catchment not addressed in our quantitative risk assessment are riparian (see Catterall 1993) and estuarine habitats. These are major but deliberate omissions due to the limited scope of the QERA component of this study.

Nevertheless, they should be included in future ecological risk assessments because of the connectivity between ecosystems from catchment to coast to sea (Jupiter et al 2003). Griffin (1985) highlighted the importance of mangrove and coastal wetland communities in the NT to three commercial fisheries, in particular barramundi. Additionally, Rogers (1990) highlighted the severe degradation of coral reefs along tropical shorelines as a result of sedimentation from dredging and terrestrial runoff from developed catchments.

Land clearing has the potential to increase surface flow and reduce groundwater recharge rates in the Daly River catchment (Wilson et al 2006a, b; Knapton 2006), and O'Grady et al (2002a, b) and Lamontagne et al (2005) argued from their studies of tree water use by riparian vegetation in the Daly River catchment that they are ground water dependent ecosystems (GDE) that require special focus because of their susceptibility to land use impacts as highlighted by Murray et al (2006), and Boulton and Hancock (2006), in other riparian studies.

For this study only surface flow extraction is simulated in our QERAs. Our rationale is that the latest water balance estimates for the Daly River catchment (Jolly 2002a; O'Grady et al 2002a) indicated that current ground water extraction is likely less than 1% of overall supply. However, Lamontagne et al (2005) argued that a decline in the regional water table as a result of groundwater pumping may affect the health of riparian zone vegetation in the Daly River because groundwater use by vegetation is significant during the dry season. Additionally, dry season groundwater extraction may increase in future with increased land use, particularly horticulture, and riparian ecosystems in Top End catchments may be highly dependent on groundwater (Groundwater Dependent Ecosystems, see O'Grady et al 2002a) and so affected disproportionately to their extent in the catchment. Hence, groundwater extraction scenarios should also be important components of future risk assessments, particularly with respect to risk to the health of riparian ecosystems not addressed explicitly in our QERA. Future risk assessments targeted to address the potential impacts of land use change on groundwater dependent ecosystems, such as riparian communities, would need to specifically examine both wet and dry season flow extraction scenarios, including groundwater extractions in the dry season (see Clifton & Evans 2001). Groundwater extraction scenarios should be important components of future water extraction scenarios, and wet and dry season flow extractions should be assessed as two coupled components linked to groundwater. Pusey and Arthington (2003) reviewed the importance of the riparian zone to the conservation and management of freshwater fish, and argued that greater attention needs to be focussed on the linkages between fish and riparian systems in order to rehabilitate degraded stream environments and to prevent further loss of fish populations in northern Australia (& see ISRS 2004).

A major limitation of our current QERA is that, with its focus on fishing and ecological values, we have excluded consideration of key flow values of local Indigenous people. Jackson et al (2005) highlighted the importance of recognising Aboriginal rights and values in river research and management, particularly for tropical rivers in the NT where Indigenous people are custodians to 85% of the coastline and about 50% of the catchments. Aboriginal perspectives and knowledge were not canvassed in earlier studies in the Daly River Region for the National River Health Environmental Flow Initiative, and which entail the original environmental flow recommendations of Erskine et al (2003). Subsequently Jackson (2004) made nine recommendations on Aboriginal perspectives on land use and water management for the Daly River Region, most of which relate to environmental water requirements. Erskine et al (2004) recognised this shortcoming and suggested that the relevant recommendations of Jackson, summarised below, be adopted.

1. Investigate models for a region-wide negotiated settlement of management arrangements.
2. Review the adequacy of current arrangements for Aboriginal participation in catchment management processes and examine structures established in other regions.
3. Investigate options for declaring a river park along the Daly River under joint management arrangements.
4. Examine water resource management institutions to ensure water efficiency.
5. Facilitate further consultation and negotiations around the current principles underpinning water allocation planning and environmental flows research.
7. Establish a process to elicit more comprehensive qualitative, and where possible, quantitative understanding of Aboriginal social values
8. Ensure that social and cultural impacts are monitored.
9. Address the concerns related above regarding sedimentation of the Daly and Katherine rivers and the environmental impacts of recreational boating.

Recommendation 6 is not listed but could be dealt with appropriately by the Aboriginal Sacred Sites Authority. We argue that a Bayesian Belief Network could accommodate most Indigenous cultural values because it would implicitly recognise the value of expert knowledge from other domains, and therefore gives respect and weighting to them. Whilst western biophysical knowledge implicitly excludes Indigenous knowledge, the BN approach is a rare manifestation that can be used to combine the two culturally distinct knowledge systems and, hence, provide a mechanism of empowerment in decision making processes. Bayliss et al (1996) described how Aboriginal communities in central Arnhem Land manage their resources using the two knowledge domains of western science and traditional beliefs and customs. The Djelk land management rangers had adopted as their logo two water lily roots representing the two knowledge domains, and coming together in a fish trap. It’s a powerful symbol of joining forces across cultures to manage land and water issues, and is directly analogous to a Bayesian Network. The expert views of other key non-technical stakeholders could be adequately accommodated also, as demonstrated by the BNs developed for recreational and commercial barramundi fishing.

4.6 References

Allan JD 2004. Landscapes and Riverscapes: The Influence of Land Use on Stream Ecosystems. *Annu. Rev. Ecol. Evol. Syst.* 35, 257-84.

ANCA: Australian Nature Conservation Agency 1996. *A Directory of Important Wetlands in Australia Second Edition*. ANCA, Canberra.

AS/NZS 2004a. *Risk Management*. AS/NZS 4360:2004, Standards Australia International, Sydney.

AS/NZS 2004b. *Risk Management Guidelines Companion to AS/NZS 4360:2004*. HB 436:2004, Standards Australia International, Sydney.

Baran E & Cain J 2001. Ecological approaches of flood-fish relationships modelling in the Mekong River Basin. P. 20-27 in Koh H.L. & Abu Hasan Y. (eds.) *Proceedings of the National workshop on Ecological and Environmental Modelling*, 3-4 September 2001, Universiti Sains Malaysia, Penang, Malaysia.

Baran E & Jantunen T 2004. Stakeholder consultation for Bayesian Decision Support Systems in environmental management. Proceedings of the Regional Conference on Ecological and Environmental Modelling (ECOMOD 2004). 15-16 September 2004, Universiti Sains Malaysia, Penang, Malaysia.

Bartley R, Henderson A, Prosser IP, Hughes AO, McKergow L, Lu H, Brodie J, Bainbridge Z & Roth CH 2003. Patterns of Erosion and Sediment and Nutrient Transport in the Herbert River Catchment, Queensland. CSIRO Land and Water Consultancy Report for Herbert River Catchment Group.

Bartolo, R.E. and van Dam, R.A. 2006. Tropical Rivers Risk Assessment, Australian National Committee on Irrigation and Drainage Conference, Darwin, October 15-18, pp.44-47.

Bayliss P, Bartolo R, de Lestang P, Saalfeld K & Wasson R in preparation (a). Decadal trends in river flow in the Northern Territory and synchronous trends in aquatic ecosystems: 20-year period trends in magpie geese and barramundi populations.

Bayliss P, Ferdinands K, Walden D, Boyden J & Salau B in preparation (b). Preliminary risk assessment of para grass on Magela floodplain, Kakadu National Park: implications for costs of control.

Bayliss P, Walden D, Winderlich S & Petty A in preparation (c). Decadal trends in the extent of late dry season fires across Kakadu National Park in relation to early dry season burns, rainfall and interactions between ENSO-PDO climate systems: implications for management.

Bayliss P, Bartolo R & Wasson RJ. 2007. Decadal trends in rainfall, stream flow and aquatic ecosystems in the wet-dry tropics of the Northern Territory: influence of the ENSO-IPO interaction. Abstract for Greenhouse 2007 conference, Sydney, 2-5th October 2007.

Bayliss P, van Dam R, Boyden J and Walden D 2006. Ecological risk assessment of Magela floodplain to differentiate mining and non-mining impacts. In: Evans KG, Rovis-Hermann J, Webb A & Jones DR (eds). *Eriss research summary 2004–2005*. Supervising Scientist Report 189, Supervising Scientist, Darwin NT.

Bayliss P, Schlusser KM & Yibarbuk D 1996. The Management of Wetlands in Arnhem Land by Indigenous People. *Wetlands Australia* 4, 2-3.

Bayliss P & Yeomans KM 1990a. Use of low-level aerial photography to correct bias in aerial survey estimates of Magpie Goose and Whistling Duck density in the Northern Territory. *Aust. Wildl. Res.* 17, 1-10.

Bayliss P & Yeomans KM 1990b. Seasonal distribution and abundance of Magpie Geese *Anseranas semipalmata* Latham, in the Northern Territory, and their relationship to habitat, 1983-86. *Aust. Wildl. Res.* 17, 15-38.

Bayliss P 1989. The population dynamics of magpie geese in relation to rainfall and density: implications for harvest models in a fluctuation environment. *Journal Applied Ecology* 26, 913-924.

Beamish RJ, Noakes DJ, McFarlane GA, Klyashtorin L, Ivanov VV & Kurashov V 1999. The regime concept and natural trends in the production of Pacific salmon. *Can. J. Fish. Aquat. Sci.* 56, 516-526.

Beamish RJ, Neville C-EM & Cass AJ 1997. Production of Fraser River sockeye salmon (*Oncorhynchus nerka*) in relation to decadal-scale changes in the climate and the ocean. *Can. J. Fish. Aquat. Sci.* 54, 543-554.

Beamish RJ, Neville C-EM & Thompson BL 1994. A Relationship between Fraser River Discharge and Interannual Production of Pacific salmon (*Oncorhynchus* spp.) and Pacific herring (*Clupea pallasi*) in the Strait of Georgia. *Can. J. Fish. Aquat. Sci.* 51, 2843-2855.

Begg GW, van Dam RA, 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, Supervising Scientist, Darwin NT.

Berry W, Rubinstein N & Melzian B 2003. The Biological Effects of Suspended and Bedded Sediment (SABS) in Aquatic Systems: A Review. Internal Report, United States Environmental Protection Agency, Office of Research and Development National Health and Environmental Effects Laboratory, Atlantic Ecology Division. Duluth, MN, USA. 58 pp.

Bevan K 2000. On model uncertainty, risk and decision making. *Hydrological Processes* 14, 2605-2606.

Billington K 2005. *The River Murray and Lower Lakes Catchment Risk Assessment Project for Water Quality-Concepts and Methods.* Environmental Protection Authority, South Australia.

Blanch S, Rea N & Scott G 2005. *Aquatic conservation values of the Daly River Catchment, Northern Territory, Australia*. A report prepared by WWF-Australia, Charles Darwin University and the Environment Centre NT, Darwin.

Borsuk ME, Burkhardt-Holm P & Reichert P 2002a. A Bayesian network for investigating the decline in fish catch in Switzerland. Pages 108-113. *In* Proceedings of the IEMSS Conference, Lugano, Switzerland.

Borsuk ME, Stow CA & Reckhow KH 2002b. Integrative environmental prediction using Bayesian networks: A synthesis of models describing estuarine eutrophication. *In*, Proceedings of the IEMSS Conference, Lugano, Switzerland.

Borsuk M, Higdon D, Stow C & Reckhow K 2001. A Bayesian hierarchical model to predict benthic oxygen demand from organic matter loading in estuaries and coastal zones. *Ecological Modelling* 143,165–181.

Boulton AJ & Hancock PJ 2006. Rivers as groundwater-dependent ecosystems: a review of degrees of dependency, riverine processes and management implications. *Australian Journal of Botany* 54, 133-144.

Bowman DMJS & Wilson BA 1986. Wetland vegetation pattern on the Adelaide River flood plain, Northern Territory, Australia. *Proceedings of the Royal Society of Queensland* 97, 69–77.

Brizga SO, Arthington AH, Pusey BJ, Kennard MJ, Werren GL, Craigie NM, Choy SJ 2002. Benchmarking, a 'Top-Down' Methodology for Assessing Environmental Flows in Australian Rivers. In: Environmental Flows in River Systems. An International Working Conference on Assessment and Implementation, incorporating the 4th International Ecohydraulics Symposium. Conference Proceedings. Cape Town, March 2002. www.southernwaters.co.za/conference

Brockwell PJ & Davis RA 1996. Introduction to Time Series and Forecasting. Springer-Verlag, New York. 420 pp.

Brodie J & Mitchell A 2006. *Sediments and nutrients in north Queensland tropical streams: changes with agricultural development and pristine condition status*. CRC Reef Research Technical Report No. 62.

Brodie J & Mitchell A 2005. Nutrients in Australian tropical rivers: changes with agricultural development and implications for receiving environments. *Marine and Freshwater Research* 56, 959-968.

Brodie J, McKergow LA, Prosser IP, Furnas M, O. Hughes A & Hunter H 2003. *Sources of Sediment and Nutrient Exports to the Great Barrier Reef World Heritage Area*. Australian Centre for Tropical Freshwater Research (ACTFR) Report Number 03/11. 192 pp.

Brodie J 2002. *The effects of landuse on water quality in Australian north-east coastal catchments and coastal waterways*. Australian Centre for Tropical Freshwater Research (ACTFR) Report No. 02/07.

Bryan B & Garrod M 2006. *Combining rapid field assessment with a Bayesian network to prioritise investment in watercourse protection*. CSIRO Land and Water Science Report 10/06.

Bunn SE & Arthington AH 2002. Basic Principles and Ecological Consequences of Altered Flow Regimes for Aquatic Biodiversity. *Environmental Management* 30(4), 492-507.

Burgman MA 2005. *Risks and Decisions for Conservation and Environmental Management.* Cambridge University Press, Cambridge, UK.

Cain J 2001. *Planning improvements in natural resources management. Guidelines for using Bayesian networks to support the planning and management of development programmes in the water sector and beyond.* Natural Environment Research Council, Centre for Ecology and Hydrology (CEH) Wallingford, UK. 124 pp.

Caraco NF & Cole J 1999. Regional-scale export of C, N, P and sediment: what river data tell us about key controlling variables. In, *Integrating Hydrology, Ecosystem Dynamics, and Biogeochemistry in Complex Landscapes*. Chp 12, 239-253. Edit. JD Tenhunen & P Kabat. John Wiley & Sons, NY.

Caraco NF 1995. Influence of human populations on P transfers to aquatic systems: a regional scale study using large rivers. In *Phosphorus in the Global Environment:*

Transfers, Cycles and Management. Chp 14, 235-244, (Edit. H Tiessen). John Wiley & Sons, New York.

Carpenter SR, Caraco NF, Correll DL, Howarth RW, Sharpley AN & Smith VH 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. *Ecological Applications* 8(3), 559-568.

Carter MW 1990. *A rainfall-based Mechanism to Regulate the Release of water from Ranger Uranium Mine*. Technical Memorandum 30. Supervising Scientist for the Alligator Rivers Region. Australian Government Publishing Service, Canberra. 26 pp.

Catterall CP 1993. The importance of riparian zones to terrestrial wildlife. In SE Bunn, BJ Pusey & P Price (eds), *Ecology and Management of Riparian Zones*, Occasional paper no. 05/93. LWRRDC, Canberra, pp 41–52.

Caughley G 1981. What we do not know about the dynamics of large mammals. 361-371. In, *Dynamics of Large Mammal Populations*. CW Fowler & TD Smith (edits.). John Wiley & Sons, NY.

Chapin FS III, Sala OE, Burke IC, Grie JP, Hooper DU, Lauenroth WK, Lombard A, Mooney HA, Mosier AR, Naeem S, Pacala SW, Roy J, Steffen WL & Tilman D 1998. Ecosystem consequences of changing biodiversity. *Bioscience* 48(1), 45-52.

Chatto R 2006. *The Distribution and Status of Waterbirds Around the Coast and Coastal Wetlands of the Northern Territory*. Parks & Wildlife Commission of the NT. Technical Report 76. 237 pp.

Clark WG, Hare SR, Parma AM, Sullivan PJ & Trumble RJ 1999. Decadal changes in growth and recruitment of Pacific halibut (*Hippoglossus stenolepsis*). *Can. J. Fish. Aquat. Sci.* 56, 242-252.

Clifton C & Evans R 2001. A framework to assess the environmental water requirements of groundwater dependent ecosystems. Pages 149 – 155. *In*, Third Australian Stream Management Conference, Brisbane Queensland, 27-29 August 2001 (edits. I Rutherford, F Sheldon, G Brierly & C Kenyon). Proceedings published by the CRC for Catchment Hydrology.

Cohen JT, Lampson MA & Bowers TS 1996. The Use of two-Stage Monte Carlo Simulation Techniques to Characterize variability and Uncertainty in Risk Analysis. *Human and Ecological Risk Assessment* 2(4), 939-971.

Coleman APM 2004. *The National Recreational Fishing Survey: The Northern Territory*. Northern Territory Department of Business, Industry and Resource Development. Fisheries Report 72.

Cook GD & Setterfield SA 1996. Shrub invasion of a tropical wetland: implications for weed management. *Ecological Applications* 6(2), 531-537.

Cook GD 1996. *The program to control Mimosa pigra on Aboriginal Land in the Northern Territory by chemical and mechanical methods: An assessment*. Unpublished report to the Mimosa Steering Committee.

Cook GD 1992. *Control of Mimosa pigra at Oenpelli: Research and monitoring program*. Preliminary Report to the Steering Committee of the program to control mimosa on Aboriginal Lands in the Northern Territory. Unpublished report. Division of Wildlife and Ecology, Commonwealth Scientific and Industrial Research Organisation, Canberra.

Crowe A & Hay J 2004. *Effects of fine sediment on river biota.* Motueka Integrated Catchment Management Programme. Cawthron. Report No. 951. 35 pp.

Dambacher JM, Li HW & Rossignol PA 2003. Qualitative predictions in model ecosystems. *Ecological Modelling* 161, 79-93.

Davis R, Hamblin A, O'Loughlin E, Austin N, Banens R, Cornish P, Cox J, Hairsine P, McCulloch M, Moody P, Olley J, Prove B, Smalls I & Weaver D. 1998. *Phosphorus in the Landscape: Diffuse Sources to Surface Waters*. Land and Water Resources Research and Development Corporation, Occasional Paper 16/98.

Davis, TLO 1986. Migration patterns in barramundi, *Lates Calcarifer* (Bloch), in Van Diemen Gulf, Australia, with estimates of fishing mortality in specific areas. *Fisheries Research* 4, 243-258.

Davis TLO 1985a. Seasonal changes in gonad maturity, and abundance of larvae and early juveniles of barramundi, *Lates calcarifer* (Bloch). In Van Diemen Gulf and the Gulf of Carpentaria. *Australian Journal of Marine and Freshwater Research* 36, 177-90.

Davis TLO 1985b. The food of barramundi, *Lates calcarifer* (Bloch), in coastal and inland waters of Van Diemen Gulf and the gulf of Carpentaria, Australia. *Journal of Fish Biology* 26, 669-682.

Davis TLO & Kirkwood GP 1984. Age and growth studies on barramundi, *Lates calcarifer* (Bloch), in Northern Australia. *Australian Journal of Marine & Freshwater Research* 35(6), 673-689.

Davis TLO 1982. Maturity and sexuality in barramundi, *Lates calcarifer* (Bloch), in the Northern Territory and South-eastern Gulf of Carpentaria. *Australian Journal of Marine & Freshwater Research* 33, 529-45.

DBIRD (Department of Business, Industry & Resource Development, Northern Territory) 2005. *Fishery Status Reports 2004*. Fishery Report No. 78.

DBIRD (Department of Business, Industry & Resource Development, Northern Territory) 2004. *Fishery Status Reports 2003*. Fishery Report No. 78.

DBIRD (Department of Business, Industry & Resource Development, Northern Territory) 2003. Proposal to close Port Patterson & Bynoe Harbour to commercial fishing: A submission from the Amateur Fishermen's Association of the Northern Territory Inc.

Deere D & Davison A 2005. The Ps and Qs of Risk Assessment. *Water* March, 38-43.

Delcroix T, Cravatte S & McPhaden MJ 2007. Decadal variations and trends in tropical Pacific sea surface salinity since 1970. *Journal of Geophysical Research* 112, 1-15.

de Lestang P & Griffin RK 2000. *Impacts of Saline Intrusion on Barramundi in the Mary River Region.* Northern Territory Department of Primary Industry and Fisheries. Fishery Report No. 50.

Diamond JM & Serveiss VB 2001. Identifying Sources of Stress to Native Aquatic Fauna Using a Watershed Ecological Risk Assessment Framework. *Environmental Science & Technology* 35, 4711-4718.

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

Dilshad M, Motha JH & Peel LJ 1996. Surface runoff, soil and nutrient loss from farming systems in the Australian semi-arid tropics. *Australian Journal of Experimental Agriculture* 36, 1003-1012.

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.

Douglas MM & O'Connor RA 2004. Weed invasion changes fuel characteristics: Para grass (*Urochloa mutica* (Forssk.) T. Q. Nguyen) on a tropical floodplain. *Ecological Management & Restoration* 5(2), 143–145.

Douglas MM & O'Connor RA 2003. Effects of the exotic microphyte, para grass (*Urochloa mutica*), on benthic and epiphytic macroinvertebrates of a tropical floodplain. *Freshwater Biology* 48, 962–971.

Douglas MM, Bunn SE, Pidgeon RJW, Davies PE, Barrow P, O'Connor RA & Winning M 2001. *Weed management and the biodiversity and ecological processes of tropical wetlands*. Draft final report. National Wetlands R & D Program, Environment Australia & Land and Water Australia.

Douglas MM, Finlayson CM & Storrs MJ 1998. Weed management in tropical wetlands of the Northern Territory, Australia. In *Wetlands in a dry land: Understanding for management*. Environment Australia Biodiversity Group, Canberra, ACT 239–251.

Downing JA, McClain M, Twilly R, Melack JM, Elser J, Rabalais NN, Lewis WM Jr, Turner RE, Corredor J, Soto D, Yanez-Arancibia A, Kopaska JA & Howarth RW 1999. The impact of accelerating land-use change on the N-Cycle of tropical aquatic ecosystems: Current conditions and projected changes. *Biogeochemistry* 46, 109-148.

DPIF (Department of Primary Industry and Fisheries, Northern Territory) 1991-1997. *Reports on the chemical and mechanical Mimosa control program on Aboriginal land.* Department of Primary Industry and Fisheries Resource Protection Division, Weeds Branch, Darwin, Northern Territory.

DRCRG (Daly River Community Reference Group) 2004. *Daly Region Community Reference Group Draft Report, November 2004.*

Drewry JJ, Newham LTH, Greene RSB, Jakeman AJ & Croke BFW 2006. A review of nitrogen and phosphorus export to waterways: context for catchment modelling. *Marine and Freshwater Research* 57, 757-774.

Erskine WD, Jolly P & Smith I 2004. *Environmental Water Requirements of the Daly River: Revision of Recommendations of Erskine et al. (2003).* Based on Daly Region Water Allocation Workshop held in Darwin on 5th May 2004. Report No. 30/2004D. Northern Territory Department of Infrastructure, Planning and Environment.

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*. National River Health Program. Environmental Flows Initiative. Report 4.

Faulks JJ 1998a. *Daly River Catchment. Part 1. An Assessment of the Physical and Ecological Condition of the Daly River and its Major Tributaries.* Northern Territory Department of Lands, Planning and Environment. Technical Report TR99/10. 179 pp.

Faulks JJ 1998b. *Daly River Catchment. Part 2. Accompanying Sub-catchment Information.* Northern Territory Department of Lands, Planning and Environment. Technical Report TR99/11. 290 pp.

Ferdinands K 2006. *Assessing the relative risk of para grass invasion in the Magela Creek wetlands*. Unpublished consultancy report to the Supervising Scientist Division, Darwin, NT.

Ferdinands K, Devonport C & Whitehead P 2001. The use of GIS-based predictive models to examine the potential impact of Para grass *Urochloa mutica* (Forssk) on landscape structure in the Mary River floodplains, NT. *Proceedings of the North Australian Remote Sensing & GIS Conference*, Darwin 2001.

Ferson S 2002. RAMAS Risk Calc: Risk Assessment with Uncertain Numbers. Lewis Publishers. New York, Washington DC. 230 pp.

Fierer NG & Gabet EJ 2002. Carbon and Nitrogen Losses by Surface Runoff following Changes in Vegetation. *J. Environ. Qual.* 31,1207–1213.

Finlayson CM 1993. Vegetation change and biomass on an Australian monsoonal floodplain. In *Wetlands and Ecotones: Studies on Land-water Interactions* (Edits. B Gopal, A Hillbricht-Ilkowska & RG Wetzel), pp 157-171. International Scientific Publications, New Delhi.

Finlayson CM, Cowie ID & Bailey BJ 1990. Characteristics of a seasonally flooded freshwater system in monsoonal Australia. In 'Wetland ecology and management: case studies'. (Eds DF Whigham, RE Good & J Kvet) pp. 141–162. (Kluwer Academic Publishers: Dordrecht).

Finlayson CM, Bailey BJ, Freeland WJ & Flemming MR 1988. Wetlands of the Northern Territory. In *Conservation of Australian Wetlands*. Eds McComb AJ & Lake PS, Chipping Norton, NSW, Surrey Beatty and Sons, 103-126.

Francis RC & Hare SR 1994. Decadal-scale regime shifts in the large marine ecosystems of the North-east Pacific: a case for historical science. *Fish Oceanogr*. 3, 1-13.

Frith HJ & Davies SJJF 1961. Ecology of the Magpie Goose, *Anseranas semipalmata* Latham (Anatidae), *CSIRO Wildlife Research* 6, 91-141.

Furlonger L 2004. *Sources of Sediment in the Daly River, Northern Territory*. Unpublished Master of Science Thesis, Charles Darwin University.

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

Georges A, Webster I, Guarino F, Jolly P, Thoms M & Doody S. 2003. *Modelling Dry Season Flows and Predicting the Impact of Water Extraction on Flagship Species*. Final Report – Project 23045. Northern Territory Department of Infrastructure, Planning and Environment. 73 pp.

Gehrke PC 2005. Sustainable futures for Australia's tropical rivers: introduction to current research and information needs. *Marine and Freshwater Research* 56, v-vii.

Glaister JP 1978. The impact of river discharge on distribution and production of the school prawn *Metapenaeus macleayi* (Haswell)(Crustacea: Penaedae) in the Clarence River region, northern New South Wales. *Australian Journal of Marine and Freshwater Research* 28, 311-323.

Grayson RB, Gippel CJ, Finlayson BL & Hart BT 1997. Catchment-wide impacts on water quality: the use of 'snapshot' sampling during stable flow. *Journal of Hydrology* 199, 121-134.

Grey DL 1985. *Report of Barramundi Task Force*. Report to NT Government, Department of Ports and Fisheries. Darwin, Australia. 97 pp.

Griffin RK 2007. *Half a Century On: Barramundi Research in Australia – The Linkage between Research and Management*. Fishery Report No. 84, NT Department of Primary Industry, Fisheries and Mines.

Griffin RK 1988. A Comparison of Exploited and Unexploited Seabass *Lates calcarifer* Populations in Two Rivers in the Northern Territory, Australia. *Asian Fisheries Science* 1, 107-115.

Griffin RK 1987. Life history, distribution, and seasonal migration of barramundi in the Daly River, Northern Territory, Australia. *American Fisheries Society Symposium* 1, 358-363.

Griffin RK 1985. The Importance of Mangrove/Coastal Wetland to three commercial fisheries in the Northern Territory; Particularly for Barramundi (*Lates calcarifer*). *In* Coasts and tidal wetlands of the Australian monsoon region (eds KN Bardsley, JDS Davie & CD Woodroffe). Australian National University North Australia Research Unit. *Mangrove Monograph No. 1*a.

Griffin RK 1979. *A Survey of Amateur Angling for Barramundi (Lates calcarifer) in the Northern Territory*. Northern Territory Fisheries Division, Technical Report No. 2.

Guisan A & Zimmermann NK 2000. Predictive habitat distribution models in ecology. *Ecological Modelling* 135, 147-186.

Hamblin A 2001. *Land, Australia State of the Environment Report 2001 (Theme Report)*. CSIRO Publishing on behalf of the Department of the Environment and Heritage, Canberra.

Hamilton SK & Gehrke PC 2005. Australia's tropical river systems: current scientific understanding and critical knowledge gaps for sustainable management. *Marine and Freshwater Research* 56, 243-252.

Hamlet AF & Lettenmaier DP 1999. Columbia River stream flow forecasting based on ENSO and PDO climate signals. *Journal Water Resources Planning Management* 125(6), 333-341.

Hare SR & Mantua NJ 2001. *An historical narrative on the Pacific Decadal Oscillation, interdecadal climate variability and ecosystem impacts*. Presentation to the 20th Pacific Pink and Chum Workshop, Seattle, WA, 22 March 2001.

Hare SR & Francis RC 1994. Climate change and salmon production in the Northeast Pacific Ocean, p. 357-372. *In* RJ Beamish (edit.) Climate Change and Northern Fish Populations. *Can. Spec. Publ. Fish. Aquat. Sci.* 121.

Harris GP 2001a. Biogeochemistry of nitrogen and phosphorus in Australian catchments, rivers and estuaries: effects of land use and flow regulation and comparisons with global patterns. *Marine and Freshwater Research* 52, 139-49.

Harris G 2001b. *A Nutrient Dynamics Model for Australian Waterways: Land Use, Catchment Biogeochemistry and Water Quality in Australian Rivers, Lakes and*

Estuaries. Australian State of the Environment Second Technical Paper Series (Inland Waters), Department of Environment & Heritage, Canberra.

Harris GP 1999a. This is not the end of limnology (or of science): the world may well be a lot simpler than we think. Opinion Article. *Freshwater Biology* 42(4), 689-706.

Harris GP 1999b. Comparison of the biogeochemistry of lakes and estuaries: ecosystem processes, functional groups, hysteresis effects and interactions between macro- and microbiology. *Marine and Freshwater Research* 50, 791-811.

Harris GP 1998. Predictive models in spatially and temporally variable freshwater systems. *Australian Journal of Ecology* 23, 80-94.

Harris GP 1997. Algal biomass and biogeochemistry in catchments and aquatic ecosystems: scaling of processes, models and empirical tests. *Hydrobiologia* 349, 19-26.

Harris GP 1994a. *Nutrient Loadings and Algal Blooms in Australian waters – a Discussion Paper*. CRC for Freshwater Ecology. LWRRDC Occasional Paper No. 12/94. 99 pp.

Harris GP 1994b. Pattern, process and prediction in aquatic ecology. A limnological view of some general ecological problems. *Freshwater Biology* 32, 143-160.

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.

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

Hidalgo HG & Dracup JA 2003. ENSO and PDO effects on hydroclimatic variations of the upper Colorado River Basin. *Journal of Hydrometeorology* 4, 5-23.

Hilborn R & Mangel M 1997. *The Ecological Detective: Confronting Models with Data*. Monographs in Population Biology 28. Princeton, New Jersey: Princeton University Press.

Hill BJ & Grey D (eds) *1979. Barramundi. Summary of the Workshop convened in Darwin 4-6 September 1979*. Northern Territory Department of Primary Industry and Fisheries. Technical Report No. 1. 13 pp.

Hosking EJ 2002. *Land Clearing in the Northern Territory*. Report No. 24/2002. Northern Territory Department of Infrastructure, Planning and Environment, Darwin.

Humphrey CH, Buckle D & R Pidgeon 2006. Fish communities in channel billabongs. In Evans KG, Rovis-Hermann J, Webb A & Jones DR (eds). *eriss research summary 2004–2005*. Supervising Scientist Report 189, Supervising Scientist, Darwin NT.

ISRS 2004. *The effects of terrestrial runoff of sediments, nutrients and other pollutants on coral reefs*. Briefing Paper 3, International Society for Reef Studies, pp: 18.

Jackson S, Storrs M & Morrison J 2005. Recognition of Aboriginal rights, interests and values in river research and management: Perspectives from northern Australia. *Ecological Management and Restoration* 6(2), 105-110.

Jackson S 2004. *Preliminary Report on Aboriginal perspectives on land-use and water management in the Daly River Region, Northern Territory*. CSIRO Sustainable Ecosystems. A Report to the Northern Land Council.

Johnson AKL, Ebert SP & Murray AE 2000. Land cover change and its environmental significance in the Herbert River catchment, northeast Queensland. *Australian Geographer* 31(11), 75-86.

Johnston A & Prendergast B 1999. *Assessment of the Jabiluka Project: Report of the Supervising Scientist to the World Heritage Committee*. Supervising Scientist Report 138, Supervising Scientist, Canberra.

Johnston CA 1991. Sediment and nutrient retention by freshwater wetlands: effects on surface water quality. *Critical Reviews in Environmental Control* 21(5,6), 491-565.

Johnston CA, Detenbeck NE & Niemi GJ 1990. The cumulative effects of wetlands on stream water quality and quantity. A landscape approach. *Biogeochemistry* 10, 105-141.

Jolly P 2002a. *Daly River catchment water Balance*. Northern Territory Department of Lands, Planning and Environment. Report 10/2002D.

Jolly P 2002b. *Water balance for the Daly River catchment, Northern Territory, Australia*. Balancing the Groundwater Budget Conference, Darwin 12-17 May, 2002.

Jolly P, George D, Jolly I & Spiers Z 2000. *Analysis of groundwater fed flows for the Flora, Katherine, Douglas and Daly rivers*. Report 36/2000D, Natural Resources Division, Department of Lands, Planning and Environment, Northern Territory.

Jonsson N 1991. Influence of water flow, water temperature and light on fish migration in rivers. *Nordic J. Freshw. Res.* 66, 20-35.

Junk W, Bayley PB & Sparks RE 1989. The flood pulse concept in river-floodplain systems. *Canadian Special Publication of Fisheries and Aquatic Sciences* 106, 110-17.

Jupiter SD, Phinn S, Duke NC & Potts DC 2003. *The buffer zone: connectivity between mangrove and coral reef ecosystems in the context of land use change*. Proceedings of the 30th International Symposium for Remote Sensing of Environment, Honolulu, HI, Nov. 10-14, 2003.

Kennedy S 2004. *The Daly Report. Part One – An assessment of the terrestrial conservation values of the Daly River Catchment and the threats posed by land clearing and agricultural development*. Report prepared for the Northern Territory Environment Centre and the Wildness Society.

Kimmerer WJ 2002. Effects of freshwater flow on abundance of estuarine organisms: physical effects or trophic linkages? *Marine Ecology Progress Series* 243, 30-55.

Kinsey-Henderson AE, Post DA & Prosser IP 2005. Modelling sources of sediment at sub-catchment scale: An example from the Burdekin catchment, North Queensland, Australia. *Mathematics and Computers in Simulation* 69, 90–102.

Kinzig, AP, Ryan P, Etienne M, Allison H, Elmqvist & Walker BH 2006. Resilience and regime shifts: assessing cascading effects. *Ecology and Society* 11(1), 20.

Kirchner JW, Feng X & Neal N 2000. Fractal stream chemistry and its implications for contaminant transport in catchments. *Nature* 403, 524-527.

Knapton A 2006. *Predicting the Effects of Land Clearing on Groundwater Discharge to the Daly River*. Northern Territory Department of Natural Resources the Environment and the Arts, Technical Report No. 16/2006D.

Kundzewicz ZW 2004. Change detection in hydrological records – a review of the methodology. *Hydrological Sciences* 49(1), 7-19.

Lamon III EC & Stow GA 2004. Bayesian methods for regional-scale eutrophication models. *Water Research* 38, 2764-2774.

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, 280-310.

Latif M 1998. Dynamics of Interdecadal Variability in Coupled Ocean-Atmosphere Models. *Journal of Climate* 11, 602-624.

Levins R 1966. The strategy of model building in population biology. *American Scientist* 54, 421-31.

Lin HI, Berzins DW, Myers L, George WJ, Abdelghani A & Watanabe KH 2004. A Bayesian Approach to Parameter Estimation for a Crayfish (*Procambarus* spp.) Bioaccumulation Model. *Environmental Toxicology and Chemistry* 23 (9), 2259-2266.

Lloret J, Lleonart J, Sole I & Fromentin J-M 2001. Fluctuations of landings and environmental conditions in the north-western Mediterranean Sea. *Fisheries Oceanography* 10(1), 33-50.

Loneragan NR & Bunn SE 1999. River flows and estuarine ecosystems: implications for coastal fisheries from a review and case study of the Logan River, southeast Queensland. *Australian Journal of Ecology* 24, 431-440.

Lonsdale WM, Miller IL & Forno IW 1995. *Mimosa pigra*. In *The biology of Australian weeds*, eds RH Groves, RCH Shepherd & RG Richardson, RG & RJ Richardson, Melbourne, Vol 1, 169–188.

Lonsdale WM 1993. Rates of spread of an invading species – *Mimosa pigra* in the Northern Territory. *Journal of Ecology* 81, 513-521.

Lonsdale WM, Harley KLS & Gillett JD 1988. Seed bank dynamics of *Mimosa pigra*, an invasive tropical shrub. *Journal of Applied Ecology* 25, 963–976.

Lonsdale WM, Harley KLS & Miller IL 1985. The biology of *Mimosa pigra*. *Proceedings (2), 10th Asian Pacific Weed Science Society Conference*, Chiangmai Thailand, 484–490.

Loomis J & Cooper J 1990. Economic benefits of instream flow to fisheries: a case study of California's Feather River. *Rivers* 1(1), 23-30.

Manly BFJ & MacKenzie DI 2003. CUSUM environmental monitoring in time and space. *Environmental and Ecological Statistics* 10, 231-247.

Mantua NJ, Hare SR, Zhang Y, Wallace JM & Francis RC 1997. A Pacific interdecadal climate oscillation with impacts on salmon production. *Bulletin of the American Meteorological Society* 78, 1069-1079.

Marcot BG, Holthausen RS, Raphael MG, Rowland MM & Wisdom MJ 2001. Using Bayesian belief networks to evaluate fish and wildlife population viability under land management alternatives from an environmental impact statement. *Forest Ecology and Management* 153, 29-42.

Maynard Smith J 1973. Models in Ecology. Cambridge University Press, Cambridge. 146 pp.

McFarlane GA, King JR & Beamish RJ 2000. Have there been recent changes in climate? Ask the fish. *Progress in Oceanography* 47, 147-169.

McGilchrist CA 1975. Note on a distribution free CUSUM technique. *Technometics* 17(3), 321-325.

Miller IL & Lonsdale WM 1987. Early records of *Mimosa pigra* in the Northern Territory. *Plant Protection Quarterly* 2(3), 140–142.

Miller IL 1983. The distribution and threat of *Mimosa pigra* in Australia. *Proceedings International Symposium Mimosa pigra Management,* Chaingmai, Thailand 38–50. International Plant Protection Centre, Corvallis.

Miller IL, Nemestothy L & Pickering SE 1981. *Mimosa pigra in the Northern Territory*. Technical Bulletin No 51, Northern Territory Department of Primary Production, Darwin.

Mittag H-J & Rinne H 1993. *Statistical Methods of Quality Assurance.* Chapman & Hall. 664 pp.

Moliere D 2008. Hydrology. In *A Compendium of ecological information on Australia's Northern Tropical Rivers. Sub-project 1 of Australia's Tropical Rivers – an integrated data assessment and analysis* eds Lukacs G & Finlayson M. A report to Land & Water Australia. Australian Centre for Tropical Freshwater Research, National Centre for Tropical Wetland Research, Townsville, Qld, in press.

Moliere DR, Evans KG, Saynor MJ & Erskine WD 2004. Estimation of suspended sediment loads in a seasonal stream in the wet-dry tropics, Northern Territory, Australia. *Hydrological Processes* 18, 531–544.

Moore R 1979. Natural sex inversion in giant perch, *Lates calcarifer. Australian Journal of Marine and Freshwater Research* 30, 803-813.

Moss AJ, Rayment GE, Reily N & Best EK 1992. *A Preliminary Assessment of Sediment and Nutrient Exports from Queensland Coastal Catchments*. Queensland Department of Environment and Heritage.

Murray BR, Hose GC, Eamus D & Licari D 2006. Valuation of groundwater-dependent ecosystems: a functional methodology incorporating ecosystem services. *Australian Journal of Botany* 54, 221-229.

Neal EG, Walter MT & Coffeen C 2002. Linking the pacific decadal oscillation to seasonal stream discharge patterns in Southeast Alaska. *Journal of Hydrology* 263, 188-197.

Netica 1997. Netica^{TM} Application User's Guide. Version 1.05. Norsys Software Corp., Vancouver, BC, Canada.

Nigam S, Barlow M & Berbery EH 1999. Analysis links pacific decadal variability to drought and streamflow in the United States. *Transactions American Geophysical Union* 80, 621-625.

NLWRA (National Land and Water Resources Audit) 2002. *Australian catchment, river and estuary assessment 2002*.

Norris RH, Prosser I, Young B, Liston P, Bauer N, Davies N, Dyer F, Linke S & Thoms M 2001. *The Assessment of River Conditions (ARC): An Audit of the Ecological Condition of Australian Rivers*. Final Report submitted to the National Land and Water Resources Audit Office. CRC for Freshwater Ecology and CSIRO Land and Water.

O'Grady AP, Eamus D, Cook P, Lamontagne S, Kelley G & Hutley L 2002a. *Tree Water Use and Sources of Transpired Water in Riparian Vegetation along the Daly River, Northern Territory*. Unpublished Report. 157 pp.

O'Grady AP, Cook P, Lamontagne S & Eamus D 2002b. *Water use by riparian vegetation along the Daly River in the Northern Territory*. Balancing the Groundwater Budget Conference, May 2002, Darwin NT.

Oreskes N, Shrader-Frechette & Belitz K 1994. Verification, Validation, and Confirmation of Numerical Models in the Earth Sciences. *Science* 263, 641-646.

Owen G & Meakin C 2003. *Land Use Mapping in the Northern Territory. Report No. 02/2003D*. Northern Territory Department of Infrastructure, Planning and Environment, Darwin.

Padovan A, Townsend S & Vandenberg A 1999. *Review of Water Quality Data in HYDSYS Database for the Daly River Basin*. Northern Territory Department of Lands, Planning and Environment. Report No. 10/99D.

Pallisade 2002a. *Guide to Using Best Fit™*. Pallisade Corporation. Newfield, NY USA. 177 pp.

Pallisade 2002b. *@RISK™. Risk Analysis and Simulation Add-in for Microsoft Excel*. *Version 4.5*. Pallisade Corporation. Newfield, NY USA. 499 pp.

Pallisade 2000. *Precision Tree™*. Pallisade Corporation. Newfield, NY USA. 215 pp.

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.

Power S, Haylock M, Colman R, Wang X 2006. The Predictability of Interdecadal Changes in ENSO Activity and ENSO Teleconnections. *Journal of Climate* 19, 4755-4771.

Power S, Casey T, Folland C, Colman A & Mehta V 1999. Interdecadal modulation of the impact of ENSO on Australia. *Climate Dynamics* 15, 319-324.

Power S, Tseitkin F, Torok S, Lavery B, Dahni R & McAvancy B 1998. Australian temperature, Australian rainfall and the Southern Oscillation, 1910-1992: Coherent variability and recent changes. *Aust. Meteor. Mag*. 47, 85-101.

Prosser IP, Rutherford ID, Olley J, Young WJ, Wallbrink PJ & Moran CJ 2001a. Large-scale patterns of erosion and sediment transport in river networks, with examples from Australia. *Marine and Freshwater Research* 52, 81-99.

Prosser IP, Rustomji P, Young B, Moran C & Hughes A 2001b. *Constructing River Basin sediment Budgets for the National Land and Water Resources Audit*. CSIRO Land and Water Technical Report Number 15/01.

Puckeridge JT, Sheldon F, Walker KF & Boulton AJ 1998. Flow variability and the ecology of large rivers. *Marine and Freshwater Research* 49, 55-72.

Pusey BJ, Kennard K & Arthington AH 2004. *Freshwater Fishes of North-Eastern Australia*. CSIRO Publishing, Collingwood, Vic. 684 pp.

Pusey BJ & Arthington AH 2003. Importance of the riparian zone to the conservation and management of freshwater fish: a review. *Marine and Freshwater Research* 54, 1-16.

PWCNT (Parks & Wildlife Commission of the Northern Territory) 2003. *Management Program for the Magpie Goose (Anseranus semipalmata) in the Northern Territory of Australia*. Parks & Wildlife Commission of the Northern Territory. Unpublished Report. 30 pp.

Quiñones RA & Montes RM 2001. Relationship between freshwater input to the coastal zone and the historical landings of the benthic/demersal fish *Eleginops maclovinus* in central-south Chile. *Fisheries Oceanography* 10, 311-328.

Radziejewski C & Kundzwicz ZW 2004. Detectability of changes in hydrological records. *Hydrological Sciences Journal* 49(1), 39-51.

Rea N, Dostine P, Cook S, Wester I & Williams D 2002. *Environmental water requirements of Vallisneria nana in the Daly River Northern Territory.* Report No. 35/2002. Department of Infrastructure, Planning and Environment, Darwin, NT.

Reckhow, KH 1999. Water quality prediction and probability network models. *Can. J. Fish. Aquat. Sci.* 56(7), 1150-1158.

Rîmbu N, Boroneanț C, Buta C & Dima M 2002. Decadal variability of the Danube Rive flow in the lower basin and its relation with the North Atlantic Oscillation. *International Journal of Climatology* 22, 1169-1179.

Robins JB, Halliday IA, Staunton-Smith J, Mayer DG & Sellin MJ 2005. Freshwater-flow requirements of estuarine fisheries in tropical Australia: a review of the state of knowledge and application of a suggested approach. *Marine and Freshwater Research* 56, 343-360.

Rodrigo FS, Esteban-Parra MJ, Pozo-Vazquez D & Castro-Diez Y 2000. Rainfall variability in southern Spain on decadal to centennial time scales. *International Journal of Climatology* 20, 721-732.

Rogers C 1990. Response of coral reefs and reef organisms to sedimentation. *Marine Ecology Progress Series* 62, 185-202.

Russell DJ & Garrett RN 1985. Early Life History of Barramundi, *Lates calcarifer* (Bloch), in North-eastern Queensland. *Aust. J. Mar. Freshw. Res.* 36, 191-201.

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

Sawynok B 1998. *Fitzroy River effects of freshwater flows on fish: impact on barramundi recruitment, movement and growth*. Report for the National Fishcare Project 97/003753. Infofish Services, Rockhampton.

Scheidegger KJ & Bain MB 1995. Larval Fish Distribution and Microhabitat Use in Free-Flowing and Regulated Rivers. *Copeia* 1995(1), 125-135.

Scheuerell MD & Williams JG 2005. Forecasting climate-induced changes in the survival of Snake River spring/summer Chinook salmon (*Oncorhynchus tshawytscha*). *Fisheries Oceanography* 14(6), 448-457.

Schindler DE, Rogers DE, Scheuerell MD & Abrey CA 2005. Effects of changing climate on zooplankton and juvenile sockeye salmon growth in south-western Alaska. *Ecology* 86(1), 198-209.

Schlosser IJ 1991. Stream fish ecology: a landscape perspective. *Bioscience* 41(10), 704-712.

Schueler TR 1997. Impact of Suspended and Deposited Sediment. *Technical Note #86 from Watershed Protection Techniques* 2(3), 443-444.

Schult J & Metcalfe R 2006. *High nitrate concentrations in the Douglas River and their impact on Daly River water quality*. Northern Territory Department of Natural Resources the Environment and the Arts, Report 9/2006D.

Serveiss VB 2002. Applying Ecological Risk Principles to Watershed Assessment and Management. *Environmental Management* 29 (2), 145–154.

Silverman BW 1990. *Density Estimation for Statistics and Data Analysis*. Chapman and Hall, London, New York. 175 pp. First edition 1986.

Standards Australia/Standards New Zealand/CRC for Australian Weed Management 2006. Handbook. *National Post-border Weed Risk Management Protocol*. 77 pp.

StatSoft 2001. *Statistica™ System Reference*. StatSoft Inc. Tulsa OK, USA. 1098 pp.

Staunton-Smith J, Robins JB, Mayer DG, Sellin MJ & Halliday IA 2004. Does the quantity and timing of fresh water flowing into a dry tropical estuary affect year-class strength of barramundi (*Lates calcarifer*)? *Marine and Freshwater Research* 55, 787-797.

Story R, Galloway RW, McAlpine JR, Aldrick JM, & Williams MAJ 1976. *Lands of the Alligator Rivers Area, Northern Territory*. Land Research Series No. 38, CSIRO, Melbourne, Australia.

Sutcliffe Jr WH 1973. Correlations between Seasonal River Discharge and Local Landings of American Lobster (*Homarus americanus*) and Atlantic Halibut (*Hippoglossus hippoglossus*) in the Gulf of St. Lawrence. *Journal of Fisheries Research Board of Canada* 30(6), 856-859.

Syvitski JPM, Peckham SD, Hilberman R & Mulder T 2003. Predicting the terrestrial flux of sediment to the global ocean: a planetary perspective. *Sedimentary Geology* 162, 5 –24.

Taylor AH, Allen JI & Clark PA 2002. Extraction of a weak climatic signal by an ecosystem. *Nature* 416, 629-632.

Taylor JA & Tullock D 1985. rainfall in the wet-dry tropics: Extreme events at Darwin and similarities between years during the period 1870-1983. *Australian Journal of Ecology* 10, 281-295.

Thomas C, Hart B, Nicholson A, Grace M, Brodie J & Pollino C 2005. Development Of Criteria For Simplifying Ecological Risk Models. In Zerger, A. and Argent, R.M. (eds) *MODSIM 2005 International Congress on Modelling and Simulation*. Modelling and Simulation Society of Australia and New Zealand, December 2005, pp. 772-778. ISBN: 0-9758400-2-9. http://www.mssanz.org.au/modsim05/papers/thomas.pdf

Tickell SJ, Cruickshank S, Kerle E & Willis G. 2002. *Stream Baseflows in the Daly Basin.* Department of Infrastructure, Planning and Environment, Technical Report No. 2002/36. Northern Territory Government. Darwin, NT.

Tickell SJ 2004. *Carbon dating of groundwaters in the Oolloo Dolostone Aquifer*. Northern Territory Department of Infrastructure, Planning and Environment. Report 21/2004D.

Tickell SJ 2002. *A Survey of Springs along the Daly River*. Northern Territory Department of Infrastructure, Planning and Environment. Report 06/2002.

Townsend SA & Gell PA 2005. The role of substrate type on benthic diatom assemblages in the Daly and Roper Rivers of the Australian wet/dry tropics. *Hydrobiologia* 548, 1–115.

Townsend SA & Padovan A 2005. The seasonal accrual and loss of benthic algae (*Spirogyra*) in the Daly River, an oligotrophic river in tropical Australia. *Marine and Freshwater Research* 56, 317-327.

Townsend S, Gell P, Bickford S, Tibby J, Croome R, Przybylska M, Padovan A & Metcalfe R 2002. *Periphyton and Phytoplankton Response to Reduced Dry Season Flows in the Daly River*. Final Report for Project ID 22963. Northern Territory Department of Infrastructure, Planning and Environment. Darwin, NT.

Townsend SA & Douglas MM 2000. The effect of three fire regimes on stream water quality, water yield and export coefficients in a tropical savanna (northern Australia). *Journal of Hydrology* 229, 118-137.

Travnichek VH, Bain MB & Maceina MJ 1995. Recovery of a warm water fish assemblage after the initiation of a minimum-flow release downstream from a hydroelectric dam. *Transactions of the American Fisheries Society* 124, 836-844.

Tullock DG & McKean JL 1983. Magpie Goose Population on the Coastal Plains of the Northern Territory (1958 – 1980). *Corella* 7(2), 32-36.

Turner TF, Trexler JC, Miller GL & Toyer KE 1994. Temporal and Spatial Dynamics of Larval and Juvenile Fish Abundance in a Temperate Floodplain River. *Copeia* 1994(1), 174-183.

US EPA (U.S. Environmental Protection Agency) 1998. *Guidelines for Ecological Risk Assessment.* EPA/630/R-95/002F. Risk Assessment Forum, Washington, DC, USA.

US EPA (U.S. Environmental Protection Agency) 2003. *Framework for Cumulative Risk Assessment.* EPA/630/P-02/001F. Risk Assessment Forum, Washington, DC, USA.

Viles HA & Goudie AS 2003. Interanual, decadal and multidecadal scale climatic variability and geomorphology. *Earth-Science Reviews* 61, 105-131.

Walden D, van Dam R, Finlayson M, Storrs M, Lowry J & Kriticos D 2004. *A risk assessment of the tropical wetland weed Mimosa pigra in northern Australia*. Supervising Scientist Report 177, Supervising Scientist, Darwin NT.

Walden D & Bayliss P 2003. *An ecological risk assessment of the major weeds on the Magela Creek floodplain, Kakadu National Park*. Internal Report 439, Supervising Scientist Division, Darwin.

Walden D, Bayliss P, Boyden J & Ferdinands K In prep. *An ecological risk assessment of the major weeds on the Magela Creek floodplain, Kakadu National Park*. Supervising Scientist Report, Darwin.

Walker KF & Thoms MC 1993. Environmental effects of flow regulation on the Lower River Murray, Australia. *Regulated Rivers Research & Management* 8, 103-119.

Walters CJ 1986. *Adaptive Management of Renewable Resources*. Macmillan Publishing Company, New York.

Walters C, Buckworth R, Gribble N, Griffin R, Johnson G, Loneragan N & Ram D 1997. Status and Management Options for the Barramundi Fishery. In *Towards the Sustainable Use of Northern Territory Fishery Resources: Review Workshops led by Carl J. Walters* (edit. DC Ramm). Northern Territory Department of Primary Industry and Fisheries. Fisheries Report 39. 33 pp.

Wassenaar LI, Hendry MJ & Harrington N 2006. Decadal geochemical and isotopic trends in nitrate in a transboundary aquifer and implications for agricultural beneficial management practices. *Environmental Science and Technology* 40, 4626-4632.

Wasson RJ, Olive LJ & Rosewell CJ 1996. Rates of erosion and sediment transport in Australia. In, Erosion and Sediment Yield: Global and Regional Perspectives. Proceedings of the Exeter Symposium, July 1996. *International Association of Hydrological Sciences Publ.*, 236, 139-48.

Wasson RJ 1994. Annual and decadal variation of sediment yield in Australia, and some global comparisons. *International Association of Hydrological Sciences Publ.* 224, 269-279.

Webster IT, Rea N, Padovan AV, Dostine P, Townsend SA & Cook S 2005. An analysis of primary production in the Daly River, a relatively unimpacted tropical river in northern Australia. *Marine and Freshwater Research* 56, 303-316.

White GF 1998. *The Northern Territory Barramundi Classic Fishing Tournament 1982 to 1997*. Fisheries Report No. 44. Fisheries Division, Northern Territory Department of Primary Industry and Fisheries.

Whitehead PJ & Saalfeld K 2000. Nesting phenology of magpie geese (*Anseranus semipalmata*) in monsoonal northern Australia: response to antecedent rainfall. *J. Zool. London* 251, 495-508.

Whitehead PJ, Wilson BA & Bowman D 1990. Conservation of coastal wetlands of the Northern Territory of Australia: the Mary River floodplain. *Biological Conservation* 52, 85-111.

Wilkinson L 1979. Tests of significance in stepwise regression. *Psychological Bulletin* 86, 168-174.

Wilson D, Cook P, Hutley L, Tickell S & Jolly P 2006a. *Water Quality in the Daly River – A multi-disciplinary management approach*. Northern Territory Department of Natural Resources the Environment and the Arts, Technical Report No. 18/2006D.

Wilson D, Cook P, Hutley L, Tickell S & Jolly P 2006b. *Effect of land use on estimates of evapotanspiration and recharge in the Daly River catchment.* Northern Territory Department of Natural Resources the Environment and the Arts, Technical Report No. 17/2006D.

Wilson BA, Brockelhurst PS & Whitehead PJ 1991. *Classification, Distribution and Environmental Relationships of Coastal Floodplain Vegetation, Northern Territory, Australia*. Conservation Commission of the Northern Territory. Technical Memorandum 91/2.

Wu F-C & Tsang Y-P 2004. Second-order Monte Carlo uncertainty/variability analysis using correlated model parameters: application to salmonid embryo survival risk assessment. *Ecological Modelling* 177, 392-414.

Wygralak AS 2006. *Daly River Catchment –towards an integrated catchment management.* AESC2006, Melbourne, Australia. School of Resources, Environment & Society, Australian National University.

Young WJ, Marston FM & Davis JR 1996. Nutrient exports and land use in Australian catchments. *Journal of Environmental Management* 47, 165-183.

Young P 2003. Top-down and data-based mechanistic modelling of rainfall-flow dynamics at the catchment scale. *Hydrological Processes* 17(11), 2195-2217.

Young P 1998. Data-based mechanistic modelling of environmental, ecological, economic and engineering systems. *Environmental Modelling and Software* 13, 105-122.

Zar 1984. *Biostatistical Analysis*. Second Edition. Prentice Hall, Inc., Englewood Cliffs, N.J. 718 pp.

Zhang Y, Wallace JM & Battitti DS 1997. ENSO-like interdecadal variability, 1900-93. *Climate* 10, 1004-1020.

4.7 Appendices

Excel program & data files, @Risk files & Netica files (provided on separate DVD)

Daly River flow (ML) and stage height (m) data. (combination of G8140040 & G8140041 for Mt Nancar).

Floodplain Health

a. Magpie geese aerial survey data for the Daly River floodplain (1983-2000)

b. Daly River floodplain complete spatial data set A

c. Daly River floodplain complete spatial data set B

d. Magpie geese nesting model

e. @Risk Magpie geese nesting model

f. Netica Bayesian Network – Daly River floodplain health

g. Netica Bayesian Network – Daly River floodplain health with weed control

In-stream Health

a. Raw NT Fisheries catch-effort data Daly River (1983-2005)

b. Processed NT Fisheries catch-effort data Daly River

c. Barramundi catch model

d. @Risk barramundi catch model

e. Barramundi population (CPUE) model

f. @Risk barramundi population (CPUE) model

g. Barramundi catch model using Kernels

h. @Risk barramundi catch model using Kernels

i. Netica Bayesian Network – Daly River in-stream health using barramundi catch

j. Netica Bayesian Network – Daly River in-stream health using barramundi CPUE

Catchment Health

a. Summary available Daly River Water Quality data

b. Sub-catchment sediment & nutrient export land use model

c. @Risk sub-catchment sediment & nutrient export land use model

d. Netica Bayesian Network – catchment health using a 20% pristine threshold effect

e. Netica Bayesian Network – catchment health using a 50% pristine threshold effect

Native Floodplain Vegetation Communities in the NT

Comparison of two basic vegetation communities found on NT wetlands. The Daly-Reynolds floodplain-river systems are highlighted in bold.

River System	Paperbark Forest Km^2	Open floodplain (grassland, sedge land) Km^2	Percent of all floodplains by River system
Moyle River	- (>0%)	718 (<100%)	6.9
West Daly	**- (>0%)**	**265 (<100%)**	**2.5**
Daly/Reynolds	**84(6%)**	**1382 (94%)**	**14**
Finniss	140(20.5%)	543(79.5%)	6.5
Adelaide	- (>0%)	1017(<100%)	9.7
Mary/Swim Ck	230(23.4%)	755(76.6%)	9.4
Wildman/West Alligator	154(21%)	576(79%)	7.0
South Alligator	161(16.4%)	821(83.6%)	9.4
East Alligator	-	803(<100%)	7.7
Cooper Ck	- (>0%)	190(<100%)	1.8
Murganella Ck	- (>0%)	536(<100%)	5.1
Liverpool/Tomkinson	60(42.6%)	81(57.4%)	1.3
Blyth/Cadell	- (>0%)	21(<100%)	<1.0
Milingimbi	76(49%)	78(51%)	1.5
Arafura Swamp	559(88%)	75(12%)	6.1
Arnhem Bay	38(24%)	115(76%)	1.5
Caledon Bay	- (>0%)	266(<100%)	2.5
Other Areas	50	145	1.9

* Figures based on the 1:1 000 000 vegetation map of the NT (Wilson et al 1991).