Australian Government
National Water Commission

Australian Water Resources 2005

A baseline assessment of water resources for the National Water Initiative

Level 2 Assessment

River and Wetland Health Theme

Assessment of River and Wetland Health: Potential Comparative Indices

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WRON

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Raising National Water Standards

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Authors: Richard H. Norris, Fiona Dyer, Peter Hairsine, Mark Kennard, Simon Linke, Linda Merrin, Arthur Read, Wayne Robinson, Chris Ryan, Scott Wilkinson, and David Williams.

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Also available at: <http://www.water.gov.au>

National Water Commission

95 Northbourne Ave
Canberra ACT 2600
Email: awr@water.gov.au
Phone: 02 6102 6000

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Executive summary

The National Framework for the Assessment of River and Wetland Health (FARWH) is being developed as part of the Australian Water Resources 2005 project being undertaken by the National Water Commission (the Commission) under the National Water Initiative (NWI).

The Australian Water Resources 2005 (AWR 2005) is reporting under three headline parameters: Water Availability, Water Use, and River and Wetland Health.

The AWR 2005 Discovery Phase, undertaken in early 2006, examined the availability of data to undertake a national river health assessment based on the last national assessment under the Australian Catchment, River and Estuary Assessment 2002. It was determined that while in some areas of Australia there were significant gaps in available data, other areas had methodologies and techniques that had advanced beyond those of 2002.

For this reason the Commission decided to progress a national framework for river and wetland health assessment, to be developed with extensive consultation with partner governments, to allow for the future application of a robust national assessment, which utilises existing work to the maximum extent possible.

The resulting framework, FARWH, is designed to provide the information needed to:

- establish 'environmental and other public benefit outcomes' (NWI paragraph 35)
- 'address currently over allocated and/or overused systems' (NWI paragraphs 41–45)
- support 'integrated management of environmental water' (NWI paragraphs 78–79).

Having an agreed framework that is acceptable to all states and territories is an essential requirement. The aim of the framework is to provide national assessments, from comparable state and territory assessments, of the aggregate impacts of resource use on rivers and wetlands as the basis for reporting on waterway condition. The increasing role of regional authorities in monitoring natural resources means it is also important to engage more broadly so as to maximise the ability to align, wherever possible, all relevant monitoring regimes with the framework.

The FARWH has been developed to guide the national assessment of river and wetland health to determine if there is long-term change in condition, including change resulting from water management regimes. It is closely linked to other major programs such as:

the Victorian Index of Stream Condition (ISC) [http://www.vicwaterdata.net/vicwaterdata/data_warehouse_content.aspx?option=5];

the Victorian Index of Wetland Condition [http://www.dse.vic.gov.au/DSE/nrence.nsf/LinkView/3EA5B6AEFB53EE3DCA25708B00145F44522C816829EBF3F7CA25700C00240E63];

the Tasmanian Conservation of Freshwater Ecosystem Values Framework (CFEVF) [http://www.dpiw.tas.gov.au/inter.nsf/WebPages/JMUY-5QF35H?open];

the Queensland Wetlands Program [http://www.epa.qld.gov.au/publications/p01948aa.pdf/Monitoring_wetland_extent_and_condition.pdf]; and

the Natural Resource Management Ministerial Council and the National Natural Resource Management Monitoring and Evaluation Framework (NNRMM&EF) [http://www.nrm.gov.au/monitoring/].

The FARWH is based on a hierarchical model of river and wetland function, which addresses: environmental components to be represented by a national assessment, reporting scale, reference condition, discussion on selection of indices, methods for integrating and aggregating indices for assessment, sensitivity analysis, range standardisation, and managing missing data. This document on potential comparative indices is additional to the FARWH and presents examples of indices used during the 2002 National Land and Water Resources Audit, the Victorian Index of River Condition (ISC), the Tasmanian Conservation of Freshwater Ecosystem Values program (CFEV), and the Murray–Darling Basin Sustainable Rivers Audit (SRA).

The FARWH proposes that six key components are appropriate for the assessment of river and wetland health, all of which are considered to represent ecological integrity. These are:

- catchment disturbance
- hydrological change
- water quality
- physical form
- fringing zone, and
- aquatic biota.

The FARWH describes how to develop and combine indices so that nationally comparable assessments of river and wetland health can be achieved. This is designed to enable states and territories to include data that are already being collected (e.g. AUSRIVAS) and to compare these data between regions. In some cases, such as the Victorian Index of Stream Condition, little change may be needed to report the data in the framework.

The potential comparative indices included here describe indices that have been developed for each of the environmental components noted above according to the framework, largely during the first National Land and Water Resources Audit. Indices developed as part of the Victorian Index of Stream Condition and the Tasmanian Conservation of Freshwater Ecosystem Values also conform to this framework. Any of these may be selected by various jurisdictions providing they meet their needs. They may also be selected to fill gaps in the environmental components that may not be covered in existing state and territory programs. Although these indices are not being prescribed, some of them, such as the AUSRIVAS and hydrology indices (Flow Stress Ranking, FSR), have already been accepted for use by several jurisdictions and may be used quite widely.

The potential comparative indices described here therefore provide a series of techniques that can be used in the assessment of river and wetland health, including a new method that has been developed during the Australian Water Resources 2005 for the fringing (riparian) zone.

1 Introduction: Selection of indices

The following section outlines a list of indices that meet the criteria agreed by Australian states and territories in the framework presented here. These are offered as 'off the shelf' methods that can be used where no assessment system, or an incomplete system, is currently in place. The indices included in this section stem mainly from the Assessment of River Condition (ARC) program; however, there are many other programs that may be drawn upon for selection of appropriate indices. It is important to note that the framework does not require or even suggest that the adoption of these indicators is the most appropriate solution for any region of Australia. These indices are offered merely as examples and as a starting point for investigation of potential indicators that meet framework requirements.

Other appropriate indices may be developed in-house (in accordance with the design parameters outlined in the framework) or adopted from other wide-scale assessment programs such as the Victorian Index of Stream Condition (ISC), Tasmania Conservation of Freshwater Ecological Values (CFEV) program, Sustainable Rivers Audit (SRA), or other programs currently in place or in development. Examples of programs under development include the National Land and Water Resources Audit (NRLWRA), National Indicators program, and the Queensland Stream & Estuarine Assessment Program (SEAP) and others. Wetland indicator programs are also in-place or under development in several jurisdictions, including the Index of Wetland Conditions (IWC) in Victoria and Queensland.

The development of indicators for river and wetland health assessment is a very fluid research area and it is suggested that the most recent research is reviewed before deciding on a set of indices rather than relying completely on this document.

The following sections present an updated version of the ARC program indices as an example of indicators that are compliant with the framework. As outlined previously, during the development of their programs jurisdictions may draw on any part of this document, or other programs, or develop new indicators.

2 Biota Index

2.1 Assessment of river condition using biota (ARC_B)

In 1992 the Australian Federal Government, through Environment Australia and the Land and Water Resources Research and Development Corporation, initiated a nationwide program of biological assessment of river health called the Monitoring River Health Initiative (MRHI), formed under the National River Health Program (NRHP) (Sloane et al. 1999). Under the MRHI, the Australian River Assessment System (AUSRIVAS) predictive models for the biological assessment of river health were developed. The First National Assessment of River Health (FNARH) was an extension of the NRHP and involved sampling approximately 6000 sites across Australia. The AUSRIVAS models [http://ausrivas.canberra.edu.au] were used to assess these sites, providing a measure of the biological health of rivers throughout Australia.

2.2 Components of the ARC_B

Only aquatic macroinvertebrates were used for the ARC_B because they represented the only database with sufficient extent and consistency to meet the needs of the reporting scale of the National Land and Water Resources Audit. Ideally, other components of the biota such as fish, water plants, algae, riparian vegetation, and water birds might also be used in the future when appropriate data become available.

2.3 Sources of data

Macroinvertebrate data were collected by state agencies under the Federal National River Health Program (NRHP). The NRHP and the First National Assessment of River Health (FNARH) have been managed by the Land and Water Resources Research and Development Corporation (now Land and Water Australia, LAWA) and Environment Australia (EA). It has been a collaborative effort in data collection and model-building between the states and the Australian Government in conjunction with LAWA, the Cooperative Research Centre for Freshwater Ecology, and many other scientists involved in collection of data, technical advice, and management. Dr Peter Davies has played a key role in directing the program and bringing it to its current status.

The data used to compile this part of the assessment were supplied to Environment Australia by the lead agency in each state and territory. These were:

- Queensland Department of Natural Resources and Mining
- NSW Environment Protection Authority
- Environment ACT
- Victorian Environment Protection Agency

- Land and Water Resource Assessment Branch, Tasmanian Department of Primary Industries, Water and Environment
- South Australian Environment Protection Authority
- Western Australia Department of Conservation and Land Management
- Northern Territory Department of Lands, Planning and Environment, Natural Resources Division.

Data were collected and analysed using the standardised AUSRIVAS methods (Coysh et al*.* 2000, http://ausrivas.canberra.edu.au, Simpson and Norris 2000). Approximately 6000 sites were sampled throughout Australia; at most sites two habitats were assessed with many re-assessed four to six times since 1996.

2.4 AUSRIVAS modelling

2.4.1 A brief overview of AUSRIVAS

AUSRIVAS models assess biological condition by comparing the types of aquatic invertebrates observed at sites of unknown or suspect biological condition (test sites) with the biota predicted to occur in the absence of disturbance. This ratio of the number of observed types of animals to those expected to occur in the absence of disturbance caused by humans (O/E ratio), is an intuitive and ecologically meaningful measure of river health. Simpson and Norris (2000) have described the methods in detail.

In brief, predictive models have been built from biological and environmental data collected from reference sites minimally affected by humans. Biotic composition (the different kinds of animals) at these sites is related to environmental features unlikely to be influenced by human activity (e.g. latitude, elevation, channel slope, alkalinity, etc.). The same suite of environmental variables are measured at test sites and then used to match them with reference sites and predict the kinds of animals that should occur at these sites in the absence of damage by humans. The output from AUSRIVAS models – the ratio of the number of taxa observed at a test site to the total number of taxa expected from the model (O/E ratio) – is used as a measure of biological impairment. O/E ratios substantially less than 1.0 imply test sites have lost many kinds of animals expected to live there and consequently are adversely affected by some environmental stressor. Forty-eight such models have been created – six for each Australian state and territory based on two habitats (usually edge and main channel) for spring and autumn and spring/autumn combined. The models are available at the AUSRIVAS website http://ausrivas.canberra.edu.au (Coysh et al. 2000).

The reference condition

Reference sites provide the basis for the models and the comparisons that finally determine the condition of test sites. Reference sites, chosen to be minimally disturbed, were selected by each state or territory agency based on clear criteria such as intact riparian zones, no stock access, several kilometres from a dam or weir, no road crossing within the vicinity, and so on. Specific criteria are available from the AUSRIVAS website.

It is recognised that many areas, especially lowland rivers, have limited sites that can be considered to be in good condition. Reference site selection in these areas has been assisted by the knowledge that aquatic invertebrates are strongly controlled by local conditions and so sites could be selected from local sections of rivers that met the selection criteria. Also, sites could be selected from tributaries that had less change than the main stem.

2.4.2 AUSRIVAS modelling process

The first step in creating an AUSRIVAS model is to classify the reference sites into groups, based on the faunal composition using UPGMA (Unweighted Pair-Group arithMetic Averaging) as the classification algorithm (Simpson and Norris 2000). A Stepwise Multiple Discriminant Function Analysis (MDFA) is carried out to determine which environmental variables discriminate best between the faunal groups (Simpson and Norris 2000). To predict the expected community from a certain combination of environmental variables at a test site, the discriminant functions are used to determine the standardised, multivariate distance of the site from the groups (Figure 1).

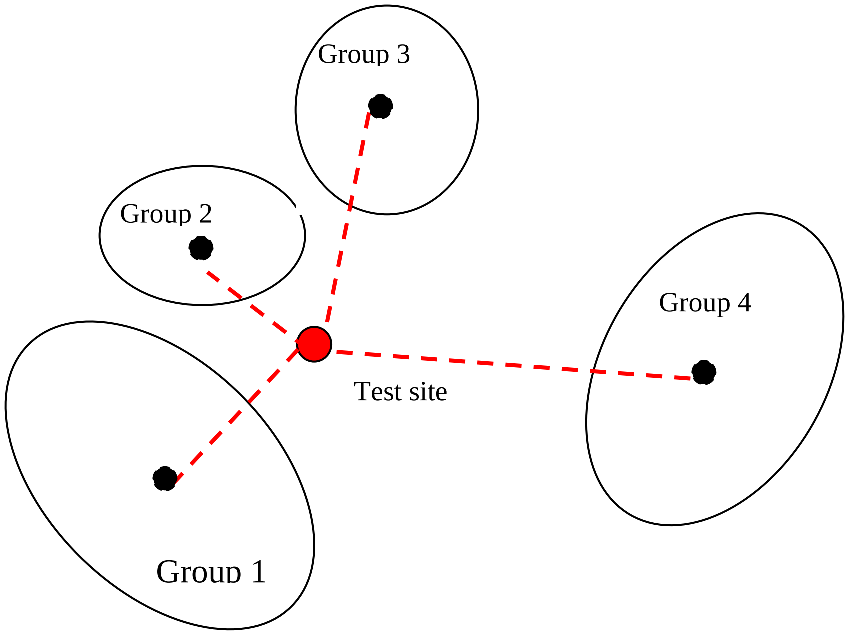

Figure 1 Graphical representation of the probability weighting applied in AUSRIVAS.

Based on the distance of a test site from a group, a weighted average of the probability of the taxon occurring at the test site is calculated (Table 1) as described in Clarke et al. (1996) and Moss et al. (1987).

Table 1 Calculation of the probability of a taxon occurring at a site. The final probability is the sum of the contributions of each group, calculated by the probability of the site belonging to that group and the frequency of the taxon being found in that group (adapted from Wright 1995).

Classification Group	Probability That Test Site Y Belongs to Group	Frequency of Taxon X in Group (percent)	Contribution to Probability That Taxon X will Occur at Site Y (percent)
1	0.1	60	6
2	0.6	50	30
3	0.2	60	12
4	0.1	90	9
			Σ =Total Probability = 57 percent

2.4.3 Taxa probability levels

Unlike the British RIVPACS (Moss et al. 1987), only taxa that have probability of occurrence of ≥50 percent are considered. The level was chosen to exclude taxa with a low chance of occurrence from the prediction, so that sampling variability will have a low impact on the sensitivity of the model, while including enough taxa to be able to measure effects on the biological community. Simpson and Norris (2000) showed that the 50 percent cut-off seems appropriate for achieving both robustness and sensitivity (Figure 2), where taxa with a probability >50 percent provide most of the information.

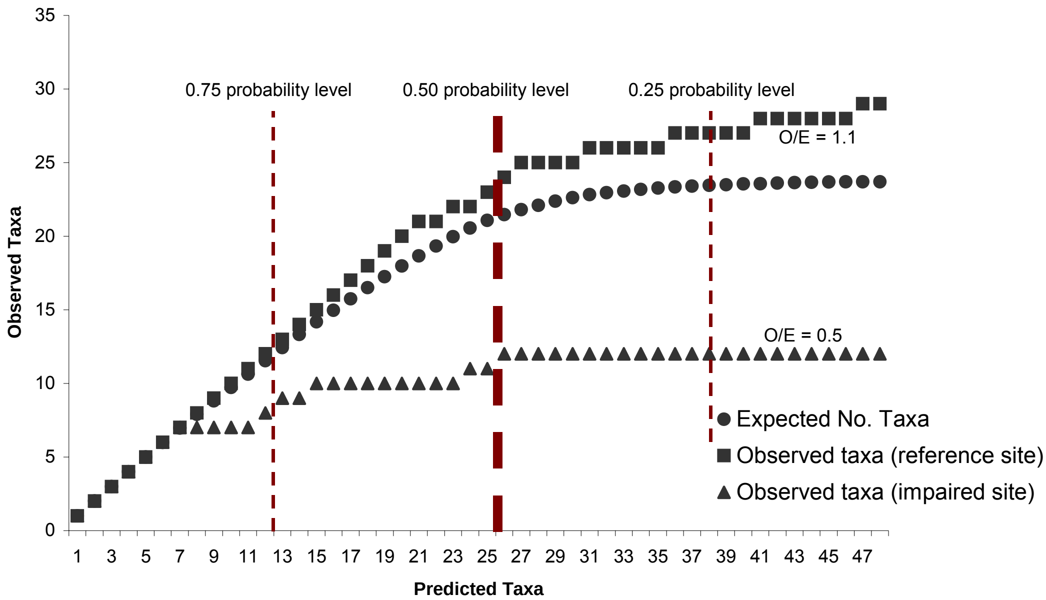

Figure 2 Taxa accretion curves for a predicted reference site (circles), an actual reference site (squares), and an impaired site (triangles). Taxa below the 0.5 probability of occurrence cut-off contribute little useful information for making a biological assessment of a site (from Simpson and Norris 2000).

2.4.4 AUSRIVAS banding scheme

The observed number (*O*) of taxa is the number of taxa found at a site with >50 percent chance of occurrence, while the expected number of taxa (*E*) is the sum of the probabilities of those taxa predicted to occur at the test site. When all the expected taxa occur, the ratio of observed to expected (*O/E*) will be close to one. In case of an unnatural change in the community, the number of observed taxa will drop and the *O/E* will decrease. The acceptable range of *O/E* scores in AUSRIVAS has been defined as the range between the 10th and the 90th percentile of the reference sites (Simpson and Norris 2000). An *O/E* below the 10th percentile indicates an unnatural loss of taxa; an *O/E* higher than the 90th percentile is judged to be richer than expected and the site is reviewed. To summarise the AUSRIVAS outputs, a banding scheme was been developed by the NRHP (Table 2).

The bandwidth is determined by the width of band A, i.e. the 10th and 90th percentiles. Band B starts at the 10th percentile (typically about *O/E* = 0.85) and has the same bandwidth as band A. Band C will have the same bandwidth again, whereas the width of band D will be determined by the difference of its start and an *O/E* of 0. Sites richer than reference will be assigned band X, which usually characterises mild organic enrichment and should be reviewed.

Table 2 AUSRIVAS/ANNA banding schemes.

Band label	Band name	Comments
X	Richer than reference	More taxa found than expected Potential biodiversity "hot spot" Mild organic enrichment Continuous irrigation flow in a normally intermittent stream
A	Reference	Index value within range of central 80 percent of reference sites
B	Below reference	Fewer taxa than expected Potential impact either on water quality or habitat quality or both resulting in a loss of taxa
C	Well below reference	Many fewer taxa than expected Loss of taxa because of substantial impacts on water and/or habitat quality
D	Impoverished	Few of the expected taxa remain Severe impairment

2.5 ANNA modelling

2.5.1 Function of the ANNA models

The function of the ANNA models (Linke et al. 2005) was to enable prediction of the biotic condition of a reach that had not been sampled. The AUSRIVAS modelling approach uses two measures for each site: a list of taxa predicted to occur if the site is in good condition (*E* value), and the taxa observed to occur at the site (*O* value). Together these two values give the *O*/*E* score.

For reaches that had not been sampled, there was a need to predict both the taxa expected at the site if it were in good condition (*E* value), and to also predict taxa that occurred at the site under current conditions (modelled observed value *MO*). Together these values provide the *MO*/*E* score, a measure of condition and a surrogate for the AUSRIVAS *O*/*E* score. The ANNA modelling approach was judged more suitable for the needs of the NLWRA than the AUSRIVAS modelling approach because, while it produces similar outputs, it avoids the classification step and is computationally more efficient. An overview of the ANNA modelling procedure is shown below (Figure 3).

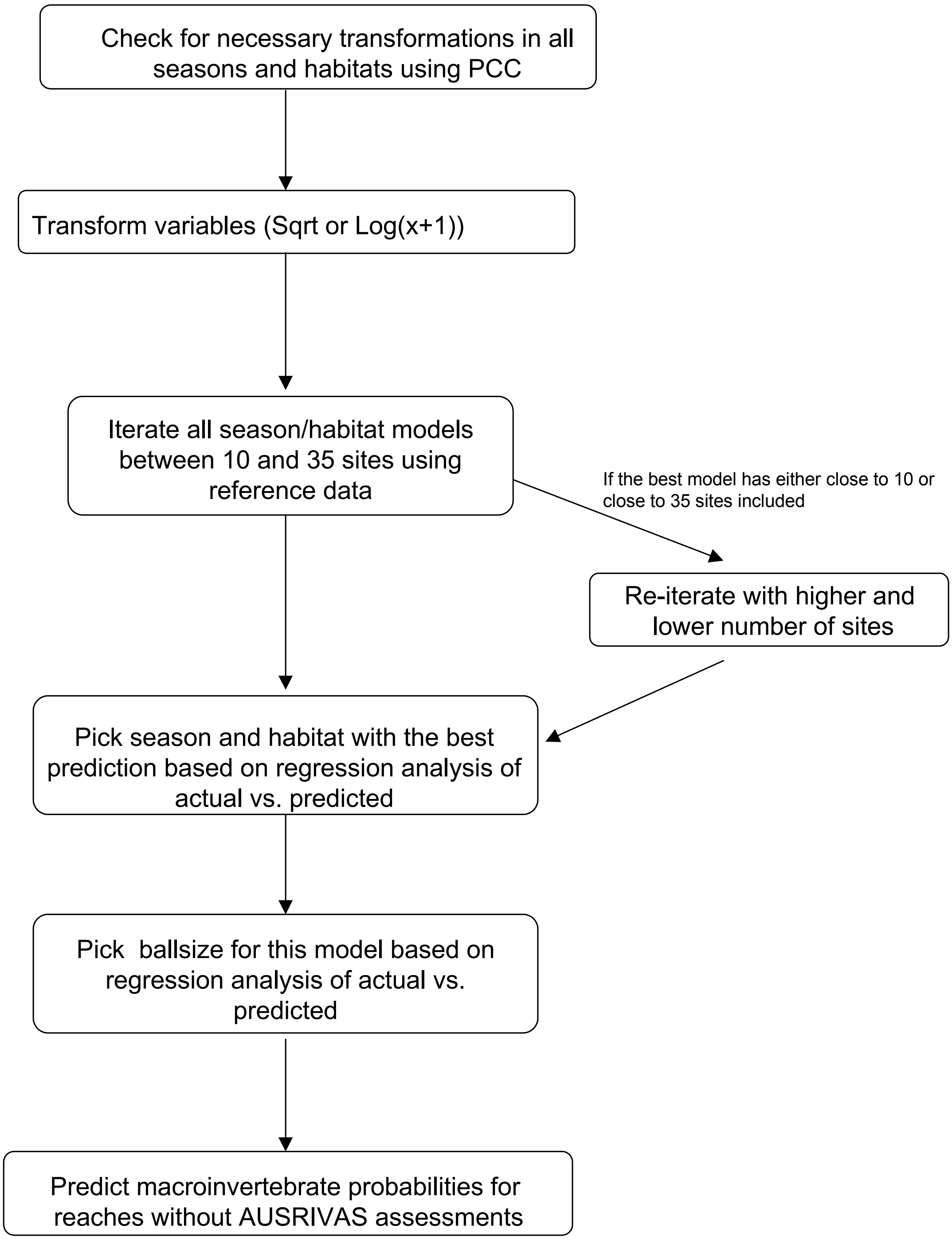

Figure 3 Processes used for building ANNA models (see Linke et al. 2005).

2.5.2 ANNA modelling approach

ANNA, an approach originally developed by Stockwell and Faith (1996), is similar to AUSRIVAS/RIVPACS models in that it can be used to calculate an observed/expected ratio of taxa richness. The fundamental difference of ANNA is that the classification and discriminant function analysis steps in AUSRIVAS/RIVPACS are not needed. Instead of going through a series of multivariate steps, the ANNA procedure finds the most similar sites based on a number of environmental variables by calculating the Euclidean distance between the test site and the set of reference sites. Euclidean distance, also called ecological distance, is the simplest and oldest similarity coefficient (Washington 1984) and has been widely used in ecological studies (Faith et al*.* 1987, Hruby 1987) to describe differences between habitats. The most similar sites are then used to establish the reference condition. This is similar to the Canadian bioassessment approach BEAST (Reynoldson et al*.* 1995), although in ANNA the reference sites are grouped specifically for each test site, instead of allocating the test site to static groups as done in the BEAST method.

To predict the number of taxa for a site, the site is placed in the ordination space of the reference sites (Figure 4). The probability of occurrence for each taxon at a site is then calculated similar to AUSRIVAS (Table 1). An important difference is that probabilities are calculated from each reference site in the ball, not from group centroids as with AUSRIVAS.

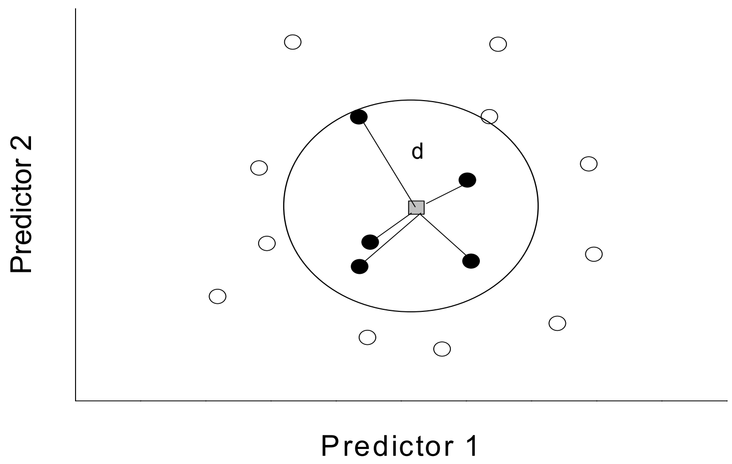

Figure 4 Conceptual diagram of ANNA. The closest reference sites to a test site are considered in the prediction. The contribution of each reference site is weighted by the square root of the Euclidean distance *d* (Linke et al. 2005).

The square-root of the Euclidean distance of reference sites from the test site was used to weight the contribution of each reference site to the community prediction at the test site. This was done so those reference sites that were very close to the test site did not overwhelm the prediction.

2.5.3 Assessing model quality

To evaluate the quality of the ANNA and AUSRIVAS model predictions, the reference sites were run through the models. When validating AUSRIVAS models, reference site data had to be run through the model that originally included them, potentially adding a slight positive bias to the validation. In ANNA, the reference site about to be tested was removed from the model and its community was predicted from the remaining reference sites. A ‘good model’ was defined as one that could predict the taxa community at reference sites as closely as possible.

Plotting the observed against the expected taxa ratios for reference sites assesses the quality of the models. Ideally, a regression line should pass through the origin with a slope close to one (Figure 5a). This would indicate that, on average, all reference sites have an *O/E* of 1 and there is no bias. That is, reference sites with a low number of taxa would be predicted with a low number of taxa and vice versa. Figure 5b shows a biased model, where the intercept is negative and the slope is higher than unity. This would result in an inability of such models to predict very high or very low values and will be hereafter called ‘over-averaging’. A high r^2 (correlation coefficient) in these regression models also indicates accurate models and a good choice of environmental variables to predict reference communities.

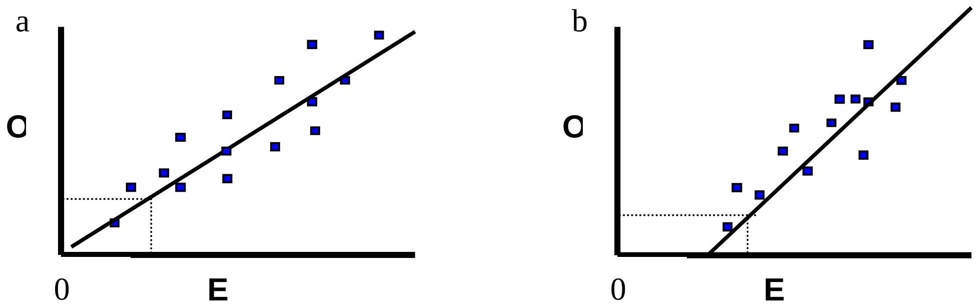

Figure 5 Regression assessment of reference site prediction. Model (a) is a ‘good’ prediction that explains a large amount of the variation between expected and observed. Model (b) shows a bias and will fail to accurately predict low observed values (dashed lines).

2.5.4 How many sites are needed in an ANNA prediction?

Stockwell and Faith (1996) in their original model set the radius of the ANNA to include, on average, 50 percent of the reference sites. They based this on the argument that a ball size that was too large would allow little resolution because most sites are included in the prediction and the predicted community would be close to the grand mean of all communities. If the radius was too small, many sites would have no neighbours, and therefore no predictions would be made. The optimal size/distance for the prediction of a community depends on several factors, including:

- the strength of ecological gradients
- unexplained natural variation
- sampling error
- size of the geographical area of the model
- number and density of sites in the reference database.

Thus, the appropriate number of sites may be model-specific and should be defined individually for each model. Trying to assess and combine the above variables in this study was difficult; therefore, an empirical approach was used to find the appropriate ball size for each model. The reference sites of a dataset were run through the ANNA model 38 times, with ball sizes set to 2, 3, 4 … 40 each time. Each iteration of the model was evaluated by the regression procedure described above, leaving 38 sets of regression parameters. In general, the r^2 was used to determine the ball size (providing the model had an acceptable slope and intercept). A slope was deemed acceptable if was between 0.85 and 1.15 and the absolute value of intercept acceptable at ±2. If the slope or intercept were assessed as not acceptable, the model that was closest to acceptable was chosen. Another criterion to choose the best model was continuity. If the neighbouring models were much worse than the best model, it was suspected that this model was only good by chance, and was rejected.

2.5.5 Variable transformation

A similar problem that cannot be solved on a theoretical basis is determining transformations for the variables. Whether species react to a gradient in a linear, log–linear, or other way is not known for almost all of the variables. An empirical approach was used to determine the appropriate variable transformation.

A method based on a technique analogous to PCC (Principal Axis Correlation) in the Pattern Analysis Package (PATN, Belbin 1994) was used. The first step in a PCC is to ordinate the sites in 2- or 3-dimensional space, based on the presence/absence matrix of the communities. In PATN this is done using a multi-dimensional scaling algorithm. Multi-dimensional scaling (MDS) is a multivariate ordination technique that attempts to project a high-dimensional distance matrix onto a lower dimensional space. PATN uses an MDS algorithm called SSH (semi-strong hybrid multidimensional scaling), which was developed by Faith et al*.* (1987) and is now commonly used by ecologists (Marchant et al*.* 1995, Marchant et al*.* 1999, Moss et al*.* 1999). To stay within one software package, we decided not to use the PATN algorithms and construct a PCC using SAS (SAS Institute 2000). The outputs were compared with the PATN results for verification.

As a measure of the dissimilarity of the benthic communities, the Bray–Curtis dissimilarity index (Bray and Curtis 1957) was used because it ignores joint absences (species absent at both sites). Joint absences are the predominant case in species abundance matrices, and have a strong potential to influence the dissimilarity coefficient and blur the real differences (Field et al*.* 1982, Gauch 1982). Hruby (1987) and Faith et al*.* (1987) found that the Bray–Curtis index is particularly efficient in benthic impact studies in its ability to detect differences in community structure. The index was calculated using the SAS percent distance macro (method = nonmetric) on presence/absence data of the reference invertebrate communities (SAS Institute 2000).

A nonmetric scaling algorithm was used for the MDS, based on the recommendations of Kenkel and Orloci (1986). Kruskal (1964) suggests determining the dimensionality of the projection by the stress level and Wish and Carroll (1982) recommend considering interpretability and parsimony of data representation in the choice of dimensionality. After reviewing outputs from many ordinations, we decided to use a 3-dimensional representation for all of the datasets in this study to automate the process in the model-building procedures. Three-dimensional projections of data are commonly used in aquatic ecology and none of the datasets analysed in this study had an unacceptable stress level ($k > 0.25$), indicating that the ordination solutions represented the data adequately.

Because the original algorithm of fitting environmental variables to the community ordination in PATN is sparsely documented, we decided to use a multidimensional functional regression with the environmental variables as predictors and the three MDS scores as a response for each site. PROC TRANSREG in SAS (SAS Institute 2000) was employed to compute the correlation.

A problem with using functional regressions is that the least squares are fitted to all dimensions at the same time. This makes it impossible to calculate r^2 values. PATN uses largely undocumented Monte Carlo permutations to determine the strength and significance of the correlations, while SAS computes a convergence algorithm that is not clearly specified. The final results of the correlation analyses are vectors with a certain direction and a certain length in the three-dimensional ordination space.

To determine whether a variable should be transformed, three PCCs were performed on each predictor:

- Untransformed
- Square-root transformed
- Log (x+1) transformed.

If the r^2 of one of the transformations was much better than the untransformed $r^{2,}$ the variable was transformed for model building and prediction of communities.

2.5.6 Weighting variables

Another issue in the construction of ANNA models is the problem that in the original approach (Stockwell and Faith 1996), all the environmental variables that were measured contribute to the prediction. Several studies (Faith and Norris 1989, Marchant et al. 1997) have shown that a macroinvertebrate assemblage responds to different disturbances in a variety of strengths and magnitudes. Thus, the environmental predictors in the ANNA model were weighted.

In Figure 6, all the predictors are weighted equally and the test site (marked with an arrow) is the closest test site to the reference site in the middle.

Imagine predictor 1 has more influence on the structure of the community than does predictor 2. In this case we stretch the axis of this predictor by weighting it with a coefficient of its importance and observe how another site appears to be closer (Figure 7). With this approach we ensure that the environmental space from which we choose our most similar sites for prediction is constructed according to the environmental factors that drive the community.

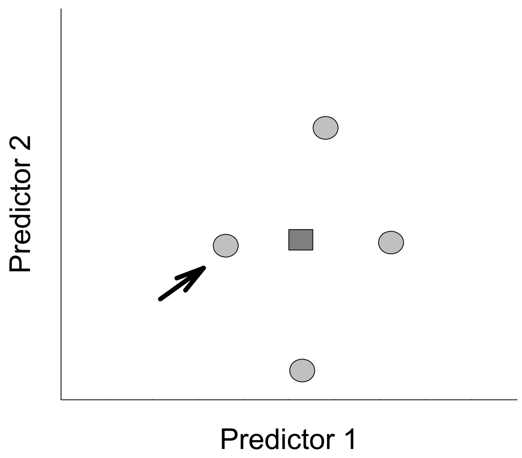

Figure 6 Site selection in an unweighted approach. The arrow indicates the closest site.

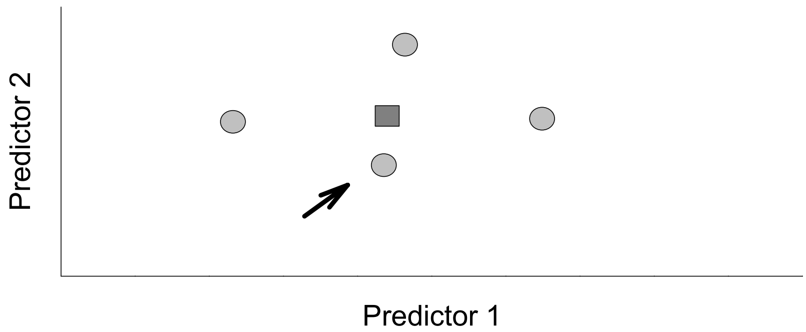

Figure 7 Site selection in a weighted approach where the predictor 1 axis has been stretched. The arrow points to the closest test site.

Weighting mechanism

In initial versions of ANNA, the weights were determined by the PCC-based approach used for the transformations. One problem with a PCC-based approach is the fact that a functional regression does not eliminate or reduce correlated predictor variables. In a predictive model this can cause amplification of a single gradient if this gradient is represented by more than one variable.

Stepwise regression is a popular approach to eliminate correlated variables and selects the strongest combination of variables while dropping minor and correlated influences (Neter et al*.* 1996). Unfortunately, stepwise approaches cannot be applied to functional regression, because no statistical estimator (r^2) is provided to compare the quality of the model. Instead of applying a functional regression to the whole dataset, an MDS was carried out as described above and a stepwise multiple linear regression of the environmental variables was run on each of the axes. This stepwise regression was calculated using PROC REG in SAS with the criterion $r^2 = 0.1$ for entry and removal of variables. Three sets of weights are obtained from the regression analysis.

Unlike the PCC-based approach where the single set of weights can be directly applied to the standardised environmental predictors, the set of three weights from the regressions were combined. This was done by substituting environmental predictors from the regression equations into the Euclidean distance. The intercepts can be left out, because in the distance formula they would be subtracted from each other. The modified Euclidean distance is calculated as follows (Equation 1):

$$d = \sqrt{(\sum a_n chem_{in} - a_n chem_{jn})^2 + (\sum b_n chem_{in} - b_n chem_{jn})^2 + (\sum c_n chem_{in} - c_n chem_{jn})^2}$$

Equation 1

with *a,b,c* = regression coefficients

i,j = observations (sites)

n = number of chem. variables.

The graphical representation (Figure 8) gives a clearer explanation of the procedure and is the basis for the computational strategy.

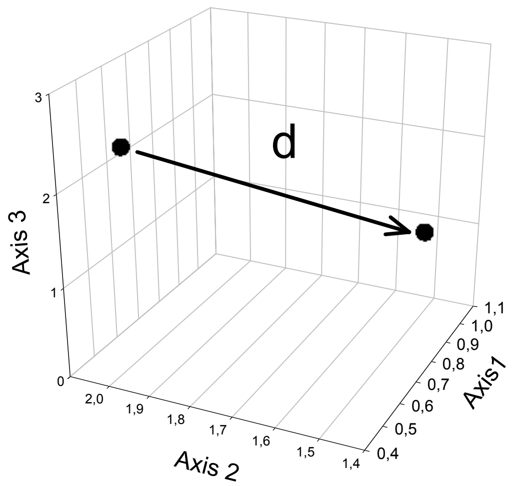

Figure 8 Graphical representation of the steps in the 'stepwise' weighting approach. Axes represent the distance scores according to each of the three weights. Dots represent sites, and *d* is the final Euclidean distance.

With the substitution, two sites are ordinated in a three-dimensional space. The distance of the sites on a single axis was calculated by the Manhattan distance of their weighted predictors in the corresponding set of scores. The use of Manhattan distance, in this study calculated with the percent distance macro in SAS (method = city), derives from the brackets in Equation 1 above. After determining the axis scores, these are combined into the final distance by computing the Euclidean distance of the ordinated points in the three-dimensional coordinate system. This distance represents the Euclidean distance in environmental space, weighted by the stepwise regression approach, which was used for all the models.

Interpolation of missing data

One of the major advantages that ANNA has over AUSRIVAS or other prediction methods is that the model can deal with missing data in the predictor variables. The discriminant function analysis in AUSRIVAS requires a complete set of predictor variables. Variables with missing data have to be removed from the dataset. ANNA can avoid this problem by interpolating the Euclidean distance between two sites even when some predictor variables have missing values. For example, in the full dataset the Manhattan distance between the two sites is 1, because only 1 variable (Variable 3) is different in the test site (Table 3). In the second example, Variable 2 is missing. Leaving the total distance at a value of 1 would make the assumption that Variable 2 has a value of 1. In reality, it could be any value between 0 and 1 (Table 3).

Table 3 Calculation of Euclidean distances with missing data.

	Var 1	Var 2	Var 3	Var 1	Var2	Var 3
Reference site	1	1	1	1	1	1
Test site	1	1	0	1	.	0
		Distance = 1			*Distance = 1.5*	

To avoid this assumption, the final distance was calculated as follows.

$$d_{comp} = d_{prem} \cdot \frac{n}{n_{pres}}$$ **Equation 2**

where: d_{comp} = final distance, n = number of total variables, d_{prem} = preliminary distance, and n_{pres} = number of non-missing variables.

In this case, if the preliminary distance equals 1 when using two present variables (as in Table 3), the final distance that incorporates three variables would be 1.5.

An additional problem when dealing with missing data in ANNA is that the variables are weighted differently. Missing variables that have a higher weighting also need more influence in the estimate of the final distance. Therefore Equation 2 was modified as follows (Equation 3):

$$d_{comp} = d_{miss} \cdot \frac{\sum w}{\sum w_{pres}}$$ **Equation 3**

where d_{comp}= final distance, Σw = sum of total variables, d_{miss} = distance incl. missing, and Σw_{pres} = sum of present variables.

In the example table (Table 4), where only half the weights are present in Variable 1 and Variable 3, the final distance is greater because the variable with the missing value is the predictor with the highest influence.

Predictor Variables for modelling the taxa expected (*E*)

Variables used in AUSRIVAS models (to predict the expected, *E*) were taken as a guide for the variables that could be used to build ANNA models capable of predicting *E* (Table 5). In some instances surrogates for variables had to be used because many of the AUSRIVAS predictor variables are measured on site, whereas the ANNA modelling in this project did not involve site visits.

Table 4 Calculation of Euclidean distances with missing data in weighted variables.

	Var 1	Var 2	Var 3	Var 1	Var2	Var 3
Weight	0.5	1.5	1	0.5	1.5	1
Reference site	1	1	1	1	1	1
Test site	1	1	0	1	.	0
	Distance = 1			*Distance = 2*		

Table 5 Variables used in AUSRIVAS models and in ANNA models to predict the expected number of taxa.

Variables in AUSRIVAS models	Variables used in ANNA *E*-type models
Location	
Altitude	Altitude
Latitude	Latitude
Longitude	Longitude
Stream order	Catchment area
Distance from source	Catchment area
Catchment area	Catchment area
Water chemistry (conductivity, alkalinity)	Catchment soils
Geomorphological characteristics (depth, riffle area, stream width, velocity, bank width, stream slope	Stream power
Stream vegetation (river shading, shrubs/vines, trailing bank vegetation, trees >10 m, vegetation cover, vegetation type, riparian width	No surrogate available
Substrate (reach bedrock, boulders, cobble, pebble, gravel, mean substratum, heterogeneity	Stream power
Macrophyte diversity	No surrogate available
Habitat area	No surrogate available
	Additional predictors
	Catchment relief
	Geology
	Soil

2.5.7 Additional predictors

Three additional variable predictors were included in the suite of predictors included in the model building process: catchment relief, geology, and soil characteristics. The literature indicates that these factors can be important in determining biotic community structure.

Catchment relief gives an indication of how incised a catchment is; it is a characteristic that influences runoff and movement of sediment in a catchment. Catchment relief was characterised by calculating the mean slope of the reach sub-catchment.

The underlying geology in a river basin determines channel morphology and sediment type and hence the macroinvertebrate community (Richards et al. 1997). Geology also has an influence on stream chemistry and, through it, periphyton communities, with alkaline rock types showing the most effect (Biggs 1995). Cannan (1999) grouped the geology along the Frome River, southern England, into three categories (bedrock, chalk, alluvial gravel) and found a significant correlation with biotic differences.

The geological predictor variables for this were derived from the Department of Agriculture, Fisheries and Forestry Australia's (AFFA) AGSO 25mg database. This database is a polygon coverage of Australia's bedrock geology. Geological types in that database were assigned to one of five categories (Table 8) based on the dominant rock type. Within each reach sub-catchment, the percentage of the total area of each category was determined using a modelling input.

Catchment soil characteristics are also related to macroinvertebrate communities (e.g., Richards et al. 1993, Richards et al. 1997). Soil effects on stream condition can be chemical, as a result of the parent rock chemistry, or result from their structure; in particular, these two factors can influence runoff quantities and the extent to which soil particles are mobilised by runoff. Soils were classified on both these characteristics. The Australian Soil Resources Information System (ASRIS) coverage – *atext* (A horizon texture) – was used to generate infiltration categories. Soil textures in the *atext* coverage were assigned to infiltration categories following the United States Department of Agriculture (USDA 1992) classification in which soils were grouped into categories based on their permeability and hence their propensity for generating runoff (see Table 6).

Table 6 ASRIS texture categories and corresponding ARC infiltration categories.

ASRIS A-horizon soil texture categories	ARC infiltration categories
Sands	High infiltration
Sandy loams	Moderate infiltration
Loams	Moderate infiltration
Clay loams	Low infiltration
Light clays	Low infiltration
Clays	Very low infiltration

Soils were also categorised based on their acidity; a factor that has also been found to be an important predictor variable in AUSRIVAS models. The ASRIS soil database *srt* classifies soils into eight classes based on their acidity (Table 7). For this project only four acidity classes were used (see Table 8).

Table 7 ASRIS acidity categories and corresponding ARC acidity categories.

ASRIS classes	ARC classes
Alkaline	Alkaline
Alkaline and neutral	Alkaline
Neutral	Neutral
Acid, neutral, and alkaline	Neutral
Alkaline and acidic	Neutral
Neutral and acidic	Acidic
Acidic	Acidic
Strongly acidic	Strongly acidic

Table 8 Sources of *E*-type predictors and data categories used.

Variable	Source	Continuous or categorical
Altitude (m)	Altitude of midpoint of reach	Continuous
Latitude (dec degrees)	Latitude of midpoint of reach	Continuous
Longitude (dec degrees)	Longitude of midpoint of reach	Continuous
Catchment area (km2)	Calculated from DEM	Continuous
Geology	Geology database (AGSO 25mg coverage)	Categories Acid volcanics Basic volcanics Limestone Sandstone Siltstone, shale, mudstone Metamorphics
Reach stream power	Calculated from reach slope and catchment area	Continuous
Mean slope of sub-catchment	Calculated from DEM	Continuous
Soil	ASRIS coverage	Infiltration categories High infiltration rate (sands and gravels) Moderate infiltration rate (deep, coarse textures) Low infiltration rates (fine texture) Very low infiltration rate (clayey soils) Acidity categories Strongly acidic Acidic Neutral Alkaline

Details of the predictor variables included in the E-type *modelling are given above (Table 8). The source of the data and the categories used if the predictor variable was categorical have been included.*

Predictor variables for modelling the observed number of taxa (MO)

The observed biota (*MO*) for reaches that had no AUSRIVAS assessment were modelled using the ANNA procedure. The *MO* models included both reference and test sites and additional predictor variables that provide a measure of the impacts that reduce the numbers of different kinds of animals. The set of variables used to model the expected condition was taken as the core set and augmented by the variables below identified in the literature as impacting on aquatic biota. They can be classified as follows:

- Water quality surrogates (catchment land use, point sources)
- Riparian zone quality

- Flow regulation.

Catchment land use measures

A range of land use changes have been associated with changes to water quality and aquatic biota (e.g. Boulton et al. 1997, Brewin et al. 1995, Hanchet 1990, Jowett et al. 1996, Lenat and Crawford 1994, Mitsch and Urbikas 1980, Townsend et al. 1997). From these studies a set of land use categories were identified for use in this project (Table 9). The land use data used were derived from the Audit land use mapping, with land use categories aggregated to correspond to the categories identified in the studies referred to above (Table 9).

Evidence for land use effects on aquatic biota at the catchment scale tend to be circumstantial as researchers must rely on ‘natural experiments’ in an attempt to disentangle the effects of land use from other factors. It is thought that land use effects on aquatic biota are mediated by changes to the nutrient supply, the sediment supply, flow, and pollutants. Land use categories found to be correlated with stream biota include: pasture, pine, native forest, intensive agriculture, mixed forest, and urban. The variables used were the percentage of the reach sub-catchment falling into each of the land use categories. A decision was taken not to use the land use of the entire catchment upstream of a reach as it was felt that land use in the immediate sub-catchment would have more influence on stream biota, resulting in more robust models.

Table 9 Land use categories used to model *MO*.

Categories identified for modelling	Audit land use mapping categories
Urban	Transport Utilities Urban uses Institutional uses (excluding water storage)
Agriculture – grazing	Grazing
Agriculture – tilled	Agricultural land Permanent cropping Horticulture (including vegetables, excluding fruit)
Agriculture – orchards	Horticulture (excluding vegetables, including fruit)
Conservation	Wilderness area Protected landscape National park Habitat/species management area Strict nature reserve National monument Managed resource protected areas Unmanaged land Water Institutional uses (including water storage)
Forestry	Production forests Farm forestry Plantations

2.5.8 Point sources

A wide range of materials released from point sources have the potential to impact on aquatic biota. For the majority of such materials there is limited information on their impact on biota, and even less information on the amount of material being released into streams. We used the National Pollutant Inventory (NPI) data as a source of data on point-source emissions. The NPI is a dataset compiled by Environment Australia on the annual emissions of a suite of 30 pollutants from facilities across the country. These data are limited in several important ways: the NPI program has only just commenced and contains data for only one year, it does not detail how pollutants have been lost, and it does not require reporting by agricultural facilities that may have point-source emissions (e.g. feedlots). Nevertheless, it is currently the only national database on pollutants that might impact on streams and was used to generate modelling variables.

Pollutants reported in the NPI were grouped into four categories (total phosphorus, total nitrogen, trace metals, and organic pollutants) based on differences in their impact on aquatic biota. Different pollutants within these categories have different toxicities, and so to arrive at a measure of pollutant hazard the annual emission of each pollutant was divided by its ANZECC guideline value (ANZECC 2000) and the resulting 'hazard' values summed across each category. This gave us a measure of pollution hazard for each of the four pollutant categories at each reach for which there were NPI data. These measures were used as modelling variables.

2.5.9 Riparian zone quality

The riparian zone index described in detail in the habitat methods was used as a modelling variable.

2.5.10 Flow regulation

Each reach in the area assessed was assigned to one of two flow regulation categories, regulated or unregulated, determined by the presence of a dam upstream. For this project dams were defined as structure greater than 10 m in height occurring on the AUSLIG waterbody database mapped from 1:2.5 M topographic maps. All reaches downstream of such regulation structures were defined as regulated in this project. This measure of flow regulation was used as a modelling variable.

2.5.11 Creating *MO*/*E* from *MO* and *E* values

After predicting expected (*E*) and modelled observed (*MO*) taxa for reaches without AUSRIVAS assessments, the taxa probabilities were joined to give a score analogous to the AUSRIVAS *O*/*E*. The steps to do this are summarised in Figure 9. The expected number (*E*) of taxa was calculated by summing the probabilities ≥ 0.5 in the same way as AUSRIVAS. The modelled observed (*MO*) value was calculated by summing all of the predicted probabilities. This differs from the AUSRIVAS approach where only taxa that are predicted with a greater than 50 percent chance are considered in the calculation of *O*. When dealing with real presence/absence data, presence always equals a value of 1. The taxa probabilities for the modelled observations would hardly ever reach a value of 1. If contributing taxa in the *MO* were only included at the 50 percent or 75 percent level in the same way as the expected taxa for AUSRIVAS, the score would be biased downwards because even the higher probabilities would not reach 1. Therefore, the modelled observed was calculated by summing all modelled taxa probabilities, not just those with probabilities greater than 50 percent.

2.5.12 Modelling regions

It was decided to build regional models to accommodate regional heterogeneity. The Northern Territory, South Australia, and Tasmania were modelled as entities. Queensland was modelled in two parts: coastal and inland. Western Australia was also modelled in two parts: the south-west and the rest of the state. Cross-border models were possible between New South Wales and Victoria as they used the same sampling protocol. Data from New South Wales and Victoria were modelled in four regions: a highland model (>1000 m), New South Wales east of the divide, Eastern Victoria (EPA regions 2, 3, and 4) and a model for western NSW and western Victoria.

Models in each region were built with different seasons and habitats. The best Expected type model for each region then determined the final season/habitat combination (Table 10).

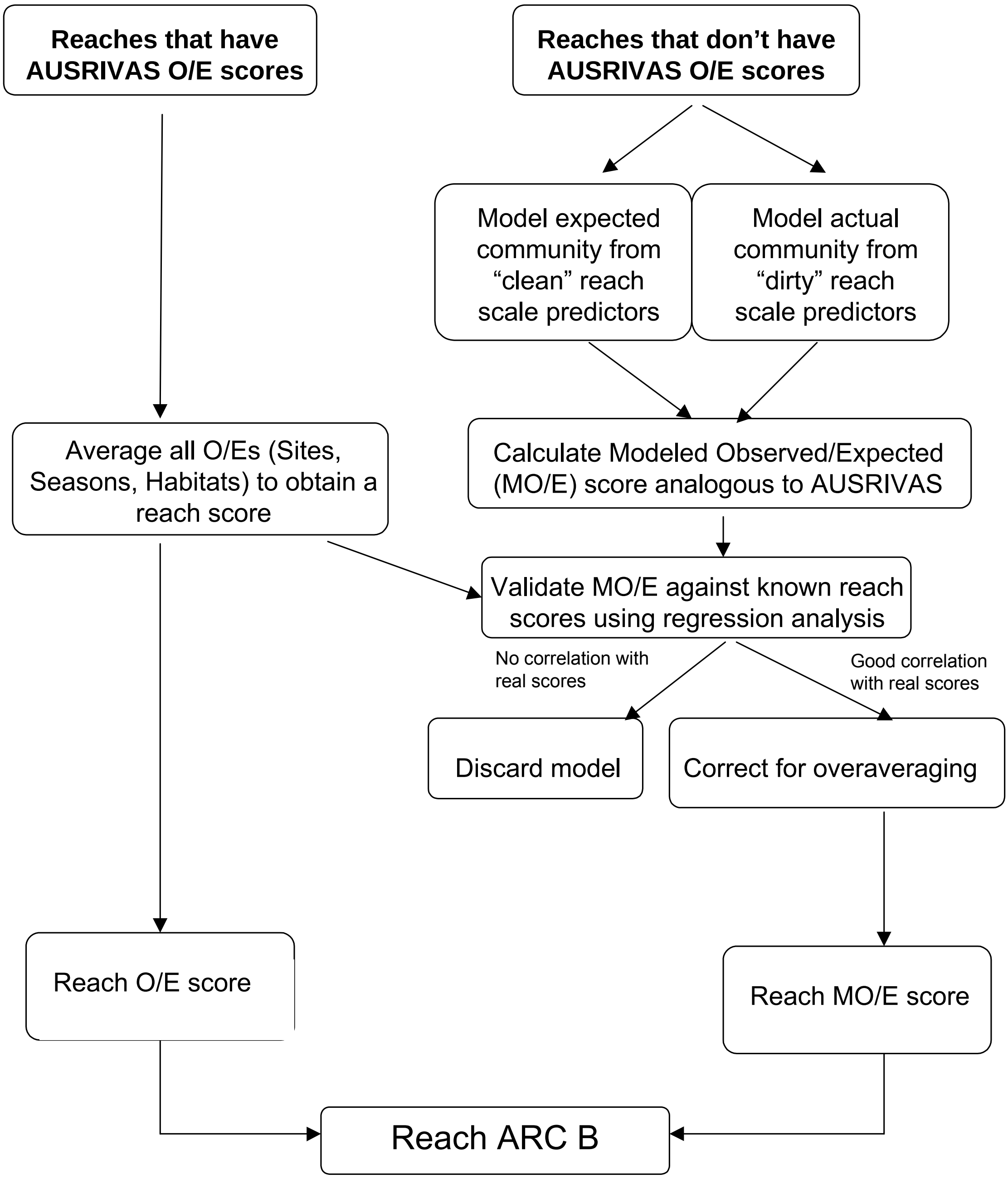

Figure 9 Assessment of River Condition – Biota (ARCB). Flowchart of steps.

Table 10 Seasons and habitats used for the ten models.

Model	*MO* r^2
NSW east	Combined edge
NSW–VIC highland	Combined edge
NSW–VIC west	Combined edge
NT	Late dry season edge
QLD_Coast	Spring edge
QLD_Inland	Autumn edge
SA	Combined edge
TAS	Spring riffle
VIC east	Combined edge
WA south-west	Wet season channel
WA rest	Dry season channel

2.5.13 Model validation

Model validation consisted of two phases. First, the models were evaluated on their own by predicting reference sites that had been removed from their own model. This was done for both the expected models that only contained natural predictors, as well as for the modelled observed, derived from measures of human disturbance. Satisfactory models could be built for all of the regions (Table 11). Most *E*-type models were better than average AUSRIVAS models.

Table 11 Correlation coefficients of the internal model validations for *E*-type and *MO*-type ANNA models.

Model	*E*-Model r^2	*MO*-Model r^2
NSW east	0.38	0.50
NSW–VIC highland	0.46	0.57
NSW–VIC west	0.65	0.75
NT	0.33	0.34
QLD_Coast	0.52	0.37
QLD_Inland	0.55	0.32
SA	0.46	0.43
TAS	0.30	0.23
VIC east	0.55	0.56
WA south-west	0.50	0.50
WA rest	0.57	0.62

The modelled scores were validated against the AUSRIVAS values. The modelled observed (*MO*) were regressed against the AUSRIVAS observed scores. All AUSRIVAS scores available within a reach were averaged to obtain an overall *O*/*E* score for a reach. The *MO*/*E* scores were then compared to the *O*/*E* scores using the same procedure. This regression was used to decide whether the model for a modelling region was acceptable or not. Only 4 out of the 10 models proved to be acceptable for modelling *MO/E* scores at reaches (Table 12).

Table 12 Correlations between AUSRIVAS *O/E*s and modelled values for the four regions with acceptable models.

Model	r^2
NSW east	0.52
SA	0.37
TAS	0.19
VIC east	0.42

Although the models for the Northern Territory and the western regions of NSW and Victoria would have been statistically defendable because they had significant regressions at the 5 percent level, evaluation of the prediction showed that the error in some predictions would have been too large to justify reporting at even the basin scale.

One possible reason why only four models were successful is that the data used for validation were also subject to error. The validation of *MO/E* requires comparison of two measures, each of which is composed of two sub-indices. The two indices comprising the *MO/E*, modelled for this project, individually explain only about 50 percent of natural variation (Table 12). Together, the *MO/E* score could be expected to explain about 25 percent of the natural variation. Additionally, *O/E* scores from an average AUSRIVAS model, against which the *MO/E* scores are validated, explain only 30–40 percent of the variation in the biota. Even the AUSRIVAS *O* score, a measured not modelled value, is subject to natural variation and sampling error.

A likely source of the error described above may be that prediction and validation was carried out at the reach scale. When examining data for macroinvertebrate communities within a reach, and their corresponding AUSRIVAS scores, we found considerable variation in both. This will influence the model quality, because in the model building process underlying gradients will be blurred by the fact that one single reach scale set of predictors is used to predict multiple communities.

A third problem arises from the use of predictor variables derived from remotely sensed data. Both AUSRIVAS and ANNA models have been developed based on both large-scale and local predictors. Local predictors (e.g. detailed information on substrate and riparian vegetation) have been important predictors in many of the AUSRIVAS and ANNA models constructed in the past. Remotely sensed data can serve at best as proxies for such information.

All of the models showed a bias, which was attributed to ‘over-averaging’. To remove this bias, the predicted *O/E*s were corrected for intercept and slope. The equations for the corrections were (Equation 4):

New South Wales east: $MO/E_{final} = (MO/E–0.52)/0.43$

South Australia: $MO/E_{final} = (MO/E–0.39)/0.58$

Tasmania: $MO/E_{final} = (MO/E–0.64)/0.29$

Victoria east: $MO/E_{final} = (MO/E–0.49)/0.42$.

Equation 4

2.5.14 Estimation of errors at the basin level

To estimate the error that the models would cause when the scores are aggregated to basin scores, simulations were run for every modelling region. For each simulation a normally distributed random error between 0 and 1 was added to each of the AUSRIVAS *O/E* scores. The basin score was then calculated from the modified reach scores and the percentage of basins that changed bands was recorded. This simulation was repeated 20 times for each region, allowing an error curve and statistics to be generated (Table 13, Figure 10).

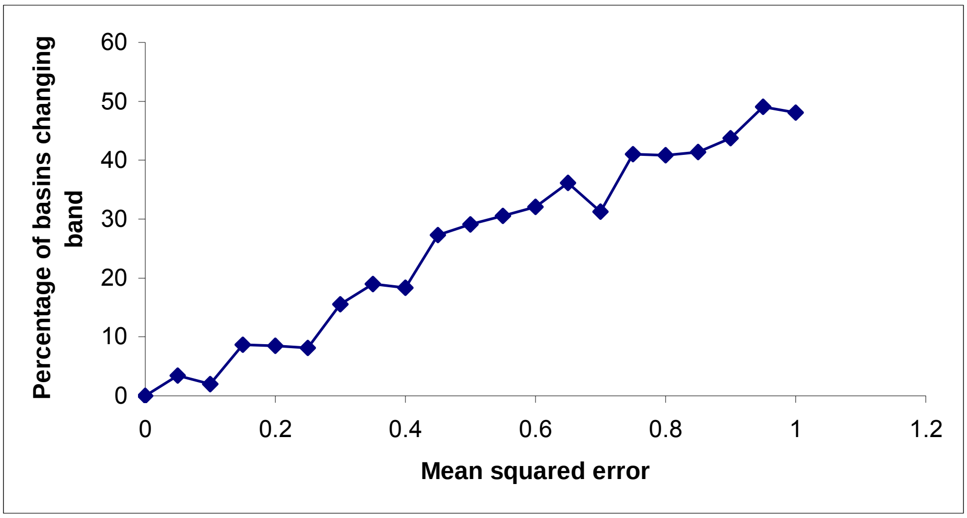

Figure 10 Number of basins misclassified with increasing model error.

Table 13 Estimated percentage of basins misclassified as a result of model error.

Region	Mean squared error	Percentage of basins assigned to incorrect band
NSW east	0.23	7
SA	0.22	10
Tasmania	0.32	13
Vic east	0.19	7

2.6 Calculating the reach and basin biota index

2.6.1 Reach biotic index

The biotic index for reaches is derived from the AUSRIVAS *O/E* scores, or ANNA *MO/E* modelled values. Where neither was available in a reach, no biotic index was recorded for that reach.

2.6.2 Reaches with AUSRIVAS scores

The *O/E* taxa score was used to derive the biotic index for reaches that had AUSRIVAS data. Many of these reaches had both multiple AUSRIVAS sites, and multiple *O/E* scores at each site. The multiple scores at each site resulted from model outputs for different habitats, or on different sampling occasions. A mean *O/E* score was calculated for reaches with more than one AUSRIVAS score, using all the *O/E* scores in the reach. This meant that *O/E* scores from different models of potentially different quality were combined. Work by Parson and Norris (1996) found that model outputs from different seasons and habitats did not significantly differ, suggesting that using the mean of all *O/E* scores is a valid way of using this comprehensive dataset.

Mean *O/E* values were mapped to fall within the range 0 to 1 by truncating values higher than 1. AUSRIVAS scores higher than 1 fall within AUSRIVAS band A (equivalent to reference), so it is appropriate to map these scores to the biotic index band ‘reference’. The mean AUSRIVAS scores mapped to the range 0 to 1 comprise the biotic index for reaches with AUSRIVAS data.

2.6.3 Reaches without AUSRIVAS data

Where reaches did not have AUSRIVAS data, ANNA models were developed to produce a single modelled score for each reach. This *MO/E* value comprised the biotic index for such reaches. Where an acceptable ANNA model could not be developed and there were no AUSRIVAS data, a biotic index could not be determined for that reach.

In areas where acceptable ANNA models could be built, there were still a small number of reaches for which an *MO/E* value was not calculated. These were reaches that did not have a complete set of predictor variables and so could not be modelled.

Basin biotic index

A biotic index was calculated for each AWRC basin for which there were sufficient data. There were two categories of basin: those for which we had only AUSRIVAS data, and those for which we had both AUSRIVAS and ANNA data. Different approaches were used to calculate the basin score in each of these categories.

2.6.4 Basin assessment: AUSRIVAS data only

For Queensland, Western Australia, the Northern Territory, the western part of New South Wales (all basins west of the coast), and the western part of

Victoria, acceptable ANNA models could not be developed and basin assessments were developed from AUSRIVAS data alone.

The determination of a spatial assessment required aggregation of reach assessments falling within each AWRC basin. A crucial consideration was whether both test and reference data should be used in the basin assessment. In the original choice of locations for test and reference sites, neither were chosen to provide an unbiased estimate of river health at the basin scale. Reference sites were chosen to represent pristine or minimally disturbed conditions. Their inclusion in a basin assessment would bias any estimate of basin condition towards the ‘reference’ condition. Test sites were chosen for a range of reasons. Many were chosen because a problem was known or suspected at a site. These test sites would bias a basin assessment towards a poorer condition. Others were chosen as representative of the conditions in a region, and so represent an unbiased measure of the basin condition. Given this understanding, only test sites were used in the basin assessment because it was judged that they would provide the least biased estimate of the basin condition.

Where there was more than one AUSRIVAS output for a test site, the mean of all outputs was taken to give an average test site score. Small tributary streams in the upper parts of catchments are usually numerous, but represent only small catchment areas. A true basin estimate of health should consider the basin area represented by each measurement. Thus, the contribution of each site to the basin estimate was weighted by the contributing catchment area. This was achieved by using the distance from the source as a surrogate for catchment area and weighting *O/E* ratios accordingly. A mean of sites was then taken (Equation 5).

$$O/E_{Basin} = \frac{\sum_{i=1}^{n} OE_i \cdot DFS_i}{\sum_{i=1}^{n} DFS_i}$$

with OE_i = *O/E* score of the *i*-th reach in the basin

DFS_i = distance from source of the *i*-th reach in the basin

Equation 5

Test sites with O/E ratios above reference (around O/E >1.15) may be naturally taxa-rich communities or they may be suffering from mild organic enrichment. Sites with OE ratios above reference were excluded from the basin assessment aggregation because the higher O/E scores would weight or create a bias towards the reference end even if the community at the site was being adversely affected. Basins in which the catchment of the assessed reaches was less than 50 percent of the total basin area were classed as “non-assessed”. Basins with fewer than three test sites were also classed as “non-assessed”.

2.6.5 Basin assessment: AUSRIVAS and ANNA data

Where acceptable ANNA models could be developed, the MO/E score was used to provide a biotic index for reaches for which there was no AUSRIVAS score. Acceptable models were produced for the NSW coastal region, Tasmania, South Australia and the eastern part of Victoria. In these areas MO/E scores, together with both reference and test AUSRIVAS data were used to produce a basin assessment. In contrast to the basins with just AUSRIVAS data, both test and reference data were used because, in conjunction with the MO/E values, each sub-catchment in the basin has a biotic index. Using the AUSRIVAS data alone, only a potentially biased subset of the sub-catchments had a biotic index.

The biotic index at the reach scale was weighted by catchment area to ensure that a basin assessment was not dominated by a number of test sites in small headwater or tributary streams. Catchment area was used to weight the reach biotic index, as data on distance from source was unavailable for the reaches with MO/E scores (Equation 6). These two variables are highly correlated.

$$O/E_{Basin} = \frac{\sum_{i=1}^{n} OE_i \cdot A_i}{\sum_{i=1}^{n} A_i}$$

with OE_i = O/E score of the *i*th reach in the basin

A_i = catchment area of the *i*th reach in the basin

Equation 6

2.7 Condition bands

With other indices, four conditions were identified by dividing the 0–1 range of the index: 0–0.25 severely modified, 0.25–0.5 substantially modified, 0.5–0.75 moderately modified, and 0.75–1.0 largely unmodified. For the AUSRIVAS data, bands had been developed before this project (Table 14). These were retained in this project as far as possible. AUSRIVAS bands are based on the distribution of the reference site scores and are different for each model. As we were combining data from different models, we needed to also combine the band information. The key characteristic for AUSRIVAS bands is the delineation between ‘band A’ and ‘band B’, with the other bands spread evenly across the remaining space. For this project the delineation between band A (reference) and band B (significantly impaired) was calculated by taking the mean of the delineation points for all the models within a state or territory. Bands below this point were defined by an even breakdown of the remaining range (Table 15).

Table 14 Description of the bands used for AUSRIVAS.

Band	Condition labels for the Audit	Description
Band X	Not used	Above reference with about 20 percent or more of the different kinds of animals than expected
Band A	Reference	Equivalent to reference with similar numbers and types of animals expected
Band B	Significantly impaired	Below reference with a loss of between about 20–50 percent of the different kinds of animals expected
Band C	Severely impaired	Well below reference with a loss of between about 50–80 percent of the different kinds of animals expected
Band D	Extremely impaired	Impoverished with loss of more than about 80 percent of the kinds of animals expected

Table 15 Bandwidths for each state.

State	Band X	Band A (reference)	Band B (significantly impaired)	Band C (severely impaired)	Band D (extremely impaired)
ACT	>1.14	0.86–1	0.55–0.86	0.28–0.55	0–0.28
NSW	>1.17	0.83–1	0.55–0.83	0.28–0.55	0–0.28
QLD	>1.19	0.81–1	0.55–0.81	0.28–0.55	0–0.28
TAS	>1.16	0.84–1	0.55–0.84	0.28–0.55	0–0.28
VIC	>1.16	0.84–1	0.55–0.84	0.28–0.55	0–0.28
SA	>1.18	0.82–1	0.55–0.82	0.28–0.55	0–0.28
WA	>1.17	0.83–1	0.55–0.83	0.28–0.55	0–0.28
NT	>1.15	0.85–1	0.55–0.85	0.28–0.55	0–0.28

For this project, no coding has been given for Band X because the AUSRIVAS protocol for assessing such sites is to review the data and possibly revisit the site. The judgement has been made that the occurrence of more kinds of animals than expected is potentially not as serious a river health problem as loss of animals. Data from sites classed as Band X were not used in this assessment.

In conclusion:

Assessments are broad-scale and represent the average condition of the rivers and streams in a basin as assessed by sites sampled relative to minimally damaged reference sites.

Assessments are based on the FNARH sites alone, which may have introduced some bias towards poorer health than might have been assessed if sites had been chosen randomly across an entire basin.

3 Catchment Disturbance Index (CDI)

3.1 Components of the Catchment Disturbance Index

The focus for this index was to detect or provide a measure of anthropogenic changes that ultimately impact the river condition and the biota. Catchments influence a river through large-scale controls on hydrology, sediment delivery, and chemistry (Allan and Johnson 1997). Much of the degradation in Australia's rivers results from land use practices in surrounding catchments (Boulton and Brock 1999). The measures chosen for use in the catchment disturbance index are ones that characterise the changes in land surfaces.

Sources of the data

The data for the index were restricted to those that would show land surface changes (see Table 16). Conservative catchment descriptors like soil type and geology were not used. Land use data were used to provide details of the current land uses occurring in the catchment. To obtain a measure of change in land use over time, the Agricultural Land Cover Change dataset was used. Included in the dataset was a measure of the loss of woody vegetation over the period 1990–95. Although these data were collected recently and represent a short period, they provide information on change in the recent past.

The third type of land surface information used was spatial data on infrastructure, remembering that many types of infrastructure have the potential to impact streams and rivers. Infrastructure information is not included in the land use coverage, and needs to be included to ensure that the effects of structures in the catchment are measured.

A dataset that was not available was a national spatial coverage of current land management practices. Land management practices have the potential to significantly change the impacts of particular land uses. In a future compilation of the ARC it would be useful to include information of this nature if it were available at a national level.

Table 16 Sources of data used to calculate the catchment disturbance index.

Input data	Source	Coverage	Data type
Infrastructure	Wild Rivers dataset	All Australia	Cartographic data: Presence/absence of different infrastructure features Raster – 250 m grid
Agricultural Land Cover Change (ALCC)	Audit office	Majority of Australia	Satellite imagery: Vegetation cover and change Raster – 100 m grid
Land use	Audit data	All Australia	Satellite imagery: Land use categories Raster – 100 m grid

3.1.1 Wild Rivers data – infrastructure

The Wild Rivers dataset contains data layers on a suite of variables including land use, settlements, point sources, infrastructure, impoundments, diversions, and levees. Of these, only the land use and the infrastructure data describe impacts on the catchment that could impact on the aquatic biota. More recent land use data were obtained from the Audit, thus only the infrastructure data from the Wild Rivers data were used in the calculation of the CDI.

The Wild Rivers infrastructure data consisted of a separate coverage for each of the seven categories of infrastructure. Each grid cell within each coverage had a value of either 0 or 1 depending on the presence of a structure of that particular category (e.g. railway). The seven coverages were aggregated into a single infrastructure coverage. Where two different infrastructure types occupied the same grid cell, only the structure with the highest weighting was recorded (Table 17).

The catchment disturbance generated by infrastructure was assessed by the areal extent of each infrastructure category within the reach catchment, adjusted by the weights applied to the different infrastructure types (Equation 7). The derivation of the weights is discussed in detail in the section on integration and aggregation.

$$I = 1 - ((I_1 * w_1) + (I_2 * w_2)...)$$ **Equation 4.7**

where I = infrastructure measure, I_1 = fraction of the catchment of infrastructure category 1, w_1 = the weight for infrastructure category 1, etc.

Table 17 Wild Rivers infrastructure categories and ARC infrastructure weights.

Category	ARC infrastructure weights
Main sealed road	0.70
Other sealed road	0.70
Railway	0.22
Unsealed road	0.55
Vehicle track	0.55
Utilities (power, pipes)	0.07
Walking track	0

3.1.2 Land use data

The land use data used for this project came from the Audit land use mapping project. This project provided a national coverage of data gridded at 250 m cell resolution, across a wide range of land use categories and attributes. The intention of the catchment disturbance index was to provide a measure of the impact of catchment activities on the aquatic biota. Catchment activities could result in changes to hydrology (as a result of changes to infiltration or interception), changes to water quality (as a result of changes to non-point sources), or changes to habitat. For the purposes of the ARC, the Audit land use categories were aggregated into categories that better reflected land use effects on aquatic biota (see Table 18).

Weights were applied to the land use categories to account for the different impacts of different land use categories. The derivation of the weights is discussed in detail in the section on integration and aggregation.

The catchment disturbance generated by land use was assessed by the areal extent of the each land use category within the reach catchment, adjusted by the weights applied to the different categories (Equation 8).

$$LU = 1 - ((F_1 * w_1) + (F_2 * w_2)...)$$ **Equation 8**

where LU = land use measure, F_1 = fraction of the catchment that is category 1 land use, w_1 = weight associated with land use 1, etc.

Table 18 Audit land use categories, ARC categories, and weights.

Audit land use categories	ARC categories	ARC weights
Horticulture, orchards, legumes, cotton, rice, non-cereal forage crops	Intensive and irrigated agriculture	0.70
Transport Utilities Urban uses Institutional uses	Urban	0.68
Cropping not included in intensive and irrigated agriculture	Dryland cropping	0.48
Production forests Farm forestry Plantations	Forestry – Eastern Australia	0.20
Production forests Farm forestry Plantations	Forestry – Western Australia	0.28
Grazing	Grazing	0.33
Wilderness area Protected landscape National park Habitat/species management area Strict nature reserve National monument Managed resource protected areas Unmanaged land Water	Conservation	0

3.1.3 Land cover change data

The agricultural land cover change (ALCC) data compiled by the Bureau of Rural Sciences were used to provide a measure of land cover change. The ALCC data recorded the loss of forest cover over the period 1990–1995 and the reasons for the change. Data were recorded at a 100 m grid cell resolution.

These land cover change data complement the catchment disturbance measure derived from the land use data. The land use measure reflects the result of land cover change, e.g. forest cover is reduced and grassland increased as a result of clearing. However, the land use data do not distinguish between catchments cleared in 1894 and those cleared in 1994. For the former catchments, one would be expect there to be permanent changes to nutrient and suspended sediment export and runoff, and that these changes would be reflected in the weights applied to different land uses in calculating the land use measure. For land cleared in 1994, the changes listed above would occur, but additionally there would also be impacts as a result of the clearing process (e.g. road development, disturbance to soil surfaces). In other words, the change of catchments from forested to cleared results in both acute and chronic impacts. The land use data provides a measure of the chronic impact. The land cover change data provide a measure of the acute impact.

The ALCC data only provide a measure of the vegetation change that occurred over the period 1990–1995. Although covering only a short period of time, these data can still serve as an approximation of the rates and location of clearing for the recent past. In the more distant past, the acute component of land clearing becomes less significant than the chronic component estimated in the land use measure.

The ALCC data included losses of woody vegetation attributable to a range of causes. Some losses could be considered natural, or are already considered as a consequence of the land use category (e.g. forest harvesting impacts are included in the land use impacts ascribed to forestry). Losses of woody vegetation falling into these categories were not used to calculate the land cover change measure (see Table 19).

Table 19 ALCC categories used to calculate land cover change measure.

ALCC categories: Cause of decrease in woody vegetation, 1990–95	Used for land cover change measure?
On-farm forestry	Not used[1]
Forest management	Not used[1]
Plantation management	Not used[1]
Orchard management	Not used[2]
Agriculture	Used
Abandonment	Used
Grazing	Used
Natural fire	Not used[3]
Managed fire	Not used[3]
Grassland conversion	Used
Development	Used
Other	Used

Note 1: Change in woody vegetation as a result of forestry activities will not be used in this measure because loss of woody vegetation is a normal component of forestry land use, and the on-going impacts of forestry activities are accounted for in the land use measure.

Note 2: Not used for same reason as applied to forestry activities above.

Note 3: Not used as fire is a natural process and its long-term effects are accounted for in the land use measure.

The total area of land cover change in each reach catchment was calculated. A weighting was applied to ensure that the land cover change measure was comparable to the other measures comprising the catchment disturbance index (Equation 9). The weighting was derived by investigating quantitative measures of land clearing impact. Douglas et al*.* (1992) and Miller (1984) found that in the year immediately after forest clearing, sediment yield rates increase by a factor ranging between 7.8 and 18. In subsequent years the sediment yield trends back to the pre-clearing yield rate. Megahan (1987) found that the average sediment yield over a ten-year period following logging was double the yield pre-logging. Taking into account both the immediate major increase and its diminution over time, clearing elevates forest sediment yield to roughly equivalent to that from urban areas (Lawrence 2001). In consequence, the same weighting factor that applies to urban areas in the land use measure (0.68) was applied to areas in which the vegetation was cleared during 1990–1995.

$$LCC = 1 - \frac{Area_d * w}{Area_t}$$ **Equation 9**

where *LCC* = land cover change measure, $Area_d$ = area of catchment in which woody vegetation decreased, $Area_t$ = total area of catchment for which there are data, *w* = weight (0.68).

When calculating the LCC, some catchments did not have data for each cell. Incomplete data were accommodated in the equation above (Equation 9) by dividing by the area of the catchment for which there was data.

3.1.4 Derivation of the weighting scheme for the Infrastructure and Land Use data

Land use activities and infrastructure types can impact the aquatic biota of the river reaches within that catchment in a range of ways. The principal sorts of impacts on aquatic biota were identified to enable us to quantify the impact different activities have on biota. Weights were then derived for each of the land use and infrastructure categories according to their potential to contribute to such impacts (Table 20). The types of impacts judged to be the most important were determined from literature review and professional judgement.

Table 20 Potential impacts that land use and infrastructure activities can have on streams.

Types of impacts	Produced by land use or infrastructure?
Augmentation of the nutrient supply to a stream	Both
Increase in salinity	Land use only
Release of biocides (pesticides, herbicides, fungicides)	Both
Change to the hydrological regime	Both
Augmentation of the sediment supply to a stream	Both
Loss of native riparian vegetation	Land Use only
Toxicants (including hydrocarbons and trace metals)	Both

Each of the categories in the land use and infrastructure data were ranked according to their potential to contribute to the above impacts (Tables 4.21 and 4.22) based on literature information and professional judgement. The ranked scores were averaged across the impact types to produce an overall ranking for each land use and infrastructure category. The weights were derived from the average ranks by scaling them to a range of 0 to 0.7. Ranks were not scaled from 0–1 because a score of 1 (as applied in the land use and infrastructure equations above) implies that the impact cannot get any worse. It was felt that none of the current impacts on streams from types of land use and infrastructure were at their worst, thus the weighting was arbitrarily taken to 0.7. Weightings were checked by comparing them with the outcomes of studies on the impact of catchment land uses on components of the aquatic biota. The potential of each different activity to impact on streams is discussed in detail below.

Table 21 Rankings of different type of impact for different land uses, and resulting land use weights.

	Rankings							Mean rank	Weight
Land Use	**Nutrients**	**Salinity**	**Biocides**	**Hydrological change**	**Sediment supply**	**Riparian change**	**Toxicants**		
Urban	5	2	3	6	3	6	6	4.43	0.68
Intensive and irrigated agriculture	6	6	6	5	4	3	2	4.57	0.70
Dryland cropping	4	4	4	3	3	3	1	3.14	0.48
Grazing	2	3	3	1	2	3	1	2.14	0.33
Forestry – Eastern Australia	1	–	3	2	1	1	1	1.29	0.20
Forestry – Western Australia	1	3	3	2	1	1	1	1.71	0.26
Conservation	–	–	–	–	–	–	–	0	0

Table 22 Infrastructure. Rankings of different type of impact for different infrastructure types, and resulting infrastructure weights.

	Rankings					Mean rank	Weights
Land Use	**Nutrients**	**Agricultural biocides**	**Hydrological change**	**Sediment movement**	**Toxicants**		
Main sealed road	3	1	6	3	6	3.8	0.7
Other sealed road	3	1	6	3	6	3.8	0.7
Railway	1	1	–	1	3	1.2	0.22
Unsealed road	6	–	2	6	1	3.0	0.55
Vehicle track	6	–	2	6	1	3.0	0.55
Utilities (power, pipes)	1	–	–	1	–	0.4	0.07
Walking track	–	–	–	–	–	0	0

3.1.5 Augmentation of the nutrient supply to a stream

Nutrient enrichment of streams can have a range of ecological consequences, including changes to water chemistry and increased periphyton cover, which alters habitat and food resources (Jolly and Chapman 1966). This may favour pollutant-tolerant macroinvertebrates such as worms and midges, and disadvantage sensitive macroinvertebrates such as mayflies and stoneflies (Jolly and Chapman 1966, Gaufin 1973). Nutrient enrichment has been connected to changes in abundance of macroinvertebrates (Cullen and Norris 1982) as well as changes in community composition (Chapman and Simmons 1990).

Nutrient enrichment is often attributable to the input of treated sewage effluent, fertiliser use from agricultural practices, stock, and urban runoff. The ranking system was based on the findings from a number of studies that found that horticultural land uses and orchards exported the greatest total nitrogen (TN) and total phosphorus (TP) per year, followed in turn by urban, tilled agriculture grazing, forestry, and conservation land uses (Gourley et al*.* 1996, Young et al*.* 1996). Nutrients can also be attached to soil particles and so land uses and infrastructure categories that export greater amount of sediment are likely to contribute more nutrients to a stream. For example, unsealed roads and vehicle tracks have the potential to export more sediment to streams than sealed roads, and so these two categories were ranked more highly (Table 22).

3.1.6 Increase in stream salinity

Metzeling et al*.* (1995) reviewed the available information on the effects of salinity on aquatic systems. They reported that some fish species can tolerate salt concentrations of 10 g/L. However, at such levels some reproductive and osmoregulatory abilities may be lost. Certain species of macrophytes have been reported to have high mortality rates when exposed to salinity concentrations of 6 g/L (James and Hart 1993). After reviewing available information on the effects of salinity on macroinvertebrates, Metzeling et al*.* (1995) concluded that even though there is uncertainty about the extent of impact and adaptive capabilities of some fauna, it is likely that the loss of sensitive species will result in a loss of biodiversity. Thus, increases in salinity levels may have widespread effects on aquatic ecosystems.

Salinisation is a growing problem in Australia. Whereas some Australian soils and streams have naturally high salinity levels, the current increasing levels of stream salinity have been attributed to the extensive clearing of deep-rooted vegetation and the increasing use of irrigation (Metzeling et al*.* 1995, NLWRA 2001). In this project, land uses that require irrigation (e.g. cotton and rice) were given the highest ranking in terms of their impact on stream salinisation (Table 21). The remaining land uses were ranked in order of those that have cleared vegetation to the greatest extent (Table 21).

Release of biocides to streams (insecticides, herbicides, and fungicides)

Much work has been undertaken on the effects of biocides on aquatic systems and their biota (Hill et al*.* 1994). These chemicals can affect the survival, growth, reproduction, and food sources of both target and non-target organisms (Brock and Budde 1994). Many biocides have a high toxicity to fish and macroinvertebrates, causing the loss of certain species and changes in abundance and community composition (Siegfried 1993, Hill et al*.* 1994, Leonard et al*.* 2000).

Biocides can enter aquatic environments from agricultural applications, particularly spray-drift and runoff (Brock and Budde 1994). The weighting scheme applied in this project was based largely on the use pattern of biocides for different land uses. Land uses which rely on broad-acre application of biocides were ranked highest (Table 21). For the infrastructure categories, the categories were ranked according to the use of biocides for weed control (Table 22).

3.1.7 Change to the hydrological regime

Change to the hydrological regime caused by land use changes may result in either an increase or decrease in flow as well as changes in the seasonal and daily timing of hydrological events. A number of studies have identified flow as a major factor influencing macroinvertebrate distribution and abundance (e.g. Statzner and Higler 1986, Barmuta 1990, Stark 1993). Low flows may have a substantial influence on macroinvertebrate assemblages through changes in water chemistry, temperature, turbidity, habitat, and food availability (Resh and Rosenberg 1984, Wright et al*.* 1995). These changes can cause a variety of macroinvertebrate community responses, ranging from alterations in abundance to the loss or addition of particular macroinvertebrates (Wright et al*.* 1994).

High flows can influence macroinvertebrate communities by increasing mortality associated with bed movement and washing taxa and algae away (Swanson 1980, McElravy et al*.* 1989). Recolonisation after such events may be erratic, depending on the source of colonisers and the quality of the habitat (Fisher 1990, Wallace 1990).

Seasonal changes to the hydrological regime can adversely affect fish species that spawn at certain times of the year or during particular hydrological events. Fish species such as golden perch that undergo long-distance upstream migration at the onset of rising water levels can be adversely affected by changes to the hydrological regime (Reynolds 1983).

Land use and some types of infrastructure can alter the hydrological regime by changing the rates and quantity of infiltration and overland flow as well as the extraction and release of water. These changes affect how much water flows down the river. The ranking for each of the land uses and infrastructure categories in regard to hydrological change were derived by determining how much the interception characteristics of the land surface have been modified from its natural condition, and how much irrigation a particular land use requires. Land surfaces that are dominated by impermeable surfaces (urban land use and sealed roads) were ranked more highly because of their propensity to markedly change the hydrological regime (Tables 4.21 and 4.22). Land uses requiring irrigation were also ranked highly. Conversely, land uses that change the hydrological regime for only a limited time (e.g. forestry) were ranked lower.

3.1.8 Augmentation of the sediment supply to a stream

The addition of sediment to streams can adversely affect many pollution-sensitive macroinvertebrates by preventing feeding, respiration, or movement, or by filling substrate interstices (Swanson 1980, Lemly 1982, Minshall 1984). Fish species can be adversely affected by sedimentation through changes to turbidity levels, in-filling of pool habitats, and the smothering of food and spawning areas (Metzeling et al*.* 1995, Waters 1995). The effects of sedimentation can alter primary production and cause changes in species diversity and the composition of fish and macroinvertebrate communities (Waters 1995).

Extensive clearing for forestry and agriculture, grazing and cropping, destruction of the riparian zone, urbanisation, road construction, and extractive industries contribute to the increased erosion and sedimentation of streams and rivers (Metzeling et al*.* 1995, Waters 1995, Boulton and Brock 1999). In a comparison of three different land uses (forestry, agricultural, and urban), Lenat and Crawford (1994) found that suspended sediment concentrations under low to average flow conditions were highest at sites surrounded by agricultural catchment, followed in turn by urban and then forestry land uses. Similarly, Lemly (1982) found higher turbidity and suspended load in streams surrounded by logging operations and residential construction. Elevated suspended solids levels were reported downstream of urban development (Hogg and Norris 1991). Based on these and other studies, land uses were evaluated in terms of how they disturbed the land surface and stream banks (Table 21). The final ranking system for sediment supply attempted to incorporate impact from both fine material (changes in turbidity) as well as increases in bedload movement from each of the land use categories used. For example, agricultural areas were ranked higher than urban and forestry areas in export of suspended sediment and augmentation of turbidity levels. Infrastructure categories were ranked in a similar manner (Table 22).

3.1.9 Loss of native riparian vegetation

Riparian vegetation can directly or indirectly influence macroinvertebrates and fish (Plafkin et al*.* 1989, Gregory et al*.* 1991) by providing shading and coarse particulate organic matter (CPOM) essential in certain stream ecosystem food webs (Karr and Schlosser 1978, Gregory et al*.* 1991). The riparian zone also plays an important role in terms of maintaining bank stability and regulating nutrient fluxes (Karr and Schlosser 1978, Gregory et al*.* 1991). Changes to the riparian vegetation can adversely affect macroinvertebrates and fish through habitat change and altered sediment and nutrient fluxes.

Changes to the riparian zone have resulted from clearing to improve stock access to rivers, to produce firewood, and to maximise yields from the land. Invasion of exotic riparian species has changed the nature of the riparian zone along many streams, decreasing its habitat quality. These changes can produce bank erosion, increased nutrient inputs, decreased carbon inputs to the stream, decreased shading, and reduction of in-stream habitat. The rankings ascribed to land uses and infrastructure types were based on the extent to which they result in change to the riparian zone (Table 21). Land uses that lead to the most significant destruction of the natural riparian zone (e.g. urban and grazing land uses) were ranked the highest.

3.1.10 Toxicants (including hydrocarbons and trace metals)

Chemical pollutants (Garie and McIntosh 1986) and trace metals (Weatherly et al*.* 1967, Nicholas and Thomas 1978, Norris 1986) can cause reduced taxa richness and population density in stream ecosystems as well as shifts in community composition.

The rankings for toxicants were based on the propensity of different land surfaces to produce spillages of hydrocarbons and other toxicants. Urban runoff has been found to contain a variety of pollutants including fertilisers, pesticides, and petroleum products, all of which can impact on a large proportion of the aquatic invertebrate community (Garie and McIntosh 1986). As a result of the range and quantity of toxicants produced by industries within urban areas, urban land use was ranked most highly in relation to toxicants (Table 21). The ranking for other land uses was determined accordingly; for example, the ranking of agriculture was based on the intensity of machinery use, and the concomitant hazard from spillage and disposal of toxic chemicals. Of the infrastructure types, sealed roads were ranked highest because of their potential for spillages and release of toxicants from traffic, from which point they are efficiently transported to watercourses (Table 22).

3.2 Overall weighting

3.2.1 Land use

Urban and intensive agricultural land uses were identified as having the greatest potential impact on aquatic biota (Table 21). These rankings are confirmed by studies on the overall effect of land uses on stream biota. Streams surrounded by urban land have been found to have low species richness, low abundance, and an absence of sensitive fish (Lenat and Crawford 1994, Hlass et al*.* 1998) and macroinvertebrates (Pratt et al*.* 1983, Garie and McIntosh 1986, Lenat and Crawford 1994, Hlass et al*.* 1998). Similarly, streams flowing through agricultural land have been reported to have reduced macroinvertebrate species richness and increased numbers of tolerant groups (Lenat and Crawford 1994). Forestry activities within a catchment were found to affect the aquatic biota the least, with higher species richness and diversity of fish and macroinvertebrates at sites surrounded by forestry than by agricultural or urban land uses (Lenat and Crawford 1994).

3.2.2 Infrastructure

The overall weighting system for the infrastructure categories indicated that sealed roads have more impact on the aquatic biota than unsealed roads or other thoroughfares (Table 22). There is little information available directly linking the effects of different types of infrastructure and their relative effects on aquatic biota.

3.3 Calculating the catchment disturbance index from the three measures

The three sub-indices (infrastructure, land use, and land cover change) were integrated into a single catchment disturbance index for each reach. In this process we had the opportunity to weight them if their relative impacts on aquatic biota differed. Although there is evidence linking these factors with changes in aquatic biota, there is little evidence on their relative importance. This is understandable, as with any particular combination of land use or infrastructure within a catchment there may be marked differences in management practices. Consequently, no weighting was applied when the three measures were integrated.

The impacts on streams from infrastructure, land use, and land cover change tend to be cumulative, and the measures were standardised using the ranking process described above so that they reflected similar degrees of impact. To calculate the catchment disturbance index, the impacts of the three measures (i.e. their deviation from pristine) were summed (Equation 10). This equation can give index values less than zero if all three measures show significant impact. In this instance the value of the index was set to zero.

$$CDI = I + LC + LU - 2 \qquad \textbf{Equation 10}$$

where *CDI* = catchment disturbance index, *I* = infrastructure measure, *LC* = land cover change measure, and *LU* = land use measure.

The land use and infrastructure measures were generally present for each reach, but a number of reaches did not have a land cover change measure (through incomplete data coverage). Loss of any measure results in a positive bias to the score; however, in most catchments the land cover change measure is quite small relative to the other two measures. Thus, for the calculation of the catchment disturbance index we required a minimum of the land use and infrastructure measures.

3.3.1 Aggregation of the catchment disturbance index to the basin scale

Catchment disturbance index scores at the reach scale were aggregated to the basin scale by calculating the standardised Euclidean distance measure for all the reach scores. In this process reach scores were weighted by their catchment area, so that reaches with larger catchments (and larger land surface influences) would have a greater influence on the basin catchment disturbance index score than those with small catchments.

4 Hydrological Disturbance Index (HDI): Hydrologic modelling and assessments

4.1 Introduction

Flow regime is one of the key drivers of river condition. Changes to the flow regime can occur for many reasons and affect many aspects of a river's hydrology. Any assessment of river and wetland health needs to include an assessment of hydrological change – a Hydrological Disturbance Index (HDI). The change most typically measured is that which results from river regulation and/or substantial flow diversion or extraction. The ecological effects of these types of changes to flow regimes are well documented. Less is known about the ecological responses to flow regime alterations resulting from land use change, and the HDI need not be designed to assess these changes. This section discusses the options for providing an HDI by reviewing three key previous studies, the National Land and Water Resources Audit, the Sustainable Rivers Audit (SRA), and the flow stress ranking (FSR) procedure recently developed in Victoria by SKM and DSE. The various data used to assess the HDI and sources and reliability of these data are described.

4.2 Background

Changes in river flow regimes are well recognised as a cause of changes to river geomorphology and hence habitat (e.g. Williams and Wolman 1984, Milhous 1982, Erskine et al*.* 1999). These features, in association with flow and water chemistry, control the distribution, dynamics, physiology, and abundance of organisms. Studies have demonstrated the effects of flow on river biota from habitat scales (e.g. Nowell and Jumars 1984, Statzner et al*.* 1988) up to river basin scales (e.g. Maheshwari et al*.* 1995, Power et al*.* 1996, Kingsford 2000), and flow has been considered as the *maestro* that orchestrates patterns and processes in river systems (Walker et al*.* 1995).

Flow variability at various scales is an important determinant of river habitat and biota (e.g. Poff and Ward 1989, Jowett and Duncan 1990, Poff and Allan 1995). Australian rivers, like those of the Murray–Darling system, have some of the most variable flow regimes in the world (Finlayson and McMahon 1988, Puckeridge et al*.* 1998). From a human perspective, they are unreliable water resources and have required extensive flow modification. Large floods that overtop the riverbanks and cover vast tracts of land are a feature of the rivers of the Murray–Darling Basin, as are periodic droughts. These events can result in large costs to rural communities. However, the animals and plants inhabiting these systems are well adapted to the variability (e.g. Boulton 1999, Walker et al*.* 1995). In fact the ecological integrity of these rivers, particularly in the lowland areas, depends upon periodic movements of water onto the floodplain as well as substantial drying-out periods.

The ecological significance of flow variability can be considered at the following levels: *flow regime*, *flow history*, *flow pulse,* and *flow hydraulics* (Figure 11).

Flow regime provides a long-term, statistical generalisation of flow behaviour. It describes variations over hundreds of years, such as changes to the flood and drought cycles driven by long-term climate variations. The natural range of flow levels and their timing is the 'flow regime'. It is also described by measures of central tendency such as the median and the mean flows**.**

Flow history describes the sequence of floods or droughts over recent decades, including the antecedent conditions of flow pulses before any point in time. It can be described by measures of flood and drought magnitude and frequency, measures of the time between floods and droughts, and measures of the seasonality of variously sized floods and droughts.

Flood pulses are single flood events and are generally defined as a rise and fall in discharge (Figure 12). Flood pulses generally extend for less than one year, and are described by measures of magnitude (such flow height, volume, and duration) and measures of the rates of rise and fall.

Flow hydraulics describes the detailed motion of the flow in terms of flow velocity, depth, shear stress, and turbulence.

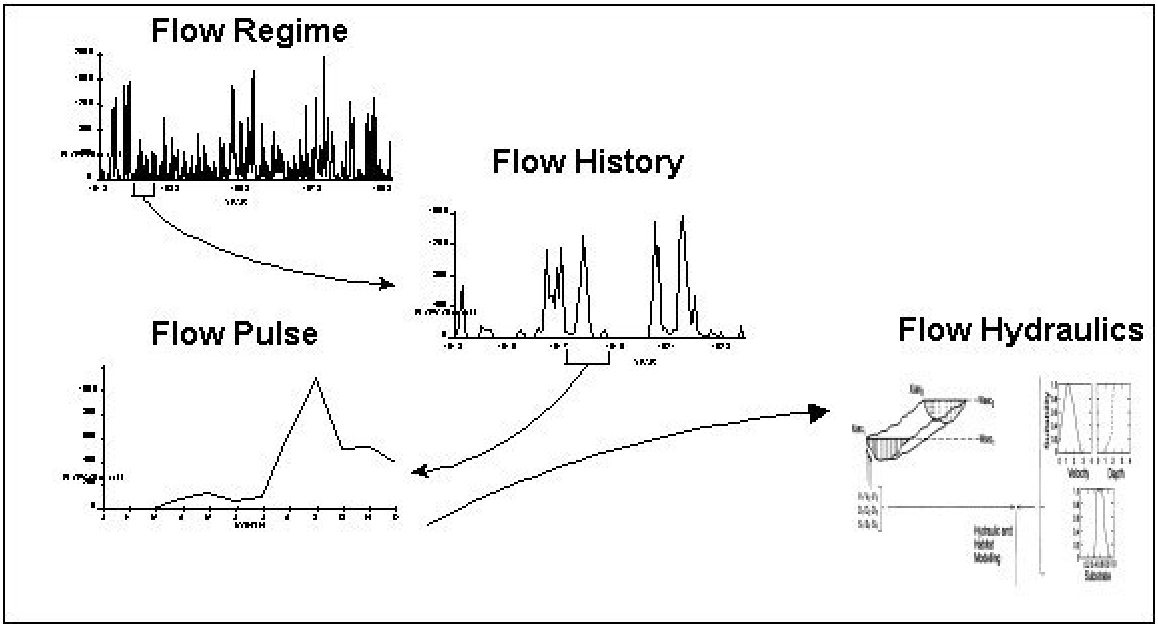

Figure 11 A hierarchical view of aspects of flow.

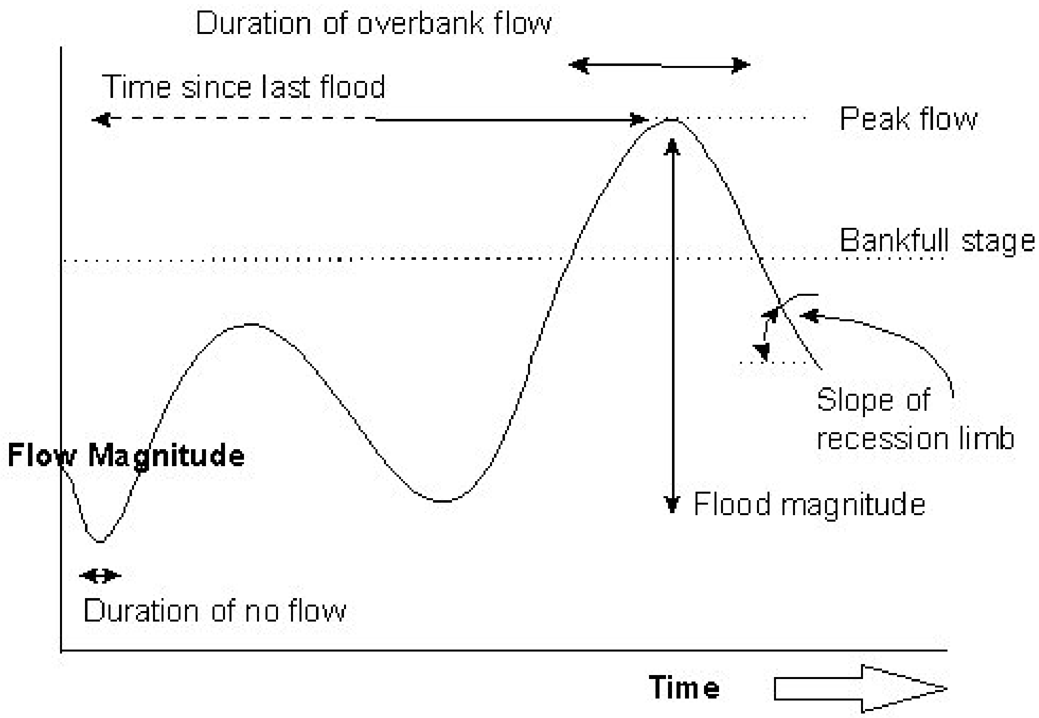

Figure 12 Features of a flood pulse that may have ecological significance.

Hydrological change through water resource or catchment development may influence all or some of these different aspects of flow variability. For example, studies on the Murray, Barwon, and Darling Rivers have shown that water resource development has had a marked but variable impact on hydrology at several time-scales (Thoms and Sheldon 2000). Hydrologic change at both long and short time scales in the Barwon–Darling may prove to have altered the ecological health of the river.

4.2.1 Flow assessment

Measures of flow changes have been proposed to describe flow regime changes and allow comparisons between rivers, between sites on rivers, and between pre- and post-development scenarios for rivers (e.g. Gehrke et al. 1995, Richter et al. 1996, Young 1999, Ladson et al. 1999). These measures range from simple single indices such as the Annual Proportional Flow Deviation index of Gehrke et al. (1995) to the suite of 32 parameters in five groups proposed by Richter et al. (1996). It is well accepted that flow regulation and abstraction can alter many different aspects of the flow regime, and the ecological responses will depend on the full range of these changes. Any single index is unable to capture the full range of possible flow regime changes. While some indices, such as the Annual Proportional Flow Deviation index, are sensitive to a number of different flow changes (for example, seasonal pattern and total volume), they do not distinguish between the types of changes as only a single index value is used. Conversely, the parameters in the comprehensive suite of Richter et al. (1996) are not all independent and cannot be simply integrated into a summary measure.

Most hydrological assessments of temporal change have three steps in common:

1) Define the data series for pre- and post disturbance periods in the river system of interest.

2) Calculate values of each hydrological variable. This is done for each year of the data series – that is, one set for the pre- and post-disturbance series.

3) Compare inter-annual statistics. Measures of central tendency and distribution of the individual parameters are calculated.

This general approach relies upon a reasonable length (at least 30 years) of pre-disturbance flow data. In Australia, such pre-disturbance data series seldom exist, and so assessments must rely on modelled flow data. The main approaches previously used in Australian assessments of flow regime are described below. As part of this review, the data requirements of the assessment (and the ability to obtain such data) are determined and the potential to incorporate them into an overall index is discussed.

The MDBC cap on diversions from surface water (and similar restrictions on surface water use on river systems outside the Murray–Darling Basin), combined with the recent extended period of below-average surface flows, has resulted in an increased interest in the impact of groundwater use on surface water flows. Thus, in considering possible hydrological indices, thought has to be given as to how to incorporate groundwater use and its associated impact on surface water flows.

4.2.2 Stressed Rivers program – New South Wales

The 'Stressed Rivers' assessments used in New South Wales since 1997 classify the sub-catchments of unregulated rivers according to current water use and environmental health, giving a guide to both water management and water policy (Table 23).

The hydrological stress on a sub-catchment is calculated as the estimated proportion of daily flow that has been made available for extraction under existing licenses. This requires estimating stream flow and water use. The index of hydrological stress is calculated for each sub-catchment as the proportion of estimated water extraction to the estimated stream flow. Water use is taken as the peak monthly water extractions lodged by licensed extractors, and stream flow is taken as the 80th percentile for the month of maximum demand. Each sub-catchment is then classified as being of low (0–30 percent extraction of flow), medium (40–60 percent), or high (70–100 percent) hydrologic stress, with additional local information used to place intermediate sub-catchments into one of these categories.

This approach can be applied to rivers that lack major regulating structures, but it relies on accurate and up-to-date extraction data, which can be difficult to obtain. The spatial resolution of extraction data has, to date, limited the use of this approach to the sub-catchment level. The method assumes that hydrological stress is primarily a function of water extraction at low flow and does not provide any indication of changes in flow variability.

Table 23 Classifications of current water use and assessments of environmental health.

	Low environmental stress	Medium environmental stress	High environmental stress
High proportion of water extracted	Category U1 Despite high levels of water extraction the river seems reasonable healthy. However, more detailed evaluation should be undertaken. It is also likely that conflict between users may be occurring during critical periods.	Category S3 Water extraction is likely to be contributing to environmental stress.	Category S1 Water extraction is likely to be contributing to environmental stress.
Medium proportion of water extracted	Category U2 There is no indication of a problem and therefore such rivers would be a low priority for management action.	Category S4 Water extraction may be contributing to environmental stress.	Category S2 Water extraction may be contributing to environmental stress.
Low proportion of water extracted	Category U4 There is no indication of a problem and therefore such rivers would be a low priority for management action.	Category U3 Environmental stress is likely to be caused by factors other than water extraction and as stress is not high these rivers would be a lower priority for management action.	Category S5 While environmental stress is likely to be caused by factors other than water extraction, the high level of environmental stress means it is important to ensure extraction is not exacerbating the problem.

4.2.3 Water Allocation Management Plans (WAMPs) – Queensland

The Water Allocation Management Plans (WAMPs) being developed for catchments across Queensland include a determination of flow changes and how these relate to river condition. The choice of individual flow statistics varies between WAMPs, but as an example the nine flow statistics that were used in the Condamine–Balonne WAMP, and that were assumed to be relevant to the ecological condition of the river, are described (Table 24).

The flow statistics were expressed as proportional changes from natural flow conditions; for example, the *proportion of natural 'high flow' events* is the frequency of high flow events under the modelled development scenario divided by the frequency of high flow events under modelled natural conditions. Ratio values of zero then indicate that high flow events no longer occur, values of one indicate that the frequency of high flow events is unchanged, and values greater than one indicatethat high flow events occur more frequently than under natural conditions. Some of the flow statistics used in the Condamine–Balonne WAMP were defined in qualitative terms, and so must be assessed on the basis of expert judgement.

The Annual Proportional Flow Deviation (APFD) used in the Condamine–Balonne WAMP summarises changes in natural flow regimes associated with water resource development (Gehrke et al*.* 1995) (Equation 11).

$$APFD = \sum_{j=1}^{p} \frac{\sqrt{\sum_{i=1}^{12} \left(\frac{c_{ij} - n_{ij}}{\overline{n}_i} \right)^2}}{p}$$

Equation 11

where: *I* are the calendar months, *p* are the years of record, *c* are the current flow values, and *n* are the natural flow values.

The APFD is scaled so that it is comparable across locations/rivers of differing flow volume, and is sensitive to changes in flow volume occurring in any given month. It is also sensitive to changes in the overall seasonality of flow, and to changes in the shape of the seasonal pattern of flow.

The hydrological assessments used in WAMPs are basin specific, and thus are directly relevant to the river condition issues for a basin. While this has advantages for basin-level management, it does not allow sensible inter-basin comparisons.

Table 24 Flow statistics used in the Condamine–Balonne WAMP.

Key statistic	Primary features of importance
Proportion of natural median annual flow	Annual discharge Sediment transport Availability of aquatic habitat
Annual Proportional Flow Deviation (APFD)	Overall modification of the flow regime Reproduction of native fish and water birds Abundance of alien fish species, e.g. carp
Proportion of natural monthly flow variability	Daily variation in flow, and seasonal patterns of flow variability Natural disturbance
Proportion of natural 'high flow' event frequency	Flooding, and near bank-full flow events Floodplain inundation Natural disturbance Movement of native fish over weirs
Proportion of natural 'medium flow' event frequency	Within-channel flow events Maintenance of channel complexity Inundation of channel benches
Proportion of natural 'low flow' duration	Connectivity of riverine pools Movement of native fish Maintenance of riffle habitat
Proportion of natural 'no flow' duration	Drying of the in-stream environment Natural disturbance Maintenance of in-stream vegetation Oxidation of nutrients
Proportion of river inundated by dams and weirs	Loss of natural riverine habitat

4.3 The National Land and Water Resources Audit Hydrological Disturbance Index

The conceptual framework of the NLWRA's Assessment of River Condition (ARC) included an assessment of hydrology as a component of habitat. The ARC was defined in terms of five indices, one of which was a hydrologic index that assessed the extent of anthropogenic change in river hydrology. This assessment was made for river sections in AWRC basins where river hydrology has been altered by flow regulation and/or flow diversion or augmentation. River hydrology may have been substantially altered in other basins because of land use change and the construction of on-farm water storages; however, these affects were not assessed in the ARC because of a lack of data to describe these types of changes.

4.4 The Hydrology Index (HI)

The Hydrology Index (HI) is defined in terms of four sub-indices:

- the Mean Annual Flow Index (*A*),
- the Flow Duration Curve Difference Index (*M*),
- the Seasonal Amplitude Index (*SA*) and
- the Seasonal Period (*SP*).

The HI is defined as the Euclidean distance between an unimpacted hydrology condition and the condition defined by four sub-indices in a four-dimensional space (Equation 12). The hydrology sub-indices are all defined in the range 0–1, where 1 represents unimpacted and 0 represents maximum impact.

$$HI = 1 - \frac{\sqrt{(1-A)^2 + (1-M)^2 + (1-SA)^2 + (1-SP)^2}}{\sqrt{4}}$$ **Equation 12**

4.4.1 The Mean Annual Flow Index (*A*)

The Mean Annual Flow Index (*A*) is defined in Equation 13, where the mean annual flow under current and natural conditions is given by Q_{MAc} and QMA_n respectively. This provides a measure of the difference in total flow volume between current and natural conditions.

if $Q_{MAc} > Q_{MAn}$ then $A = Q_{MAn}/Q_{Mac}$, else $A = Q_{MAc}/Q_{MAn}$ **Equation 13**

This statistic assumes that increases and reductions in mean annual flow have equivalent impacts on habitat condition.

4.4.2 The Flow Duration Curve Difference Index (*M*)

The Flow Duration Curve Difference Index (*M*) is the difference from 1 of the proportional flow deviation, averaged over *p* monthly flow percentile points (Equation 14), where Q_{in} is the natural monthly flow value for percentile point *i,* and Q_{ic} is the current monthly flow value for percentile point *i*. The statistic *M* gives equal weighting to each percentile flow, from the lowest flow to the highest flow, and provides a measure of the overall difference between current and natural flow duration curves. This index was based on a measure developed by Young et al*.* (2000) to assess the overall hydrological deviation of a dam release option from the transparent dam flow.

If $Q_{in} > Q_{ic}$ then $M = \frac{1}{p}\sum_{i=1}^{p}\frac{Q_{ic}}{Q_{in}}$ else $M = \frac{1}{p}\sum_{i=1}^{p}\frac{Q_{in}}{Q_{ic}}$ **Equation 14**

4.4.3 The Seasonal Amplitude Index (*SA*)

The Seasonal Amplitude Index (*SA*) assesses the change in amplitude of the seasonal pattern of monthly flows. It is defined as the average of two current:natural ratios, first, that of the highest monthly flows (Q_{HMc}:Q_{HMn}), and second, that of the lowest monthly flows (Q_{LMc}:Q_{LMn}) (Equation 15), based on calendar month means. Each of these ratios is defined as being less than or equal to 1, so in each case the denominator is the larger of current r, the natural value.

$$SA = \frac{Q_{HM_c} / Q_{HM_n} + Q_{LM_c} / Q_{LH_n}}{2}$$ **Equation 15**

The Seasonal Period Index (*SP*) assesses the change in seasonal timing of flows. It can be defined as the difference from 1 of one-twelfth of the sum of the absolute values of the differences between current and natural of, first, the numerical values of the months with the highest mean monthly flows (*HM*) and, second, the numerical values of the months with the lowest mean monthly flows (*LM*) (Equation 16). The 12 numerical values of the calendar months used in Equation 16 are assigned in sequence as: 1, 2, 3, 4, 5, 6, 7, 6, 5, 4, 3, 2, with HM_n always set equal to 1 and the value of HM_c determined relative to HM_n; and similarly, LM_n set equal to 1, and LM_c determined relative to LM_n. For example, if HM_n is June and HM_c is August, then HM_n=1 and HM_c=3. Note the actual algorithm to calculate *SP* uses a slightly different logic to that described above, but with the same result.

$$SP = 1 - \frac{|HM_c - HM_n| + |LM_c - LM_n|}{12}$$ **Equation 16**

4.4.4 Data requirements

The data used to assess the NLWRA–HI can be classified as follows:

- Modelled natural:simulated stream flow record, where the modelling is based on a no-development scenario. This implies no extractions or regulating structures such as dams, and sometimes includes simulation of runoff from natural land cover.
- Modelled current:
  - regulated streams:simulated stream flow record, where the modelling is based on current levels of development (extractions and diversions, etc.)
  - unregulated streams:extended stream flow record
- Observations:stream flow (gauging) record.

The sub-index *A* only requires estimates of annual flow volumes. The sub-indices *SP*, *SA*, and *M* require estimates of monthly flow volumes. Consequently, the minimum time-step required for the modelled data is monthly. A continuous data series of at least 30 years was deemed necessary to assess temporal changes.

The components of the hydrology index rely on paired datasets representing current and reference (natural or unregulated) conditions. Ideally, the datasets would have been simulated (modelled) natural and current; however, this was not always possible and any one of the following pairs of data were used:

- modelled natural and modelled current
- modelled natural and observations
- observations pre- and post-regulation

The primary data sources were:

- extended synthetic streamflow records for unregulated streams across the RBCIA (NLWRA Theme 1, Peel et al. 2000)
- modelled natural and matching observations for a selection of sites within each of the AWRC basins (NLWRA Theme 1).

Additional to these data, modelled and observation data were supplied from government agencies responsible for hydrology data (Table 25).

Table 25 Data supplied for the calculation of the hydrology index

State/ Territory	Data	Source
ACT	Daily flows from all gauging stations	Environment ACT
NSW	Daily flows from gauging stations except for Murray River	Pineena – NSW DLWC
	Daily and monthly modelled natural flows Daily and monthly modelled current flows	NSW DLWC SMHEA for some sites in Upper Murrumbidgee
MDBC	Daily and monthly modelled natural flows Daily and monthly modelled current flows Daily observations	MDBC
VIC	Daily flows from all regulated Victorian gauging stations	Victorian Water Resources Data Warehouse
	Monthly modelled natural flows Monthly modelled current flows	Vic NRE
TAS	Daily flows from all regulated gauging stations	Dept Primary Industries, Water and Environment
	Monthly modelled natural flows Monthly current flows	HEC

SA	Daily flows from all gauging stations Monthly modelled natural flows Monthly modelled current flows	SA Dept Environment, Heritage and Aboriginal Affairs
WA	Daily flows from all gauging stations	WA Water and Rivers Commission
NT	Daily flows from all gauging stations	Dept Land Planning and Environment
QLD	Daily flows from all gauging stations Daily and Monthly modelled natural flows Daily and Monthly calibration flows (extended gauging record)	Dept Natural Resources

The modelled data provided by government agencies were generated using the following simulation models:

- *I*ntegrated *Q*uantity *Q*uality *M*odel (IQQM; New South Wales and Queensland)
- *R*esource *A*llocation *M*odel (REALM, Victoria)
- *M*urray *S*imulation *M*odel (MSM) and BigMod (Murray–Darling Basin)
- WaterCress (South Australia)
- simple site-specific rainfall–streamflow models (Tasmania).

No results from hydrology simulation models were available for the Northern Territory or Western Australia.

The definition of current or developed conditions differs across the country. As a result of the Murray–Darling Basin Ministerial Council Cap on diversions, water resource development within the Murray–Darling Basin is to be held at 1993/94 levels. This generally defined current condition within the Murray–Darling Basin as the 1993/94 levels of development, with the following exceptions:

- Lachlan River at 1997/98 development levels
- Murray River at 1999/2000 development levels.

In South Australia, the North Para River and Jacob Creek are modelled at the 1997 development level.

4.4.5 Comments

The data required to calculate the components of the Hydrology Index for each reach in the Intensive Landuse Zone were immense and in many situations the data pairs suitable for calculating the hydrological index were not available. Under these circumstances a set of principles to extrapolate from known points and interpolate between known points were developed.

4.5 The Sustainable Rivers Audit (SRA) Approach

Following the NLWRA, the Sustainable Rivers Audit was developed to benchmark river health across the Murray–Darling Basin and to provide information to guide the long-term management of riverine resources in the Basin. It builds upon the concepts of the NLWRA assessment of river condition, expanding them to include a more comprehensive assessment of river health.

The hydrology of a river was seen as an important theme within the SRA because it refers to the distribution of water over time and space and encompasses flow volumes, rates, variability, and seasonality. As such it is closely linked to the distribution, physiology, and abundance of aquatic flora and fauna. Additionally, if there is an absence of long-term biological information, the long-term changes in a river's hydrology can provide a surrogate indication of the 'stress' on biota and their habitat.

4.5.1 The indices

The four hydrology sub-indices used in the NLWRA were initially proposed for use in the SRA Hydrology Index (SRA–HI) (Whittington et al. 2001). During the course of the pilot SRA, the potential of more than 30 hydrology indicators and their variants was assessed. These included a series of 'variance-corrected' versions developed by SKM to allow for differences in flow variability between streams. At the conclusion of a rigorous assessment and discussion process, 12 hydrology sub-indices were recommended for use in the SRA–HI (Table 26).

Table 26 Indicators and sub-indices to be included in the SRA–HI

Sub-Index	Indicators
Flow volume	Median Annual Flow Mean Annual Flow Amended APFD
Seasonality	Seasonal Period Index (frequency distribution)
Variability	Seasonal Amplitude Yearly Variation
Low and zero flow	Low Flow Event Number Low Flow Event Duration Zero Flow Days Difference
High flow	1:2 year ARI Flood Event Number 1:5 year ARI Flood Event Number 1:10 year ARI Flood Event Number

4.5.2 The Mean and Median Annual Flow Indices

The Mean Annual Flow indicators provide a measure of the difference in mean annual flow volume between current and natural conditions (Equation 17).

if $Q_c > Q_n$ then $A = Q_n/Q_c$ else $A = Q_c/Q_n$ **Equation 17**

where: Q_n = mean/median annual flow under natural conditions, and Q_c = mean/median annual flow under current conditions.

4.5.3 Amended APFD

The Annual Proportion of Flow Deviation indicator (APFD) was first proposed by Gehrke et al*.* (1995). It was subsequently amended to be more suitable in streams where monthly flows under natural conditions are often zero (AAPFD) and has been used in various studies such as the Victorian Index of Stream Condition (ISC). This indicator has been found to be related to the diversity of fish species in regulated rivers (Gehrke et al*.* 1995) and is defined as the sum of the ratio of change in monthly flow (current to natural) to average monthly flow (Equation 18) and the formula used in the Pilot SRA was:

$$AAPFD = \sum_{j=1}^{p} \frac{\sqrt{\sum_{i=1}^{12} \left[\frac{c_{ij} - n_{ij}}{\overline{n_i}} \right]^2}}{p}$$ **Equation 18**

where: p = number of years in the simulation period, c_{ij} = modelled existing flow for month i in year j, n_{ij} = modelled natural flow for month i in year j, n_i = mean natural flow for month i across p years (adapted from Gehrke et al*.* 1995).

The AAPFD scores range from 0 in an unregulated river to 3.46 where there is a 100 percent increase or decrease in flow and it is also responsive to seasonal changes. To compress the scores to between 0 and 1 and to make a score of 1 be similar to natural, the AAPFD scores were corrected using the following rating table (adapted from the Victorian Index of Stream Condition, ISC) (Table 27). In this rating table, intermediate AAPFD values were interpolated between the two corresponding ratings. For example, an AAPFD of 0.15 would have a rating of 0.85.

Table 27 Rating table used to correct the AAPFD scores.

AAPFD Rating	Rating
< 0.1	1
0.1	0.9
0.2	0.8
0.3	0.7
0.5	0.6
1.0	0.5
1.5	0.4
2	0.3
3	0.2
4	0.1
> 5	0

4.5.4 Seasonal Period Index (frequency distribution)

This indicator compares the frequency that peak flows occur for each month of the year under both current and natural conditions. It then compares the frequency that low flows occur for each month of the year under both current and natural conditions, and then averages these two values (Equation 19).

$$SP\_fd = \left\{ \left[\frac{1}{M} \sum_{i=1} MIN(YHC_i; YHN_i) \right] + \left[\frac{1}{M} \sum_{i=1} MIN(YLC_i; YHLN_i) \right] \frac{1}{2} \right\}$$

Equation 19

where: YHC = number of years the *i*th month has the peak annual flow under current conditions

YHN = number of years the *i*th month has the peak annual flow under natural conditions

YLC = number of years the *i*th month has the minimum annual flow under current conditions

YLN = number of years the *i*th month has the minimum annual flow under natural conditions

MIN = minimum

M = number of months in flow data set.

4.5.5 The Seasonal Amplitude Index (*SA*)

The Seasonal Amplitude Index (*SA*) assesses the change in amplitude of the seasonal pattern of monthly flows. It is defined as the average of two current:natural ratios, first, that of the highest monthly flows (Q_{HMc}:Q_{HMn}), and second, that of the lowest monthly flows (Q_{LMc}:Q_{LMn}) (Equation 20), based on calendar month means. Each of these ratios is defined as being less than or equal to 1, so in each case the denominator is the larger of current *r*, the natural value.

$$SA = \frac{Q_{HM_c} / Q_{HM_n} + Q_{LM_c} / Q_{LH_n}}{2}$$ **Equation 20**

4.5.6 Yearly Variation Indicator

The Yearly Variation Indicator used in the SRA was developed by SKM (2003) and is the ratio of the coefficient of variation of annual flows under reference and current conditions where the coefficient of variation is defined as the standard deviation divided by the mean (Equation 21). The Yearly Variation indicator (*AV*) is calculated as:

$$AV = \frac{AVC_n}{AVC_c}$$ **Equation 21**

where:

ACV_c = Current annual coefficient of variation

ACV_n = Natural annual coefficient of variation.

4.5.7 Low Flow Event Number

For the SRA it was agreed that it was inadequate to use the 90th percentile as a threshold for determining changes in low flows. This was because there are significant rivers and streams where the 90th percentile flow is zero under both current and natural conditions, and as such these sites would score highly even though there could have been changes to low flows. Therefore, it was decided to carry out the low flow spell analysis on the 90th percentile of non-zero flows.

The Low Flow Event Number assesses differences in the number of low flow events (*LFEN*) between current and natural conditions (Equation 22).

$$LFEN = \frac{\min(N_n, N_c)}{\max(N_n, N_c)}$$ **Equation 22**

where:

N_n = number of event exceedences < the natural 90th percentile of non-zero flows under natural conditions

N_c = number of event exceedences < the natural 90th percentile of non-zero flows under current conditions

An event is defined as independent if it was separated by 5 days or more of higher flows.

4.5.8 Low Flow Event Duration

The second low flow indicator was developed by the SRA to assess the differences in the mean duration of low flow events (LFED) between current and natural conditions (Equation 23).

$$LFED = \frac{\min(N_n, N_c)}{\max(N_n, N_c)}$$ **Equation 23**

where:

Nn = mean duration of event exceedences < the natural 90th percentile of non-zero flows under natural conditions

Nc = mean duration of event exceedences < the natural 90th percentile of non-zero lows under current conditions.

An event is defined as independent if it was separated by 5 days or more of higher flows.

With both of these indicators, an increase or a decrease in low flows under current conditions can result in a low score.

Zero Flow Days Difference

Zero Flow Days Difference is a measure of the difference of the proportion of zero flow days between the current and natural conditions (Equation 24).

$Z_d = 1 - [\text{ABS}\ (Z_c/\text{days} - Z_n/\text{days})]$ **Equation 24**

where:

Z_c = number of zero flow days under current conditions

Z_n = number of zero flow days under natural conditions

days = total number of days in record.

This formula only returns a zero score if there has been a complete change from all days having zero flow under natural conditions (no water) to where there are now no zero flow days under current conditions (permanent flow), or if there were no zero flow days (permanent flow) under natural conditions and now all the days have zero flow (no water). Also, if there is only a small change in the number of zero flow days, but a large change in the proportion of zero flow days (e.g. two zero flow days under current conditions and four zero flow days under natural conditions), then a high score is still returned.

4.5.9 1:2, 1:5, 1:10 year ARI Flood Event Number

For high flows the SRA developed separate spell analysis indicators that assessed the impacts on floods with Annual Return Intervals (ARIs) under natural conditions of 10 years, 5 years, and 2 years. The natural flow levels required to achieve these ARIs were determined by fitting a second-order polynomial to the log-transformed flow peaks (an exponential distribution was assumed) of a partial flow data series in which the number of flow peaks was set equal to the number of years of record.

The high flow event number indicator (HFEN) was determined for each of the 1:2, 1:5, and 1:10 ARIs and was calculated as the ratio of the number of event exceedances above these flow thresholds under both natural and current conditions (Equation 25).

$$HFEN = \frac{\min(N_n, N_c)}{\max(N_n, N_c)} \qquad \textbf{Equation 25}$$

where:

N_n = number of event exceedences (1:2, 1:5, 1:10 natural ARIs) under natural conditions

N_c = number of event exceedences (1:2, 1:5, 1:10 natural ARIs) under current conditions

An event is defined as independent if it is separated by 5 days or more of lower flows.

4.5.10 Combining the indices

The set of 12 indices described above were combined into an overall index using fuzzy logic. This is a process whereby a decision surface is created through a set of expert rules, which provides a single score that represents the 'expert' interpretation of the values of all the indicators. The implementation of such an aggregation approach requires the expert rules to be clearly documented – and, as such, they become transparent and repeatable. The advantage of this approach is that the expert rules may be changed over time as our understanding of river health improves.

The expert system developed for the SRA–HI involved the following steps:

1) The indicators were placed into five sub-indices depending on the type of hydrology information they provided

2) For each sub-index, the indicators were ranked and rated in a *definition table* according to the ecological significance and accuracy of their information content (some were ranked equally)

3) The five sub-indices were then subjectively ranked and rated in a *definition table* according to the ecological significance and accuracy of their information content (the ranking varied depending on which VPZ the site was located in)

4) The expert systems were developed to provide an index between 0 and 1, with 1 representing no change from natural.

To aggregate the site data to the catchment scale, an estimate of the river length represented by each site was used to weight the site scores. Thus, sites representing greater stream lengths contributed more to the SRA Hydrology Index.

4.5.11 Data requirements

The SRA–HI is based on long-term modelled data and involves a comparison between 'current' and 'natural' conditions. Thus they *require* paired data sets representing current and reference conditions. The calculation of the low flow and zero flow events sub-index and the high flows sub-index requires data at a daily time step – and thus the models need to supply daily data.

As this index relies on water resource models, it cannot be undertaken in un-modelled areas (generally upland and unregulated streams).

4.5.12 Number of years of modelled data

There are a number of data analysis issues caused by differences between the current state-based water resource models, both in their modelling software and model input. In particular, the length of modelled hydrologic data varies between catchments. This difference has the potential to introduce a bias when the historic record is used to deduce natural conditions (e.g. the pre–late 1940s were dry compared to the post–late 1940s). As part of the pilot SRA, a sensitivity analysis was performed on selected indicators using full and truncated datasets. The most susceptible indicators were the Seasonal Period, the 1:5 and 1:10 Year ARI high flow indicators, and the High Flow Maximum Interval indicator. This information was used when selecting suitable indicators for the SRA Hydrology Index. However, no minimum record length was recommended.

4.6 Index of Stream Condition and Flow Stress Ranking

The Victorian Flow Stress Ranking (FSR) project (SKM 2005) built on the earlier hydrological assessment work of the National Land and Water Resources Audit (NLWRA) and the Sustainable Rivers Audit (SRA). Key to developing the component indices of the FSR was the recognition that while indices based on daily data were highly desirable, the bulk of the available hydrologic models (required to provide the time-series flows of current and unimpacted) operated at a monthly time-step. As such, the goal of the project was to develop indices that were based on monthly data but provided adequate representation of the information provided by indices calculated from daily data. The component indices of the FSR were adopted for the Victorian Index of Stream Condition (ISC) assessments and were subsequently used for the 2004 round of assessments.

[NOTE: the terminology used in the FSR reporting differs slightly to that used in the NLWRA and the SRA. Unimpacted is equivalent to the natural or reference condition used in the other projects.]

This goal was achieved by finding 50 sites across Victoria representing the full range of climate, topography, and stream regulation. Ten hydrologic indices, selected on the basis of scientific review and expert input from both hydrologists and ecologists, were calculated using daily and monthly data for each site. Multiple indices representing the same hydrologic characteristic were calculated; for example, for low flows three indices were calculated: low flow magnitude, a low flows spells, and a proportion of zero flow.

Several of the indices were highly correlated and thus a smaller set could be used with minimal loss of information. The smaller set was selected by correlating the indices calculated from monthly data with those calculated from daily data. This analysis indicated that 5 of the indices calculated from monthly data, either alone or in combination, adequately represented the indices calculated from daily data. No significant correlation was found between these indices.

The 5 monthly indices that were selected were:

1) Low Flow index
2) High flow index
3) Proportion of zero flow index
4) Monthly variation index
5) Seasonal period index.

These indices are strongly related to 4 of the 5 component indices of the SRA–HI (Table 28). Thus they provide a practical alternative to the SRA indices with the advantage of being calculated with monthly data. The SRA–HI component index which is not directly represented by the FSR indices is the changes in flow volume, however, the use of duration curves for the FSR indices means that this information is encapsulated in the set of indices recommended.

Table 28 Relationship between the FSR indices and the SRA–HI components

Monthly Indices	SRA–HI components
Low Flow index (*LF*)	Low and zero flows
Proportion of zero flow index (*PZ*)	Low and zero flows
High flow index (*HF*)	High flows
Monthly variation index (*CV*)	Variability
Seasonal period index (*SP*)	Seasonality

4.6.1 Structure of the monthly indices

During the development of the SRA–HI components a way of incorporating variability was considered, but the options presented were still very much in the early stages of their conceptual development and were not used. The development of the FSR indices now accounts for the variability of the system, both impacted and current. Using the actual distribution of parameters over the period of the data, rather than fitting a statistical distribution to the data (i.e. a non-parametric approach), achieved this. Consequently, the indices are generally based on the change in percent of time that a given value is exceeded between unimpacted and current conditions. An example of this can be seen in Figure 13.

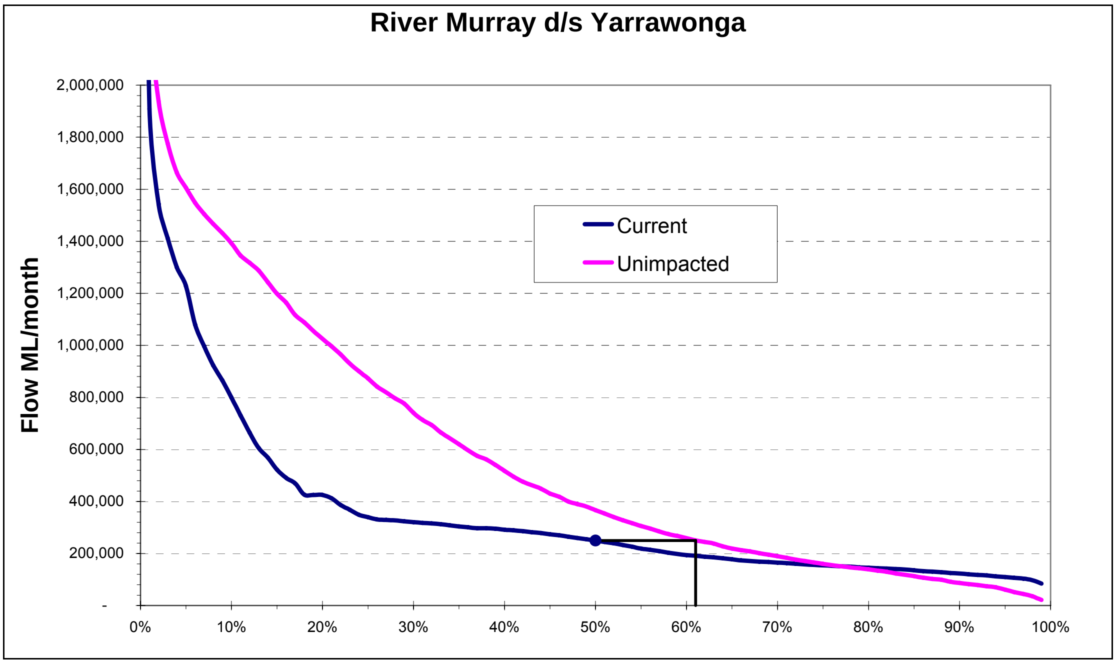

Figure 13 Calculating the percentage change for a given value. In this example the current median (50 percent) monthly flow is equivalent to the 61 percent flow of the unimpacted flow regime – hence an 11 percent change.

The five indices and their method of calculation are presented below (from Appendix C of SKM, 2005).

4.6.2 Low Flow index (*LF*)

The Low Flow index is a measure of the change in low flow magnitude under current and unimpacted conditions. These are calculated based on the 91.7 percent exceedance flow (11 months out of 12) and the 83.3 percent exceedance flow (10 months out of 12).

The index is calculated using the difference between the percentage of years that the unimpacted and current 91.7 percent exceedance flows (evaluated over the whole period of record) are exceeded by the annual 91.7 percentile flow (evaluated on a year-by-year basis) under unimpacted conditions. The same process is then followed for the 83.3 percent exceedance flows (Equation 26).

$$LF_{91.7} = 1 - 2x\left|P_{ile}\left(Q91.7_u\right) - P_{ile}\left(Q91.7_c\right)\right|$$ **Equation 26**

where:

$LF_{91.7}$ = Range-standardised low flow index based on the 91.7 percent exceedance flow

$Q91.7_c$ = Current 91.7 percent exceedance flow (ML)

$Q91.7_u$ = Unimpacted 91.7 percent exceedance flow (ML)

$P_{ile}(Q91.7_c)$ = Proportion of years that the current 91.7th percentile flow is exceeded by the annual 91.7th percentile unimpacted flow

$P_{ile}(Q91.7_u)$ = Proportion of years that the unimpacted 91.7th percentile flow is exceeded by the annual 91.7th percentile unimpacted flow (which by definition is 91.7).

The low flow index is calculated as the average of the variance-corrected low flow index based on the 91.7 percent exceedance flow and the variance-corrected low flow index based on the 83.3 percent exceedance flow (Equation 27).

$$LF = \frac{LF_{91.7} + LF_{93.3}}{2}$$ **Equation 27**

where:

LF = Range-standardised low flow index

$LF_{91.7}$ = Range-standardised low flow index based on the 91.7 percent exceedance flow

$LF_{83.3}$ = Range-standardised low flow index based on the 83.3 percent exceedance flow.

4.6.3 High Flow index (*HF*)

The High Flow index is a measure of the change in high flow magnitude from unimpacted to current conditions. The approach adopted to calculate the high flow index is similar to that used to calculate the low flow index. The monthly high flow index is calculated based on the 8.3 percent and 16.7 percent exceedance flows (1 and 2 months in 12 respectively).

The index is calculated using the difference between the percentage of years that the unimpacted and current 8.3 percent exceedance flows (evaluated over the whole period of record) are exceeded by the annual 8.3 percentile flow (evaluated on a year-by-year basis) under unimpacted conditions. The same process is then followed for the 16.7 percent exceedance flows (Equation 28).

$$HF_{8.3} = 1 - 2x\left|P_{ile}\left(Q8.3_u\right) - P_{ile}\left(Q8.3_c\right)\right|$$ **Equation 28**

where:

$HF_{8.3}$ = Range-standardised low flow index based on the 8.3 percent exceedance flow

$Q8.\ 3_c$ = Current 8.3 percent exceedance flow (ML)

$Q8.3_n$ = Unimpacted 8.3 percent exceedance flow (ML)

$P_{ile}(Q8.3_c)$ = Proportion of years that the current 8.3 percentile flow is exceeded by the annual 8.3 percentile unimpacted flow

$P_{ile}(Q8.3_u)$ = Proportion of years that the unimpacted 8.3 percentile flow is exceeded by the annual 8.3 percentile unimpacted flow

The high flow index is calculated as the average of the variance-corrected high flow index based on the 8.3 percent exceedance flow and the variance corrected high flow index based on the 16.7 percent exceedance flow (Equation 29).

$$HF = \frac{HF_{8.3} + HF_{16.7}}{2}$$ **Equation 29**

where:

HF = Range-standardised high flow index

$HF_{8.3}$ = Range-standardised high flow index based on the 8.3 percent exceedance flow

$HF_{16.7}$ = Range-standardised high flow index based on the 16.7 percent exceedance flow.

4.6.4 Proportion of zero flow index (*PZ*)

The proportion of zero flow index compares the proportion of zero flow occurring under unimpacted and current conditions (Equation 30). The value of the index varies from zero to one, and similarly to other indices, the direction of change is not evident from the value of the index. If the number of cease to flow spells is unchanged between unimpacted and current conditions, then the value of the index is 1.

$$PZD = 1 - 2 \times [\max(PZ_u, PZ_c) - \min(PZ_u, PZ_c)]$$ **Equation 30**

where:

PZ = Proportion of zero flow index

PZ_u = Proportion of zero flow over the whole record under unimpacted conditions

PZ_c = Proportion of zero flow over the whole record under unimpacted conditions.

This index can be determined from either a daily or monthly streamflow record. This index is conceptually identical to that adopted in the SRA, but here zero flow is defined as the non-zero flow exceeded 99.5 percent of the time. This definition is a scale-independent means of defining zero flows that is less sensitive to the accuracy of gauging at low flows. The adopted index also differs from that adopted in the SRA as it is based on double the difference between unimpacted and current conditions. This factoring was introduced simply to ensure that the most impacted sites in Victoria (here represented by the downstream reaches of the Goulburn and Loddon Rivers) have suitably low scores.

4.6.5 Monthly variation index (*CV*)

The variation index compares the coefficient of variation of monthly flows between current and unimpacted conditions. This index is the same as that used in the SRA. The index is calculated as the ratio of the monthly flows under unimpacted and current conditions, where the coefficient of variation is defined as the standard deviation divided by the mean (Equation 31).

$$CV = \frac{CV_u}{CV_c}$$ **Equation 31**

where:

CV = Index of monthly variability

CV_c = Current monthly coefficient of variation

CV_u = Unimpacted monthly coefficient of variation.

4.6.6 Seasonal period index (*SP*)

This index compares the unimpacted and current frequency distribution of maximum and minimum monthly flows. The first step in calculating the index is to create frequency distributions that show the percentage of years that peak and minimum annual flows fall within each given month under current and unimpacted conditions. The index is then calculated by summing the minimum proportions (from unimpacted or current) within each month. In MDBC (2003) the index is presented in terms of the number of years the peak or minimum flow falls within each given month. In this report the percentage of years the peak or minimum flow falls within each given month has been used (Equation 32).

$$SP\_fd = \frac{1}{200}\left\{\sum_{i=1}\left[MIN\left(PHC_i ; PHU_i\right)\right] + \sum_{i=1}\left[MIN\left(PLC_i ; PLU_i\right)\right]\right\}$$ **Equation 32**

where:

SP_fd = Comparison of frequency distribution seasonal period index

PHC_i = The percentage of years the *i*th month has the peak annual flow under current conditions (percent).

PHU_i = The percentage of years the *i*th month has the peak annual flow under unimpacted conditions (percent).

PLC_i = The percentage of years the *i*th month has the minimum annual flow under current conditions (percent).

PLR_i = The percentage of years the *i*th month has the minimum annual flow under reference conditions (percent).

4.6.7 Combining the indices

In developing a single score for each site, options for weighting the various parameters were considered and tested. The conclusion of this assessment was that there was no defensible method of weighting the various parameters so as to be valid across the range of catchments being considered in Victoria – with the exception of the seasonality index, which is given a weighting of 2. The Raw Seasonally Weighted Score (RSWS), or for the purposes of this discussion the FSR – Hydrology Index (FSR–HI), is calculated as follows (Equation 33):

FSR–HI (*RSWS*) = (*LF*+*HF*+*CV*+*PZ*+2**SP*)/6 **Equation 33**

This score ranges between 0 (totally changed) and 1 (pristine).

4.6.8 Data requirements

The components of the FSR–HI are calculated from paired data sets representing current and unimpacted (natural or unregulated) conditions. Considerable effort was expended in ensuring that the indices were able to be calculated using monthly flow data. Thus it will be possible to use the following pairs of data to calculate the indices:

- modelled natural and modelled current
- modelled natural and observations
- observations pre and post regulation.

A minimum of 15 years of monthly data is required to derive reliable estimates of the component indices; however, the greater the period of record, the greater the confidence in the indices.

4.7 Groundwater indices

The MDBC cap on diversions from surface water, and similar restrictions on surface water use on river systems outside the Murray–Darling Basin, combined with the recent extended period of below-average surface flows, has resulted in an increasing interest in the use of groundwater. This in turn has led to concerns that increased groundwater diversions may be impacting surface water flows. This causes a particular problem if restrictions on surface water use for the purposes of providing a minimum environmental flow result in increased groundwater diversions and a subsequent reduction in surface water flows.

As a result of these concerns, there is a very real need to include the impact of groundwater extraction and use on river health, and its incorporation into a river health assessment needs to be considered. Logically, any assessment of groundwater extraction falls within the realm of the hydrology sub-index and the following provides a discussion about its inclusion.

The impact of groundwater extraction on surface flows will be automatically included in current models of surface water flows – i.e. the losses calibrated into the models will be higher (or baseflow inflows will be lower) to account for groundwater extractions. However, groundwater contributions are not a direct model parameter that can be adjusted. As a result, the ‘natural’ flow models will be inaccurate because they simply set all extractions and dam capacities to zero. However, a key question to ask is: will the impact of changed groundwater use on the model outputs be significant compared with the impacts of catchment land use changes? (e.g. land clearing, which has increased the rate of groundwater recharge and volume of surface runoff).

4.7.1 Groundwater use

Groundwater use and the potential impact on surface water at the reach and catchment scale is a complex issue and one that is in its early stages among natural resource managers and scientists alike. The data on consumption is limited – typically only available for commercial use in defined groundwater management areas. The metering of groundwater use is incomplete at present and datasets will be of short duration. Licensing authorities have typically had little control over extraction volumes and have tended to estimate usage on area and crop types (e.g. licence to irrigate 100 ha of grass). Such systems do not provide an accurate measure of actual use as soil type, irrigation efficiency, etc., is not considered.

Domestic and Stock (D&S) use is often estimated through the number of bores in an area, but it is important to realise that the number of bores does not reflect the volume of use – e.g. a single bore with a good pump can supply an entire property; alternatively the same volume could be supplied by a windmill in each paddock. Thus, D&S assessment needs to be based on area and stocking rates.

Understanding the links between aquifers and surface water resources for each river reach that requires an assessment is not straightforward. Not all aquifers will have an impact on surface water – e.g. the Tertiary Limestone Aquifers in western Victoria do not discharge to surface waters and potentiometric level is below stream/lake bed. This is not the case for alluvial gravel aquifers on rivers such as the Mitta Mitta River in north-eastern Victoria, where it is recognised that there is good hydraulic connection to the river. It is also important to understand that the time lag between recharge and extraction can be very long – months to years if the aquifer is in limited hydraulic connectivity with the surface water. In this case the temporal pattern of extraction will have far less immediate impact on streamflows than direct diversions from the stream. Typically, information regarding these links is very limited and generally observational and anecdotal rather than quantitative. What information exists is generally held by local authorities or small research groups and will be time-consuming to obtain; if it has not been subject to technical review, it may be controversial.

4.7.2 Data requirements

Ground water is typically managed over an administrative area (cadastral boundaries, roads, etc.) – not over hydrogeological regions. Typically, these encompass areas where the aquifer has sufficient quality, yield, and pumping costs (depth to water) to render them suitable for commercial use. In some cases the groundwater system covers more than one catchment, and so apportioning impact to a stream flow in a catchment is difficult. A reasonable amount of thought and effort on this has been undertaken recently for the Victorian annual water accounts, and while there is a strong desire to report groundwater use against surface water basins, to date the data have been inadequate to allow this and the State Water Report 2003–04 only reported against declared groundwater management areas (DSE 2005).

The licensed volumes for each management area are provided within groundwater management plans compiled by local water authorities. While these give an indication of the volumes potentially used, accurate volumes are difficult to determine because typically there is a large proportion of under use. It would be necessary to access the annual reports (if prepared) that detail the volume of use.

Incorporation of groundwater information into hydrological indices

There are several options that would allow the incorporation of information regarding groundwater impacts on surface water flows. These can be divided into two types:

1) The development of new groundwater specific indices

2) The adjustment of existing indices.

Development of New Indices

The development of meaningful groundwater-specific indices that are relevant to surface water flows would require the following information:

- The relationship of the hydrogeological unit to surface water flows
- Current extraction volumes and the timing of extraction
- The impact of land clearing on recharge rates.

The preceding discussion indicates that such information is unlikely to be either readily available or widespread, and so the development of referential indices is not possible at present. It is however possible to incorporate some groundwater ‘flags’ into the reporting of the hydrological indices based on the following decision table (Table 29):

Table 29 Decision table for the incorporation of groundwater 'flags' into the reporting of hydrological indices.

Groundwater flag	Hydraulic connection of aquifer to stream	Active groundwater extraction	Groundwater extraction volumes as percentage of stream flow			Hydraulic connectivity with stream	
			Low	Medium	High	Slow	Fast
Red	Y	Y			Y		Y
Red	Y	Y		Y			Y
Red	Y	Y			Y	unknown	
Orange	Y	Y			Y	Y	
Orange	Y	Y		Y			Y
Orange	Y	Y		Y		unknown	
Orange	Y	Y		unknown		unknown	
Yellow	Y	Y		Y		Y	
Yellow	Y	Y	Y				Y
Yellow	Y	Y	Y			unknown	
Green	Y	Y	Y			Y	
Green	N						
Grey				unknown			

While not providing a quantitative assessment of groundwater impacts on streamflows, this makes use of current knowledge and will indicate the proportion of streams potentially impacted by groundwater extraction and thus the level of investment required to address knowledge gaps.

4.7.3 Adjusting surface water indices

Low flow adjustment

The contribution of groundwater to moderate to high flows will typically be small, and indices which represent annual flow volumes or high flows will be largely unaffected by the incorporation of groundwater information. Groundwater extractions will have the greatest effect on low flows (i.e. when flows are sourced largely from groundwater) and consequently the indices likely to change with the incorporation of groundwater data are those reflecting low flows. Thus the low flow indices can be adjusted (typically down) to reflect the current groundwater use as follows (Equation 34).

$$LFI_a = LF \times \left(1 - \frac{U_{GW}}{A_{LF}}\right) \qquad \text{Equation 34}$$

where: LFI_a is the adjusted low flow index

LF is the original low flow index

U_{GW} is the mean annual groundwater use from the relevant aquifer

A_{LF} is the mean annual surface water flow for the reach during the low flow period.

This method gives a very coarse estimate of the impact of groundwater use on surface water flows and is based on the following three assumptions:

1) the groundwater system has a direct hydraulic connection to the surface water system

2) the average connection (across all groundwater use) has a significant lag to stream flow, and

3) the low flow (or baseflow) is derived from groundwater.

While these are gross assumptions, they reflect the limited level of data and knowledge currently available. It is likely that the adjustment to the low flow index will be underplayed using this procedure; however, it provides a quantitative incorporation of the impacts of groundwater use without needing detailed information on groundwater–surface water connectivity and lag times.

4.7.4 Selection of a reporting method

After assessing the availability of data on groundwater use, it appears that obtaining groundwater use data based on surface drainage basins will be difficult. Most river basins do not have this information. Thus we recommend that, for an initial baseline assessment, groundwater impacts should be flagged as set out in the above section on the development of new indices.

Such an assessment will at least clearly identify the data gap for groundwater use and allow the spatial extent of its potential impact to be estimated. This will help in targeting further investment in monitoring and understanding groundwater–surface water interactions and in investigating how this can be better incorporated into hydrological models of surface water flows.

4.7.5 Data availability

Calculating hydrology indices and their components requires extensive datasets (Table 30) and obtaining the data is challenging. For many river reaches data of sufficient quality or type are simply not available. The calculation of both the NLWRA–HI and the SRA–HI is by nature confined to regulated rivers and, of these, suitable modelled data are usually only available for the lower reaches of the mainstreams and not for tributaries or lower order rivers. In combination, this results in a very small proportion of the river length being assessed for hydrological changes. For example, for the NLWRA assessment of river condition, 54 percent of the total length of *regulated* rivers within the Intensive Landuse Zone (ILZ) of Australia had suitable data available for calculating the hydrological index. This represented around 6 percent of the total river length having hydrological assessment as part of the Assessment of River Condition.

Table 30 Summary of data requirements for the hydrology indices considered for baseline assessment

	Data type	Time-step	Length of record
NLWRA–HI	Modelled natural	Monthly	> 30 years
	Modelled current	Monthly	> 30 years
	Observations	Monthly	> 30 years
	Observations pre-regulation	Monthly	> 15 years
	Observations post-regulation	Monthly	> 15 years
SRA–HI	Modelled natural	Daily	> 30 years
	Modelled current	Daily	> 30 years
FSR–HI	Modelled natural	Monthly	> 15 years
	Modelled current	Monthly	> 15 years
	Observations pre-regulation	Monthly	> 15 years
	Observations post-regulation	Monthly	> 15 years

The requirement of the SRA–HI for modelled daily data would result in even less of the total river length being able to be assessed using these indices since daily modelled data are typically only available for parts of New South Wales and Victoria.

Previous datasets

Considerable datasets are available from the NLWRA, which was conducted in 2000. These data were provided from state agencies and reflect model development at that time. It is possible to use these data and assume that the only changes to them will be minor (development conditions should still be set at MDB 'cap' levels of 1993–94 for the 'current' modelling scenario). However, this does not allow for policy changes on consumptive users' access to water, improvements in hydrologic models over the past 6 years, incorporation of better information on existing level of development, or the effects of an additional 6–10 years of data.

Running models and providing data is a costly exercise for state agencies. The data provided for the original NLWRA does not reflect the entire dataset that *could* be provided (model outputs are typically only provided for gauging stations). It is possible to obtain model runs for nodes between gauging stations, and this may improve the accuracy of the indices provided in some reaches; however, this becomes a substantial task for the agency supplying the data and for the person analysing it. Before requesting data from model nodes other than gauging stations, the spread of data points along a reach needs to be carefully investigated.

4.8 Discussion

4.8.1 Advantages and disadvantages

The approach adopted by the original NLWRA for calculating a hydrology index is simple, with sub-indices in general easily explained and interpreted. The indices can be calculated using paired monthly records. Modelled pairs are preferred; however, pre- and post-regulation pairs and a combination of modelled and observed data can also be used. As a consequence, the approach can make use of the greatest range of data available.

The inherent simplicity of the NLWRA approach also carries a disadvantage. Three of the sub-indices are correlated (the mean annual flow index, the flow duration curve difference, and the seasonal amplitude index), which biases the assessment of change toward changes in flow volume (since these sub-indices have equal weighting in the overall NLWRA–HI calculated from the four indices). The sub-indices chosen reflect basic changes in riverine hydrology, which may impact on in-stream biota; however, they lack the detail that may be useful in interpreting biotic responses.

The SRA–HI provides a far more rigorous assessment of hydrological change than does the NLWRA–HI. The SRA–HI sub-indices reflect a greater range of ecologically relevant hydrological characteristics and therefore the indices are of considerable use in interpreting biotic data. But the indices are at a higher level of complexity, and thus less easily explained and interpreted than the NLWRA–HI. However, the major disadvantage of the SRA approach is its requirement for daily modelled data pairs to calculate several sub-indices. The lack of widespread daily modelled data means that these indices can, at present, only be calculated for a small portion of the country.

Building on the previous work of the NLWRA led to the development of the FSR indices and SRA, and considerable advances have been made in their development. The indices were formulated to capture the same range of ecologically relevant hydrological characteristics as the SRA, but having the advantage of being calculable using monthly data (and only a minimum of 15 years of monthly data are required). The disadvantage of using monthly data is a lack of detail and thus loss of insight that is otherwise available from indices calculated from daily data.

The five FSR indices are independent, which means that combining them into a single index is unlikely to result in a biased overall score. The use of only 5 indices means that there is a reduced ability (compared to the SRA indices) to obtain extra interpretive information that may be contained in additional indices. However, the indices have been formulated to capture a good range of ecologically relevant hydrological characteristics, and the objective of the baseline assessment is to provide a large-scale assessment, the outputs of which can be used to drive further more detailed investigation. Four of the indices can be calculated for all or part of a year (e.g. winter or summer), which allows an additional level of detail to be obtained at times when major impacts on flow are occurring. This represents a significant advance on previous indices that were calculated on annual flows. For this project it was not intended to calculate and report these seasonal indices, but they can be calculated and used by local stakeholders as required.

Another advantage of the FSR indices is the non-parametric nature of their calculation, which takes into account the variability of the stream in the natural (or unimpacted state). This ‘variance correction’ is achieved by calculating the indices from percentiles taken from a flow duration curve – a neat, simple approach.

The disadvantage of FSR indices is that they have not been tested outside of Victoria and thus their applicability in tropical and arid zone rivers has not been investigated. It is not expected that there will be issue with their broader application; however, some key checks would need to be performed.

4.9 Recommendations

4.9.1 Selection of hydrology indices

The FSR hydrological indices developed by SKM currently represent the best option for the large-scale assessment of hydrological change. These indices are an advance on both the approach used by the NLWRA and the SRA, and they meet the criteria of the Framework. They provide an assessment of important, ecologically relevant flow characteristics and are able to use the greatest possible range of available data. They are currently used in the ISC assessments in Victoria and are likely to be used in the Tasmanian Index of River Condition.

4.9.2 Incorporation of groundwater assessment

An assessment of the data availability for groundwater use indicates that obtaining groundwater use data based on surface drainage basins will be difficult and that most river basins will not have this information. Jurisdictions should decide on their strategic need for the assessment of groundwater use, but it is recommended that until better data sets are available that groundwater impacts should at least be flagged.

Such an assessment will clearly identify the data gap for groundwater use and will allow the spatial extent of its potential impact to be estimated. This will help in targeting further investment in monitoring and understanding groundwater–surface water interactions and in investigating how this can be better incorporated into hydrological models of surface water flows.

4.9.3 Data requirements

The following data will be required to calculate the hydrology indices:

- modelled natural and modelled current monthly flows
- location (point shapefiles) of the location of the modelled flows
- Groundwater extraction information

- Hydraulic connection of aquifer to stream
- Groundwater extraction volumes.

A minimum of 15 years of current and natural (unimpacted) data from a site is required for calculating the indices. Where modelled data are not available, daily data from pre- and post-regulation can be used to provide an estimate of the indices (these indices will have a lower quality assigned to them).

It is recommended that data from the most current model runs be used to calculate the hydrology indices, as this will represent current management practices and give the longest period of data for analysis.

The daily observation data sourced for the first National Land and Water Resources Audit can be updated and used.

Interpolation and Extrapolation Rules

The data required to calculate the components of the HDI will not be available for all reaches. In the NLWRA 1, rules were developed for six different cases to allow sensible extrapolation of HDI values from reaches with hydrologic data to downstream reaches, and to allow interpolation of HDI values between reaches with hydrologic data. These rules can be applied to the recommended FSR indices.

Case 1: Unregulated reaches

If the current water extraction for an AWRC river is less than 0.5 percent of the mean annual runoff for the basin, then the HDI is set to 1 for all unregulated reaches in the basin. If the current water extraction for an AWRC river is greater than 0.5 percent of the basin's mean annual runoff, then the HDI for unregulated reaches in the basin is not assessed.

Case 2: Regulated reaches upstream of a reach with hydrologic data (tributaries are unregulated)

For reaches R1, R2, and R3 without data, HDI component values are estimated from the component values for reach R4 and from the mean annual flow in R4 and in the unregulated tributaries T1, T2, andT3 (Figure 14), according to Equation 35.

For R3: $Q_{R4}M_{R4} \cong (M_{R3}Q_{R3} + M_{T3}Q_{T3})$, or $M_{R3} \cong \frac{(M_{R4}Q_{R4} - Q_{T3})}{Q_{R3}}$, since M_{T3}=1.

Similarly, for R2 and R1: $M_{R2} \cong \frac{(M_{R3}Q_{R3} - Q_{T2})}{Q_{R2}}$ and $M_{R1} \cong \frac{(M_{R2}Q_{R2} - Q_{T1})}{Q_{R1}}$

Equation 35

where: $M_{R\#}$ is the value of the index component for reaches R1 to R4,

$M_{T\#}$ is the value of the index component for unregulated tributaries T1 to T3 (always = 1),

$Q_{R\#}$ = mean annual flow in the mainstem reaches,

$Q_{T\#}$ = mean annual flow in the unregulated tributaries.

Then, if for example, the downstream reach (R4) had M_{R4} = 0.5, and the upstream regulated reach (R3) contributed 70 percent of the total (regulated) flow, then M_{R3} for the regulated upstream reach would be = (0.5 – 0.3)/0.7 = 0.29, thus showing that the addition of the unregulated tributary had reduced the degree of regulation in proportion to the flow added.

Case 5: Regulated reaches downstream of a reach with hydrologic data (tributaries are unregulated)

For reaches R2, R3, and R4 without data, HDI component values are estimated from the component values for reach R1 and from the mean annual flow in R1 and in the unregulated tributaries T1, T2, andT3 (Figure 14), according to Equation 36.

For R2: $M_{R2} \cong \frac{(M_{R1}Q_{R1} + Q_{T1})}{Q_{R2}}$, since M_{T1}=1.

Similarly for R3 and R4: $M_{R3} \cong \frac{(M_{R2}Q_{R2} + Q_{T2})}{Q_{R3}}$ and $M_{R4} \cong \frac{(M_{R3}Q_{R3} + Q_{T3})}{Q_{R4}}$.

Equation 36

If there are major diversions in reaches R2–R4, then extrapolations of HDI values cannot be made.

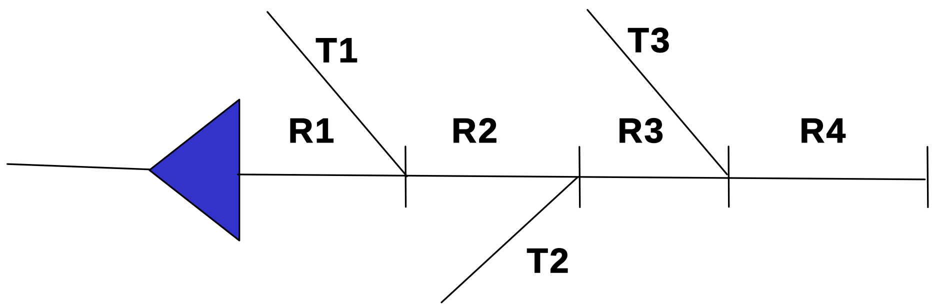

Figure 14 Schematic representation of a river downstream of a major impoundment.

Case 6: Regulated reaches between reaches with hydrologic data

For reaches R2 and R3 without data, HDI component values are estimated according to catchment area (Equation 37).

For R2: $$M_{R2} = \left(\frac{(A_{R2} - A_{R1})}{y} \times x \right) + M_{R1}$$ **Equation 37**

where: $M_{R\#}$ is the value of the index component for reaches R1–R4

$A_{R\#}$ is the catchment area of reach R#

$y = A_{R4} – A_{R1}$

$x = M_{R4} – M_{R1}$.

This assumes catchment-area increase is a reasonable surrogate for flow increase between the known points.

The following additional rule was also applied to all extrapolation and interpolation procedures:

- If any index component index is extrapolated to zero, HDI is set to non-assessed.

5 Habitat index

5.1 Components of the habitat index

The conceptual model of river condition identified three key components of the habitat: bed sediments, the riparian zone, and connectivity of the river (both upstream–downstream and with the floodplain). Measures of these components generated the habitat index.

5.2 Sources of data

Sources of data used for determination of the habitat index are shown in Table 31.

Table 31 Sources of data for the habitat index.

Input data	Source	Coverage	Data type
Bedload condition sub-index	Bedload sediment modelling	All reaches, except NT, Western Plateau and Gulf drainage divisions	Modelled loads, both current and natural
Riparian vegetation			
National Vegetation Information System (NVIS) data	Audit office	Approximately 70 percent of area assessed Used for validation	Cartographic data: Vegetation structure and floristics Vector: resolution of 1:25,000 – 1:250,000
Agricultural Land Cover Change (ALCC)	Audit office	Majority of area assessed Used for calculation of measure	Satellite imagery: Vegetation cover and change Raster – 100 m grid
Connectivity			
Longitudinal barriers – dams	Wild Rivers data Impoundments layer	All reaches	Cartographic data: Presence/absence of dams and locks Raster – 250 m grid
Lateral barriers – levees	Wild Rivers data Levees layer	All reaches	Cartographic data: Presence/absence of levees Raster – 250 m grid

5.3 Bedload condition sub-index

Several aspects of river geomorphology impinge upon river health. These include the width and depth of the channel, the condition of the banks, and the bed substrate and its forms (such as bars and pools). The bed substrate is the riverine component that suffers the greatest geomorphological impact. In many places, beds containing cobbles, boulders, rock bars, or woody debris have been smothered in deposits of sand and fine gravel. This has filled pools and other refugia, producing uniformly shallow beds that offer poor habitat and are a barrier to fish passage (Plafkin et al*.* 1989, Jeffers 1998). These sheets of sand are often referred to as sand slugs (Nicholas et al*.* 1995). They are the result of other geomorphological changes to bank erosion and channel form. Using current knowledge, it is also easier to assess the change in bed sediments in historical times across Australia than predict changes to bank erosion or channel form.

Geomorphological assessments are commonly made by site assessments and/or aerial photograph interpretation. This rarely includes a description relative to the reference condition. Case studies of channel change are sometimes produced with little ability to extrapolate to other conditions. Field surveys and aerial photograph interpretations were beyond the geographical scope and time constraints of the NLWRA. We used an alternative approach, developing a physically based conceptual model. The model assessed the change to the bed sediment regime as a result of accelerated erosion and sediment supply from upstream. The modelling used all available pertinent information and included considerable analysis of hydrological records; it also incorporated measurements of river widths and gully erosion extent. Details of the bedload modelling, conducted jointly with the sustainable agriculture theme of the NLWRA (refer to theme 5 report), are included in Appendix 1.

The bedload model predicts the mean annual historical deposition of bedload in river links as a result of the supply of sediment from bank erosion and gully erosion upstream. This deposited volume is expressed as a total bed accumulation of sand and gravel over historical times (measured in meters). Many river links have the sediment transport capacity to convey all supplied sand and gravel downstream, and these are predicted to have had no net deposition in historical times. Other riverbeds may have aggraded by as much as 10 m. Relative to these rates of deposition, it can be assumed that, on average, the beds of rivers under natural conditions are relatively stable, with no rapid accumulation or degradation. This is because of strong feedback mechanisms in river behaviour on time scales of thousands of years. Rapid deposition or aggradation of a river with an alluvial bed will force compensating effects, such as greater stream power, and a steady-state condition will eventually develop. Consequently, we assumed that the natural condition is for no net accumulation of sand and gravel on the bed at the level detectable by the bedload model. All net accumulation of sand and gravel under current sediment-supply conditions was assumed to negatively affect habitat.

Much of the valuable natural features of river beds for habitat – such as large stable cobbles and rocks on the bed, a range of flow depths, and adequate water depth at low flow – are lost within the first meter of so of sand and gravel deposition. Subsequent deposition makes the situation worse, drowning more extensive areas of the bed and filling deeper pools, but it probably has less impact than the first meter.

Taking the above factors into account, the bedload condition index (BCI) was defined as following (Equation 38):

$$BCI = 0.33 - 0.33\log_{10}(CDEP)$$ **Equation 38**

where CDEP is the predicted historical accumulation of sand and gravel on the bed. The bed sediment index varies from a value of 1 for *CDEP* = 0, through 0.67 for 1 m of deposition, to 0 for 10 m of deposition.

A logarithmic transformation of the bed deposition values was used because of the greater impact of initial deposition and because of the distribution of the results, which produced a high frequency of low values of deposition and very few cases above 5 m deposition.

5.4 Riparian sub-index

5.4.1 Basis of riparian sub-index

The condition of a riparian zone is commonly assessed using a combination of structural and floristic information. For example, the US EPA habitat assessment protocols (Plafkin et al*.* 1990) require assessment of both the vegetation cover and type. Such an approach is used by various state and territory agencies responsible for monitoring stream condition. Additionally, the extent to which the riparian vegetation is native or alien, and the linear connectivity of woody vegetation along a stream, are often recorded.

The NLWRA office commissioned a review of riparian vegetation data across Australia as a component of theme 7. The Riverine Vegetation Mapping Scoping Study (SKM 2000) found that there was no national and few state schemes in which the riparian zone was delimited or riparian vegetation defined. The review recommended use of a geomorphological basis for definition of the riparian zone (i.e. use of some feature of the floodplain). For upland regions in which there was no floodplain, the report recommended a vegetation-based definition that included floristic and structural criteria.

Similarly, the review found that the concept of riparian 'health' was poorly understood and usually not defined. It was recommended that the condition of riparian vegetation be measured in terms of the vegetated stream-length, especially in terms of the abundance and continuity of tree cover and the presence of alien species. Such an approach was seen as particularly appropriate where clearing is the major threat to riverine vegetation.

A major finding of the review was that riparian vegetation mapping across Australia had been conducted at a range of scales, used different methods, and was far from comprehensive. Considerable resources would be required to produce a national map of riparian vegetation at a useful scale.

For the NLWRA, the Agricultural Land Cover Change (ALCC) dataset was used to calculate an assessment of riparian condition. Based on 1995 satellite imagery with 100 m pixels, this dataset has seven categories of land cover, one of which is *native or exotic woody vegetation*. The riparian zone was taken to be that area of land extending 250 m on either side of a river, and riparian vegetation was taken to be that vegetation within the riparian zone. The health of the riparian vegetation was defined in terms of structural attributes, and hence a riparian zone that had all of its cells with *native or exotic woody vegetation* was taken to be in natural condition and accorded a riparian sub-index value of 1. A riparian zone with no such cover was judged to be degraded and received a score of 0.

It was acknowledged that this was a coarse generalisation. Narrow upland riparian zones may be misrepresented as they are at a smaller scale than the 100 m ALCC dataset, and wide floodplain riparian zones may receive a sub-index score of 1 even though there may have been some vegetation loss on the further floodplains. An additional issue uncovered was that the modelled location of channels used to define the reaches did not always match the actual location of channels as interpreted from imagery. Therefore in some situations the riparian index was erroneous because of this spatial mismatch.

Since the Audit, the extent of specific riparian vegetation mapping has not markedly improved. The resolution of regional vegetation mapping products is coarse, and they ignore vegetation on smaller streams. However, a major outcome of the Audit was the first national compilation of vegetation mapping, which is now consolidated as the National Vegetation Information System (NVIS). The most recent releases of these data are based on updates from the jurisdictions in 2005, and these represent on-ground dates over the previous four years (except for some large areas of NSW from 1997). At least in NSW and Victoria, there are now defined riparian vegetation types, but these are only mapped when their scale makes them visible at the standard mapping scale, which varies from 1:25,000 to 1:5,000,000. Hence, on smaller streams, riparian vegetation types are still not mapped; only the upland vegetation will be shown in what would be the riparian zone.

The data available in NVIS Stage 1 Version 3.0 (DEH 2006) include two compatible and complete national coverages, which are compilations of vegetation mapping data, generalised to 27 Major Vegetation Groups (MVG) and then rasterised to 100 m pixels. The MVGs are based on structure (dominant growth form, cover, and height) and floristics (dominant genus in the top stratum) (NLWRA 2001b).

One of the coverages, known as Present Vegetation, reflects the most recent field data available. The other coverage represents expert opinion and historical data reconstructions of pre-1750 vegetation using the same MVG descriptors. This has been used as the reference condition for the riparian vegetation condition assessment developed here.

At the regional and national scale, these NVIS layers are the best available level of vegetation description for comparing vegetation change since European influence commenced. Vegetation is dynamic, so we cannot conclude that all the mapped vegetation change is related to European effects. However, the means by which the pre-1750 layer was derived suggests that this is the major effect captured. The MVG level of description corresponds to the Level III (Broad Floristic Formation) of the NVIS information hierarchy. In terms of data quality, it is important to note that NVIS data are actually compiled at the finer scales of Level V (association) or Level VI (sub-association) before aggregation to broader units (Levels I–IV) (ESCAVI 2003).

5.4.2 Calculation of the riparian condition sub-index (RCSI)

For the development of a riparian condition index, the MVGs were further grouped into three categories of Forest, Non-forest, and Modified (Table 32). The definition of Forest was matched as closely as possible to the somewhat different image-based one used for the Forest Extent dataset (DEH, AGO version 3, 2006). A forest was defined as vegetation with a minimum of 20 percent canopy cover, potential to reach 2 m tall, and having a minimum area of 0.2 ha. Since the NVIS data had a pixel size of 1 ha the last criterion was not considered.

Table 32 Assignment of the NVIS MVGs to categories.

MVG #	Major Vegetation Group name	Category
1	Rainforest and vine thickets	Forest
2	*Eucalyptus* tall open forests	Forest
3	*Eucalyptus* open forest	Forest
4	*Eucalyptus* low open forest	Forest
5	*Eucalyptus* woodlands	Forest
6	*Acacia* forests and woodlands	Forest
7	*Callitris* forests and woodlands	Forest
8	*Casuarina* forests and woodlands	Forest
9	*Melaleuca* forests and woodlands	Forest
10	Other forests and woodlands	Forest
11	*Eucalyptus* open woodlands	Non-forest
12	Tropical Eucalypt woodlands/grasslands	Forest
13	*Acacia* open woodlands	Non-forest
14	Mallee woodlands and shrublands	Forest
15	Low closed forest and closed shrubland	Forest
16	*Acacia* shrublands	Forest
17	Other shrublands	Non-forest
18	Heath	Non-forest
19	Tussock grasslands	Non-forest
20	Hummock grasslands	Non-forest
21	Other grasslands, herblands, sedgelands, and rushlands	Non-forest
22	Chenopod shrub, samphire shrub, and forblands	Non-forest
23	Mangroves	Forest
24	Inland aquatic – freshwater, salt lakes, lagoons	Excluded
25	Cleared/modified native vegetation, buildings	Modified
26	Unclassified native vegetation	Non-forest
27	Naturally bare – sand, rock, claypan, mudflat	Excluded
28	Sea and estuaries	Excluded
29	Regrowth, modified native vegetation	Non-forest
99	Unknown/no data	Excluded

The riparian condition sub-index was based on an overlay comparison of the two NVIS data layers after the categorisation. This was assisted by considering the possible changes in vegetation category in a transition matrix (Table 33). Vegetation could be either unchanged, converted to another kind of native vegetation, or cleared and hence fall into the modified category, principally MVG #25.

Table 33 Vegetation transition outcomes from pre-1750 to present

		PRE-1750 CATEGORIES	
		Forest	Non-forest
PRESENT CATEGORIES	Forest	No Change	Converted
	Non-forest	Converted	No Change
	Modified	Cleared	Cleared

The number of pixels showing as *No Change*, *Converted,* or *Cleared* in each reach was expressed as a fraction for the zone extending to 100 m each side of the *banks*, i.e. the channel width was not included. The riparian condition sub-index (*RCSI*) was calculated as follows (Equation 39):

$$RCSI = \text{No Change} + \text{Converted}/2 \qquad \textbf{Equation 39}$$

where the variables refer to the proportion of pixels of that kind in each reach. Thus, if the entire reach was scored as *No Change* along both sides it received a score of 1, and if all vegetation was *Cleared* then it received a score of 0. A score of 0.5 would result from a partially *Cleared* reach, which had *No Change* along both banks for half its length, or *No Change* one side and *Cleared* the other, or any such combination.

5.4.3 Assumptions and limitations of the RCSI

The formulation of the *RCSI* was based on the assumptions that:

- mapped vegetation near rivers characterises the actual riparian vegetation at the level of the MVG categories *Forest*, *Non-forest,* and *Modified*
- where there is presently a corridor of *Forest* vegetation surrounded by cleared land, the corridor is mapped
- the mapped location of such corridors matches that of the modelled streamline
- *Forest* and *Non-forest* MVGs contribute equally to riparian health, and
- conversion of the native vegetation (*Forest* ↔ *Non-forest*) is only half as detrimental to riparian health as clearance (*Forest* → *Modified, Non-forest* → *Modified*), e.g. for cropping.

The importance of these assumptions would vary within and between catchments and more detailed evaluation and modelling in a range of catchments is required to assess them fully. In the NLWRA the reference condition was that riparian zones had woody vegetation. The approach here has not assumed this. However, where riparian vegetation is not specifically mapped, then there is the assumption that the local upland vegetation is in the same category as the actual riparian vegetation. These conditions will often obtain on smaller upland channels and here the assumption is likely to be valid more often.

There was some limited floristic information able to be used in determining the *RCSI*. In categorising the MVGs as *Forest* or *Non-forest*, most MVGs have a descriptor indicating the dominant genus of the top stratum, and this can be useful. It is unlikely that standard vegetation mapping as such will be able to provide full floristic assessments of riparian vegetation and hence site-based surveys and distribution models are needed to bridge the gap.

At the present time, NVIS datasets are focussed on native vegetation and do not identify levels of invasion by alien species. A riparian zone largely occupied by alien trees would be considered as modified vegetation, but it would not necessarily be mapped because of scale issues. If it was wide enough to be mapped (as on larger rivers) it would lead to a reach receiving an *RCSI* of less than 1 in proportion to the extent of alien species in the top stratum. Alternatively, if not mapped, it would show up as the same type as the extra-riparian vegetation of the reach, and so could be any category and therefore have a value of *RCSI* between 0 and 1. Obviously such a riparian zone should rate less than a zone in which the vegetation was all native, but in the absence of riparian floristic information at suitable scales, this limitation has been accepted.

The other limitation of the approach used is the occurrence of misalignment between mapped vegetation units and the riparian zone boundaries. This arises from the latter being based on modelled channel positions, whereas the former are based on image interpretation and topographic maps. This issue was raised in the NLWRA and will require more spatial analysis than possible here.

5.4.4 Calculation of the riparian structure sub-index (*RSSI*)

To assess current riparian vegetation structure for 2005, the Forest Extent 2005 product of the National Carbon Accounting System has been used. This product is based on Landsat imagery and available at a spatial resolution of 25 m for the area of Australia considered to be treed vegetation (most of the semi-arid and non-arid zone). The analytical methods used are described in Furby (2002).

In essence, the spatial data layer for Forest Extent represents the areas where the pixel reflectance patterns have been interpreted to be most likely Forest within the definition as described earlier. Although the pixel size is 25 m, the definition of Forest requires that there be a minimum area of 0.2 ha i.e. one 50-m pixel. The method was not used to distinguish alien species such as willows, and willows would be considered forest if they also met the area criterion.

To ensure comparability with the *RCSI* , the structural index, *RSSI*, was derived by summing the Forest pixels within 100 m of each bank in each reach and expressed as a proportion of total pixels. This then represents the proportion of forest cover in 2005 with 100 m of the channel for a reach.

In order to check the robustness of this index it was compared against the proportion of forest as derived from the NVIS Present vegetation mapping coverage. These two measures of vegetation structure should be similar overall, but the deviations are illuminating. Though not identical, they are comparable in terms of:

- considering the same piece of land – defined by modelled streamlines
- using a reasonably similar definition of forest – based on grouping of MVGs to match NCAS definition
- being within a similar time frame – about 10 years.

In both Tasmania and parts of Victoria, there is a close correlation (*r* ~ 0.9) at the link or reach level between these estimates of riparian structure (Figs. 4.11, 4.12). Many reaches fall into the high and low ends of these measures, and there are some substantial differences for the intermediate values. The geographic location of these differences may indicate why the measures are sometimes at variance and whether this is related to their method of calculation or to real temporal change in the vegetation. The overall conclusion is that the NVIS map data and Landsat interpreted data are broadly comparable at a coarse level of vegetation classification.

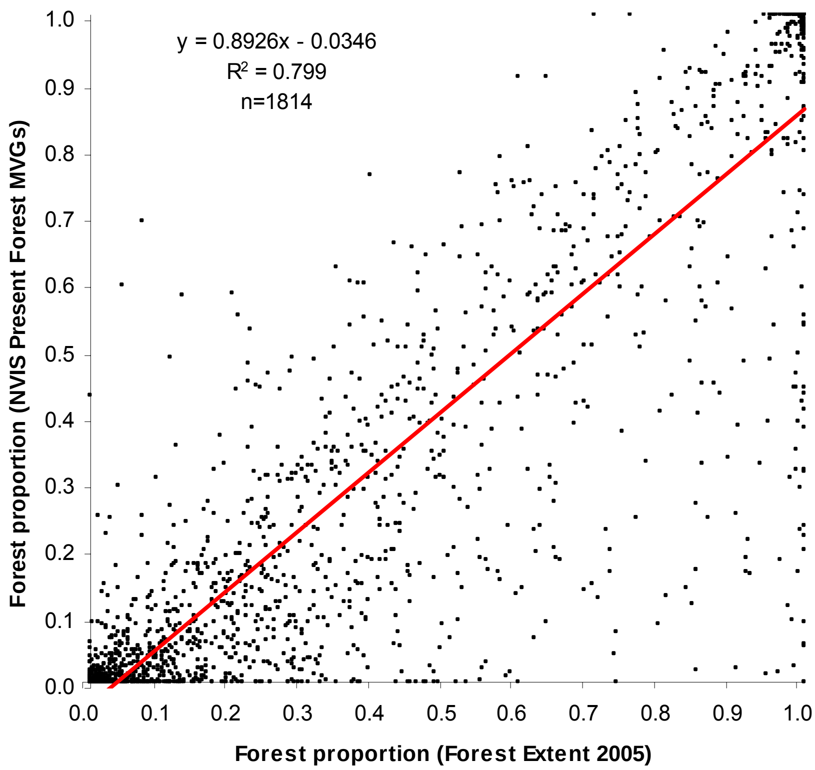

Figure 15 Correlation between two measures of present riparian vegetation structure along reaches in six Victorian SWMAs for a riparian width of 100 m.

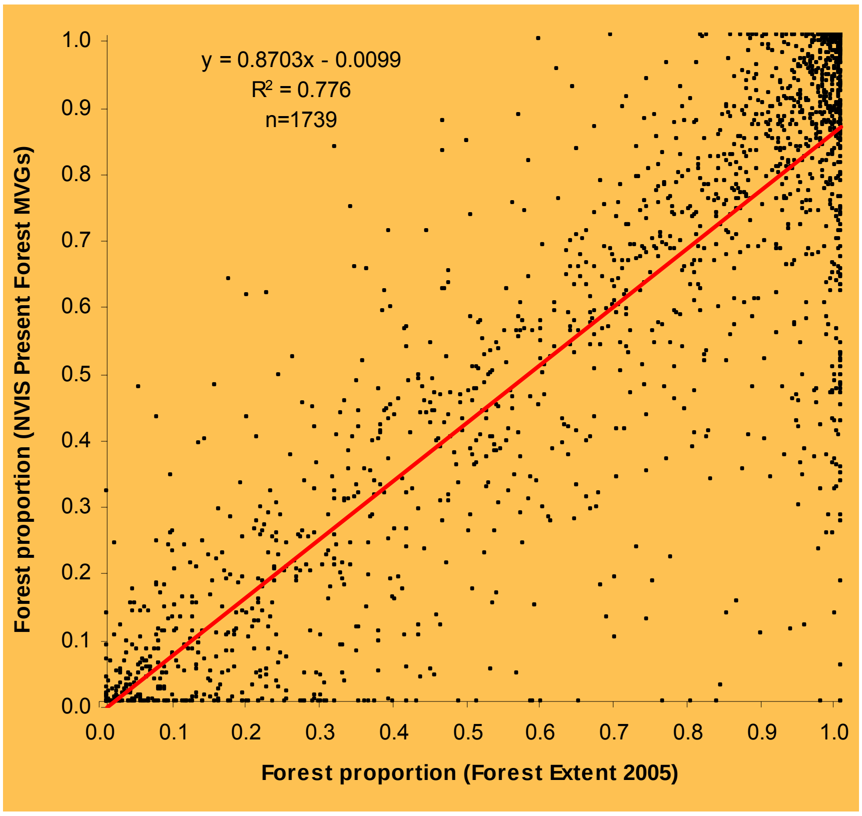

Figure 16 Correlation between two measures of present riparian vegetation structure along links in all Tasmanian rivers for a riparian width of 100 m.

5.5 Connectivity sub-index

The conceptual model of habitat identified two important components of connectivity: upstream–downstream connectivity (longitudinal), and connectivity with the floodplain (lateral). The former is important for the migration and breeding of many fish species, the latter for movement of water, biota, and material across the floodplain. The connectivity sub-index was calculated from data in the wild rivers dataset, using the impoundments data to calculate the longitudinal component and the levee data to calculate the lateral component.

It should be noted that the lateral barrier measure did not include the effect of stream channel expansion. Stream channel expansion allows a much higher channel flow capacity and reduces floodplain flooding and consequently exacerbates the effect of lateral barriers.

5.5.1 Longitudinal barriers

The Wild Rivers data contain information on four categories of longitudinal barriers: major structures, weirs, locks and sluice gates, and minor structures. Major structures include reservoirs and weir pools behind large weirs (e.g. Yarrawonga and Torrumbarry).

The weir category includes a large number of small structures that do not fall on streams, for example, farm dams. The locks and sluice gates category included only large- to medium-sized structures used for navigation or flood regulation. Minor structures are farm-dam sized structures. Only data in the categories ‘major structures’ and ‘locks/sluice gates’ was used because the structures in other categories were either off-stream or too small to influence longitudinal connectivity.

For longitudinal connectivity, the key issue was the extent to which barriers impeded longitudinal movement of biota. Although barriers will also affect flow, these impacts will be quantified in the Hydrological Disturbance Index. The extent to which structures create a barrier to natural movement of biota will depend on a range of factors including:

- The frequency that a structure effectively drowns out, taking into account fishways, etc.
- The presence of fish (and other) species in the river that need to migrate, their migration patterns, and locations of structures.

This calls for detailed biological knowledge of the species involved, of the structures in the river, and their operational characteristics. Neither of these sets of information was available on a national basis. As an interim approach for the assessment of river condition, a simple rule was used to determine the longitudinal connectivity:

- The two types of structures used from the wild rivers data were weighted (major structures, weight = 1, locks or sluice gates, weight = 0.5). Weights were based on a judgement of the relative impact of these structures on fish movement.

Barriers upstream and downstream of the reach under consideration also needed to be taken into account in calculating longitudinal connectivity. The distance that barriers will affect different fish species is not known, but advice from fish biologists was that many species would be affected at distances up to 20 km (Mark Lintermans, Tim O’Brien, Mark Kennard, pers. comm.).

Longitudinal connectivity was calculated at the river link level for greater accuracy, and then aggregated to the river reach level (Equation 40). Taking a precautionary approach, the effect of structures up to 40 km upstream and downstream were considered. Forty kilometres corresponded to approximately four river links (mean length 12.6 km), so the effects of barriers in links up to four links distant were considered. In this process the algorithm used to calculate connectivity down-weighted structures in a way that depended on their remoteness from the link being determined. The more remote the structure, the lower the influence.

$$LongI = 1 - ((b_0 * w) + \frac{(b_1 * w) + (b_{-1} * w)}{2} + \frac{(b_2 * w) + (b_{-2} * w)}{3} \ldots \frac{(b_4 * w) + (b_{-4} * w)}{5})$$

Equation 40

where *LongI* = dam impact, b_0 indicates a barrier in the reach, b_1 a barrier in the next reach upstream, b_{-1} a barrier in the next reach downstream, and w_n is the appropriate weight. *LongI* is restricted to a minimum of 0.

For example, a reach with a dam in it would score 0. A reach with no dam and a dam two reaches upstream would score 0.7.

5.5.2 Lateral barriers

The Wild Rivers dataset contains data on the presence of artificial levees in the three states where levee construction has been most prominent: New South Wales, Victoria, and South Australia. These levee data were used to derive a measure of the barriers to lateral movement of biota or material from a reach onto the floodplain. Ideally, the importance of a levee in restricting lateral flow of floodwater would be based on the ecological significance of the area of floodplain hydraulically isolated by the levee, and the degree of isolation. These data were not available. We developed a pragmatic measure of the impact of levees as lateral barriers by making some simplifying assumptions:

- all floodplain areas were assumed to be of equal importance
- where levees have been built, they totally disconnect the floodplain from the river channel.

Based on these assumptions, the extent to which levees impact on a reach was taken to be the length of the levees falling in the reach sub-catchment divided by twice the length of the reach (Equation 41). The significance of this is that if the length of the levees sums to twice that of the reach, it is assumed that levees line both sides of the entire reach. The reach would receive a lateral connectivity score of 0. Where there are no levees the reach receives a score of 1.

$$LatI = 1 - (\frac{L_l}{L_r * 2})$$

Equation 41

where *LatI* = Lateral connectivity measure, L_l = length of levees in reach, L_r = length of reach. *LatI* is restricted to a minimum of 0.

The length of levees along a reach were calculated by summing the area identified as levee within the reach sub-catchment and determining the linear extent of this area if it were at least 250 m in width (the equivalent of a wild rivers dataset grid cell). This approach assumes levees are linear and fall within the width of one grid cell. Levees on upstream or downstream reaches do not affect the lateral connectivity of the reach of interest and were not used to calculate the lateral connectivity measure.

Combining the measures of longitudinal and lateral connectivity

The longitudinal and lateral connectivity measures were combined to produce the connectivity sub-index. Consideration was given to weighting these components based on their relative impact on the biota. Ultimately, weighting was not applied because there is little information on their relative impact and because they tend to occur in different parts of a river. Levees commonly occur in lowland areas of rivers, sections in which dams, weirs, and locks are less common or less of a barrier to biotic movement. In the upper parts of catchments, dams are generally larger and levees are absent.

The connectivity sub-index was calculated as the sum of the deviations of each measure from a pristine condition (Equation 42). This approach was taken because the two measures tend to represent connectivity impacts on different parts of the catchment and so should be combined in an additive way.

$$CS = LongI + LatI - 1 \qquad \text{Equation 42}$$

where *CS* = connectivity sub-index, *LongI* = longitudinal connectivity, *LatI* = lateral connectivity. *CS* is restricted to a minimum of 0.

5.5.3 Validation

The purpose of the connectivity sub-index was to quantify the extent of the impact on biota, particularly fish, by lateral and longitudinal barriers. Research on fish movement has shown that barriers have an effect on fish movement, but the exact nature of these impacts is not yet resolved. Moreover, the time scales over which barriers produce impacts are long, making validation of the connectivity sub-index difficult. The precautionary principle has been applied in the inclusion of this sub-index, with validation of the effects of barriers to connectivity expected to occur over time with the use of the measure.

5.6 Reach-scale habitat index

The bedload condition and the riparian and connectivity sub-indices were integrated into an overall habitat index at the reach scale using standardised Euclidean distance (Equation 43). As we could find no clear indications that any of the sub-indices had greater ecological significance than the others, they were not weighted.

$$PHI_r = 1 - \left(\frac{\sqrt{(1-BI_r)^2 + (1-RSI_r)^2 + (1-CS_r)^2}}{\sqrt{3}}\right) \qquad \text{Equation 43}$$

where PHI_r = reach habitat index, BI_r = reach bedload sub-index, RSI_r = reach riparian sub-index, CS_r = reach connectivity sub-index.

5.7 Basin-scale habitat index

Basin-scale habitat indices were calculated by taking the length-weighted mean of the reach-habitat indices within a basin (Equation 44). Reach-scale indices were length-weighted so that longer reaches would exert proportionally greater influence on the basin index than would smaller reaches.

$$PHI_b = \frac{(PHI_{r1} * L_{r1}) + (PHI_{r2} * L_{r2})...}{\sum L_{r1} + L_{r2}...}$$ **Equation 44**

where PHI_b = basin habitat index, PHI_{r1} = habitat index for reach 1 within the basin, L_{r1} = length of reach 1.

6 Water Quality Index

6.1 Introduction

Water quality encompasses a range of chemical and physical attributes that are important aspects of riverine habitat character and are useful indicators of catchment and riverine transport and biochemical transformation processes. In the FARWH, water quality assessment focuses on ecosystem health, more so than consideration of water quality for drinking water and recreation.

The river nutrient transport modelling methods used in the Assessment of River Condition (ARC) in NLWRA 1 are applicable for the FARWH. The description of these methods below and in Sections 4.7 and 4.8 is derived from Prosser et al. (2001b) and Norris et al. (2001). Recent updates to the modelling methods as cited are described in Wilkinson et al. (2004).

6.2 The importance of water quality to ecosystem health

Water quality includes attributes of water such as temperature, suspended particulate material, and dissolved material. Suspended particulate material includes organic and inorganic sediments, and the carbon and nutrient content of these fractions may be measured. Organic forms of carbon are important aspects of water quality from an ecological perspective: because organic carbon is the basis for all life, the sources of organic carbon strongly influence aquatic food webs. The inorganic fractions of suspended material are also important since, together with organic suspended matter, they determine light penetration into the water. This alters water column habitat and affects thermal conditions. Light availability and temperature affect bacterial, algal, and zooplankton physiology, as well as the feeding and movements of invertebrates and fish. Nutrients include the macronutrients (phosphorus, nitrogen, sulphur, potassium, magnesium, and calcium) and micronutrients or trace elements (e.g. iron, copper, silicon). In water quality assessments, the macronutrients phosphorus and nitrogen are usually focussed on because these are commonly 'limiting' in terms of rates of biological activity or biomass, while the other macronutrients commonly occur in excess of biological requirements.

Dissolved material includes ionic forms of macro and micronutrients (e.g. phosphate, nitrate, carbonate, sulphate, calcium, and sodium ions), as well as dissolved gases including oxygen and carbon dioxide. These all affect aquatic life in differing ways, and are more or less important in different situations. Again, because the nature of carbon sources helps determine the character of aquatic food webs, measures of dissolved organic carbon and dissolved carbon dioxide are ecologically important. Levels of dissolved oxygen are also ecologically important because oxygen is essential for most aquatic organisms. Decreases in dissolved oxygen can be caused by the respiration by plants, animals, and some bacteria, as well as by chemical and bacterial processing of organic matter in the water and bottom sediments. Massive reductions in dissolved oxygen, such as may result from decomposition of algal blooms, can lead to widespread fish and invertebrate mortality.

The sum of all the ions dissolved in the water determines the 'salinity' of the water. Salinity is commonly assessed using the more easily measured surrogate, electrical conductivity. The composition of the salts in the water reflects the relative contributions from rainwater, atmospheric dust, and catchment rocks and soils. The salts derived from the catchment reflect rock and soil type, age, and vegetation. In arid and semi-arid regions, evaporation, which concentrates salts in the aquatic environment, is a key process.

The most commonly measured aspects of water quality are water temperature, pH (acidity), and the concentrations of dissolved oxygen, suspended sediments, phosphorus, and nitrogen. Turbidity is determined by the concentrations of suspended material. Surrogate measures of turbidity are often made based on the depth of light penetration. Many different measures of nutrients are used, largely because of a range of different laboratory analysis methods. Measures of total phosphorus (TP) and total nitrogen (TN) are often reported, which include the dissolved fraction and the particulate fraction. Aquatic plants and algae only use nutrients in the dissolved phase; however, under certain conditions nutrients can move from the particulate to the dissolved phase as a result of chemical processes. For example, while the majority of phosphorus in Australian rivers is attached to clay sediments, a proportion of this phosphorus can be released from the sediments (either in the water column or on the bottom sediments). The movement of phosphorus between the dissolved and sediment phases is controlled by several factors, including sediment mineralogy (which affects the strength of the chemical bonding of the phosphate ions) and the concentrations of phosphate and oxygen in water. Anoxic conditions are particularly conducive to phosphorus release from sediments. Given that the availability of sediment-bound nutrients depends on local sediment and water conditions, choosing measures of biologically available nutrients for a broad-scale assessment is difficult. Most broad-scale assessments therefore use measures of total nutrient loads and concentrations, based on measurements of both dissolved and sediment-attached nutrient forms.

The loads and concentrations of both dissolved and particulate material in rivers are a function of the sources of material in the catchment, the transport pathways, and the biochemical transformation processes. Catchment geology, soil type, topography, land use, and land management (including riparian zones) all affect the sources of material to river systems. These, together with climate and river channel form, determine the transport pathways. Because of these complexities, water quality varies greatly, not only spatially, but also temporally. Flow conditions vary enormously through time, and have a large influence on water quality. Because of these variations, different water quality assessments may have different interpretations. For example, average concentrations may indicate typical habitat conditions, while water quality under extreme conditions (such as drought) may be very different and have a large impact on the populations of aquatic plants and animals. Total nutrient loads may give a better indication of the potential total biomass of an aquatic system than does nutrient concentrations. Total loads exported are probably the most useful measures for assessing the potential impact on downstream ecosystems such as estuaries or coastal reefs.

6.3 Approaches to water quality assessments

The two most widely used approaches to assessing water quality are:

- To compare measurements of current condition to some reference condition – usually measurements of pre-disturbance conditions or measurements from comparable but undisturbed sites
- To compare measurements, 'standard values' deemed to represent threshold conditions for a particular purpose, such as drinking water or ecosystem protection.

In Australia, the most commonly applied 'standards' are the national standards established jointly by the Agriculture and Resource Management Council of Australia and New Zealand (ARMCANZ) and the Australian and New Zealand Environment and Conservation Council (ANZECC, 1999). While earlier versions of these guidelines provided recommendations for nutrient concentrations in a different ecosystem types, the most recent guidelines provide recommended median low-flow concentrations for different levels of protection of different aquatic ecosystems, including lowland rivers, upland rivers, freshwater lakes, wetlands, estuaries, and coastal and marine waters. The recommended levels are based on data for multiple reference sites in each state for each ecosystem type.

The Index of Stream Condition (ISC; Ladson et al*.* 1999) used in Victoria to assess stream condition includes a water quality index that uses a similar philosophy to the ARMCANZ/ANZECC guidelines. This approach uses condition ratings, which are given threshold values for different measures (phosphorus concentration, turbidity, pH, and electrical conductivity) for different river types (mountain, valley, plain). The threshold values for the ratings were based on existing assessment standards and professional judgement.

A workshop in Brisbane in early 1998, conducted for the NLWRA, sought input from the states and territories on water quality issues in their jurisdictions. That workshop identified six indicators as being of national importance: electrical conductivity (Ec), pH, total phosphorus (TP), total nitrogen (TN), faecal coliforms, and suspended solids (SS). These outcomes also guided the choice of water quality variables used in the theme 7, project two, assessment of river condition. The variables pH and faecal coliforms were not used: pH tends to be a robust measure, not responding unequivocally to catchment change and with a wide natural range; faecal coliform counts can be a useful surrogate for impacts on drinking or recreational water quality, but is of minor relevance to ecosystem health. Toxicants were not included in the list of water quality indicators from the Brisbane workshop, possibly as a result of the paucity of available data and the wide range of toxicants. Toxicants were nevertheless included in the ARC Nutrient and Suspended Sediment Load assessment as they have a critical impact in some rivers.

6.4 The ARC Nutrient and Suspended Sediment Load Index

The conceptual framework for the Assessment of River Condition includes a complete Water Quality Index. However, because of limited data availability for rivers in the area assessed, the water quality assessment for the ARC at the river-reach scale relies on those aspects of water quality that can be reasonably modelled to provide index values for all river reaches (Table 34). This restricts the reach nutrient and suspended sediment load index to assessments of suspended sediment load and nutrient loads, using the reference condition approach (rather than comparison against standards). Thus, for clarity, the index in the current assessment was named the *Nutrient and Suspended Sediment Load Index.* A more complete water quality index is the aim of future assessments, and might include additional variables such as water temperature and dissolved oxygen concentrations; it might also consider sediment and nutrient concentrations as well loads.

The index is primarily a comparison, using modelled data, of current to natural average annual loads of nutrients and suspended sediments. The modelled sediment and nutrient loads compare well with load estimates based on measured water quality and flow data. However, because the index values are measures relative to a modelled natural condition, they cannot be directly compared to water quality assessments based on exceedence of guideline threshold values. This is because, in exceedence guidelines, the relative increases over natural conditions implied by the thresholds do not correspond, in the ARC, to the relative increases over natural conditions used to define sediment and nutrient load assessment categories. Moreover, water quality measurements are typically biased towards low flow conditions and so do not necessarily correlate well with total loads.

The reference condition for nutrient and suspended sediment loads was established by modelling pre-disturbance conditions. Three measures were defined: a suspended sediment index (SS), a total phosphorus index (TP), and a total nitrogen index (TN). The nutrient and suspended sediment load index was determined from the SS, TP, and TN measures by taking the worst of the three measures as the overall index. The theory underpinning this approach is that if one of the three measures indicates poor condition, then the reach is in poor condition – regardless of whether the other two measures have been impacted or not. The philosophy surrounding this approach, and approaches used with other indices, is discussed in more depth in the section on integration and aggregation. No weighting of the three measures was applied in determining the nutrient and suspended sediment load index. Weighting would have been used if a measure had a demonstrated effect significantly greater than the other two. Although there is a substantial body of evidence relating these measures to changes in aquatic biota, we have poor quantitative understanding of their relative importance in Australia.

Additional to the SS, TP, and TN indices, toxicant data from the National Pollutant Inventory (NPI) and electrical conductivity data (Ec) from theme 7, project 1, were recorded for each reach where present. However, these data were too sparse to be used as components of the reach nutrient and suspended sediment load index. Although not incorporated into the reach nutrient and suspended sediment load index, an assessment of toxicants and salinity are included in the ARC database for reaches where data are available.

Assessments of salinity at the AWRC basin-scale from NLWRA theme 7, project 1, are included in the basin-scale nutrient and suspended sediment load index. Calculation of the basin index was done in several steps. An interim basin index was calculated from the reach nutrient and suspended sediment load indices by combining them using standardised Euclidean distance. Different reaches have different importance, depending on their location in the basin, so in this process reach scores were weighted by their catchment area. In AWRC basins having site data for over half the basin, salinity index (BSI) values (range 0–1) were determined from the percentage of sites in each of the *Good*, *Fair*, and *Poor* categories, as follows (Equation 45):

$$BSI = \frac{(\%Good * 1) + (\%Fair * 0.5) + (\%Poor * 0)}{100}$$

Equation 45

The final basin nutrient and suspended sediment load index was taken as the lower of the BSI and the index based on reach SS, TP, and TN values (following the philosophy used to calculate the reach nutrient and suspended sediment load index).

Table 34 Data sources for nutrient and suspended sediment load assessments.

Input data	Source	Coverage	Data type
Point source toxicants	NPI database	Limited number of sites across Australia	Potential emissions from facilities
TN, TP	Task 4 Theme 7, project 2	All reaches, except those in Western Plateau, SA Gulf, and NT drainage divisions	Modelled loads, both current and natural
SS	Task 3 Theme 7, project 2	All reaches, except those in Western Plateau, SA Gulf, and NT drainage divisions	Modelled loads, both current and natural
EC	Theme 7, project 1	Sites, aggregated up to AWRC basins	Condition of sites in relation to guideline values

6.4.1 Total Phosphorus and Total Nitrogen indices

The total phosphorus and total nitrogen indices are based on the ratio of the ‘natural’ (pre-disturbance) average annual total load for a reach to the current average annual total load for a reach. The values of average annual total loads were obtained from the modelling of phosphorus inputs to the river network and the transport and transformation of these loads through the river network (see nutrient modelling section). While in a few isolated cases the modelling predicted only small reductions in average annual total loads from natural to current, generally the current loads are lower than natural, and hence the range of the above ratio is effectively 0 to 1. For reporting ARC indices at river basin and state scales, the following standard ranges were used: 0–0.25 (severely modified), 0.25–0.50 (substantially modified), 0.50–0.75 (moderately modified), and 0.75–1.00 (largely unmodified). To allow meaningful interpretations of the total phosphorus and nitrogen sub-indices against these standard ranges, the calculated ratio values were transformed to provide final index values. The final sub-index was the ratio raised to the 0.55 power. This had the effect of raising the national mean value of the total phosphorus sub-index to around 0.5, and the mean values of the total nitrogen sub-index to around 0.77. The same transformation was used for both phosphorus and nitrogen to preserve the relationship between phosphorus and nitrogen loads. The relationship between final index values (in terms of the standard ranges) and the actual modelled changes to average annual total loads are shown in Table 35.

Table 35 Relationship between index value and change in annual load.

Sub-index range	Rating	Current average annual total nutrient load
0.75–1.00	Largely unmodified	1.0–1.7 times natural
0.50–0.75	Moderately modified	1.5–3.5 times natural
0.25–0.50	Substantially modified	3.5–12 times natural
0.00–0.25	Severely modified	Greater than 12 times natural

6.4.2 Suspended Sediment Index

Mean annual loads of suspended solids were modelled using the river network sediment budget model developed in collaboration with the agricultural productivity theme of the NLWRA (refer to modelling section below). This is the same model that was used for the bed sediment sub-index of the habitat index, although bedload and suspended load are treated as separate processes of transport through river networks. The model predicts the loading of rivers with suspended sediment derived from hillslope, gully, and riverbank erosion. Allowance is made for deposition of sediment on floodplains and in reservoirs and lakes.

Modelling, rather than measurements, was used to assess suspended sediment loads of rivers because of the scarcity of data in Australia. The most comprehensive data are for background conditions, measured at monthly or weekly intervals. The bulk of the annual load is transported during high intensity storms and floods, which are not well represented by background monitoring but are measured by event-based sampling. Unfortunately, there are very few rivers with event-based monitoring and fewer with records extending beyond five years. Thus, assessment based on measurement would require considerable extrapolation and interpolation to other basins and other reaches. The best way to achieve this is through a physically based conceptual model of the transport processes. The available measurements of river suspended sediment loads were used to calibrate two physically based parameters in the model, and to validate the results across diverse environments. Details are given in the modelling section.

The suspended sediment model predicted mean annual loads (CSS, t/yr) under current land use in catchments and average current conditions of gully extent and bank erosion. The natural benchmark loads used to compare the current loads against assessed erosion of hillslopes under natural land cover (NSS, t/yr) assumed no gullies or degraded riverbanks existed as sediment sources.

The suspended sediment sub-index (SSI) was calculated as follows (Equation 46).

$$SSI = 1 + 0.33\log_{10}(NSS/CSS) \qquad \textbf{Equation 46}$$

The suspended solid index varies from a value of 1, when the current suspended load is unchanged from the natural load, to a value of 0 when the current suspended load is 1000 times the natural load. In a few places the current suspended sediment load was predicted to be more than 1000 times the natural rate, but these values were considered unreliable and were capped at 1000. In some areas, such as parts of Tasmania, and south-west Western Australia, both natural and current suspended sediment loads were predicted to be very low (<100 t/yr). There is considerable uncertainty in the precision of values at such low magnitude and the ratio NSS/CSS becomes unreliable. In these situations the suspended sediment index was set to 1 to reflect the view that loads are so low that suspended sediment is unlikely to have a significant impact on river health. In these areas the total phosphorus and total nitrogen sub-indices were also set to 1, as these values are dependent on the modelled sediment values. A logarithmic scale was used because of the highly skewed frequency distribution of the CSS/NSS ratio. It remains a partly arbitrary scale because there are few data available that define ecosystem sensitivity to precise levels of suspended sediment.

6.4.3 Toxicant data

The National Pollutant Inventory (NPI) compiled by Environment Australia records measurements from across Australia of 30 toxicants in facility discharges. If a facility uses or produces more than a threshold level of a chemical, it must provide details of its emissions to Environment Australia. The NPI has the potential to be very useful but currently has some significant limitations:

- only a few years of data
- data for only 30 toxicants included

- discharge volumes are estimated, not measured
- no record of whether discharge is constant or episodic
- some intensive (e.g. feedlots) and extensive (e.g. grazing, horticulture) agricultural facilities are not required to report.

Even with these shortcomings, the NPI provides a national measure of the potential toxicant hazard to a number of river reaches. NPI data for discharges (to both water and land) containing inorganic and organic toxicants were used to determine a toxicant hazard sub-index. The assumption was that, in time, all discharges to land and off-river water bodies would enter streams. Toxicant hazard for a reach was based solely on the discharges from facilities within the immediate reach sub-catchment. Discharges in the catchment further upstream of the reach being assessed were assumed to have relatively small impact on the reach in question. The limited data available on the effect of point sources on macroinvertebrate communities downstream suggest that effects dissipate over the order of the average reach length (Sloane and Norris, submitted).

The chemicals listed in the NPI have different toxicities. A measure of the hazard associated with each toxicant was calculated by dividing the quantity (kilograms) emitted by the recommended value, measured in µg/L, from the ANZECC Water Quality Guidelines (ANZECC 1999) (see Table 36). Where more than one toxicant was emitted in the sub-catchment, the hazard values were summed. The hazard value for sub-catchments ranged from 0.0004 to 13,069. These values were converted to a toxicant hazard index by establishing the hazard value that should be the threshold for the band divisions (0.75, 0.5, and 0.25). These were hazard values of 1, 10, and 100. The significance of these values was that it required a ten-fold increase in toxicants to move from fair to poor condition, and a 100-fold increase to move from poor to very poor. The following equation (Equation 47) was used to map hazard values onto a hazard index:

$$HI = 0.75 - 0.25(\log_{10}(hv)) \quad \textbf{Equation 47}$$

where *HI* = Hazard index and hv = sum of the hazard values for a reach sub-catchment. Values of *H* less than zero were set to zero; values greater than 1 were set to 1.

Table 36 Guideline values from ANZECC Guidelines (1999)

Substances on NPI Table 1	Guideline value (µg/L)
Acetone	No guideline value
Arsenic and compounds	1.6
Benzene	180
1,3-butadiene (vinyl ethylene)	390
Cadmium and compounds	0.013
Carbon monoxide	Not applicable to water
Chromium (VI) compounds	9
Cobalt and compounds	0.24
Cyanide (inorganic) compounds	1
1,2-dibromoethane	No guideline value
Dichloromethane	3100
2-ethoxyethanol	1900
2-ethoxyethanol acetate	No guideline value
Ethylene glycol (1,2-ethanediol)	330
Fluoride compounds	No guideline value
Glutaraldehyde	No guideline value
Lead and compounds	1.2
Mercury and compounds	0.013
Methanol	No guideline value
Methyl ethyl ketone	No guideline value
Methyl isobutyl ketone	No guideline value
Methyl methacrylate	No guideline value
Nickel carbonyl	0.7
Nickel subsulphide	0.7
Oxides of nitrogen	Not applicable to water
Particulate matter	Not applicable to water
Polycyclic aromatic hydrocarbons	0.3
Sulphur dioxide	Not applicable to water
Sulphuric acid	No guideline value
Tetrachloroethylene	82
Toluene (methylbenzene)	180
Toluene-2,4-diisocyanate	No guideline value
Total nitrogen	Not used
Total phosphorus	Not used
Trichoroethylene	315
Xylenes (individual or mixed isomers)	180

6.5 River nutrient transport modelling methods

6.5.1 Introduction

The river nutrient transport modelling methods used in the Assessment of River Condition (ARC) in NLWRA 1 are applicable for the FARWH. This section describes the basis for nutrient modelling as derived from Norris et al. (2001).

Catchment disturbance, by both agriculture and urban development, leads to increases in nutrient export to the drainage network. These increased nutrient loads affect the river ecosystems, usually in undesirable ways. Assessing these changes in nutrient loadings is therefore an important aspect of assessing river condition, and one that highlights the linkages between a river and its catchment. Assessing nutrient loads in Australian rivers is complicated, whether done primarily using measured data or largely using modelling, because of the complex processes involved in nutrient sourcing and transport, and high temporal variability of river flow. While process modelling of river nutrient transport is usually done in conjunction with detailed daily hydrology modelling, for the purpose of broad-scale assessments of changes in nutrient loadings this effort is not required, and indeed, sufficient data are usually not available. As a part of the National Land and Water Resources Audit's Assessment of River Condition (ARC), the ANNEX model of river nutrient transport was developed to predict current and pre-disturbance nutrient loads in the rivers of the area assessed. Since the NLWRA, further refinements to ANNEX have been made and the model is now available within the SedNet software [www.toolkit.net.au/sednet]. The model includes source terms from detailed spatial modelling of catchment erosion, together with modelled soil nutrient concentrations. In addition, source terms are taken from detailed spatial modelling of the dissolved nutrients in surface and subsurface runoff. The major lost terms in the river budget are deposition of sediment-attached nutrients in reservoirs and on floodplains, and a net loss of nitrogen to the atmosphere because of denitrification. The model outputs are used in the determination of the nutrient and suspended sediment load index of the ARC for each river reach in the area assessed. In addition, the model predictions of current nutrient loads leaving river basins are important, as they are a measure of the nutrient loading of downstream water bodies, including terminal wetlands in inland rivers and estuaries and coastal waters for coastal rivers. They are an indication of the impact of catchment disturbance on both rivers and final receiving waters.

6.5.2 Background

In catchment soils nutrients occur in inorganic and organic forms. Erosion and surface and subsurface runoff carry both sediment-attached and dissolved nutrients from catchment soils and atmospheric deposition to the drainage network. These nutrients are of vital importance to the functioning of river ecosystems, particularly in headwater streams where less nutrients are fixed into the food chain by aquatic plants and algae, and because of the fewer reaches the river has there (and where flow conditions are generally less suitable). The nutrient load to rivers varies naturally between catchments as a result of differences in parent rock type and derived soils, and differences in climate, topography, and vegetation.

In disturbed catchments, however, the loads of nutrients exported from the catchment can rise considerably as a result of increased erosion (such as from tillage of croplands), reduced cover on grazing lands, and wash-off of animal faeces and fertilisers. Urban development also leads to increased nutrient loads because of nutrient-enriched urban runoff and discharges of sewerage effluent. In disturbed catchments the nutrient sources are typically classified as either diffuse (reaching the drainage line through diffuse overbank or subsurface runoff) or point (discharging at a point into the drainage network).

Of the diffuse sources, areas of intensive horticulture (such as market gardening) generally have the highest areal diffuse loading rates, followed by intensive cropping, urban areas, improved grazing, unimproved grazing, and finally forest areas (Young et al*.* 1996). However, there is considerable variation in the loading rates from individual land use types because of differences in soil type, climate, and slope. Importantly, the way a land use is managed can have a large influence on the levels of nutrient export. For example, levels of fertiliser use and the means and timing of its application, the manner and timing of tillage, stocking rates, and ground cover are some of the many management factors that influence nutrient export. The proximity of the disturbed land to a drainage line – or the connectivity of the landscape – is critical in determining what proportion of the nutrient loads associated with surface runoff will reach a stream. The riparian zone along drainage lines provides an important buffer between the hillslope and the stream, and is a zone where particulate material (and associated nutrients) is deposited and where dissolved nutrients can be used by plants, in this way preventing delivery to the stream.

Because point sources, such as sewage discharges, have a single entry-point to the river network and are often characterised by small flow volumes but high concentrations, they often have considerable local impact. However, further downstream they are usually less important. They also typically have greater impact under low flow conditions, when their contribution is a larger proportion of the total load. In contrast, diffuse sources usually dominate total loads in the medium to long term, with the majority of the diffuse load being delivered during large flood events. Typically 90 percent or more of the diffuse load can move in less than 10 percent of the time (e.g. McKee et al*.* 2000).

Nutrients delivered to headwater streams are processed as they are transported downstream. These processes include physical abrasion of organic and inorganic sediment, and biochemical transformations between inorganic and organic forms. The cycling between organic and inorganic forms, superimposed on downstream transport, sometimes leads riverine nutrient transport to be described as ‘nutrient spiralling’. The downstream transport of nutrients is complex: the changing lateral inputs of nutrients (including two-way exchanges with the floodplain in lowland rivers), the changing energy gradient, the changing water and sediment chemistry, and changing biota, all affect nutrient spiralling. These dynamic factors determine where nutrients are stored, how far they will be transported, and ultimately where the final export from the basin will occur.

Building a budget for nitrogen along river systems is generally more difficult than building one for phosphorus because of nitrogen exchanges between the water and the atmosphere. Nitrogen gas (N_2) dissolves in the water, and nitrate (NO_3^-) can be denitrified by bacteria to produce atmospheric nitrogen. In contrast, there are no exchanges of phosphorus between the water and the atmosphere, and all phosphorus stays in fluvial transport or fluvial stores, or is bound up in riverine biota. Some of the phosphorus bound in plants and animals can be lost from the catchment system through natural or anthropogenic causes, such as waterbird migration or harvesting of aquatic weeds. These events, however, are generally very small components of a river's nutrient balance. Larger, and far more important, is the deposition of sediment and associated nutrients on the floodplain. While floodplain sediments can be re-entrained (and hence considered a long-term store), floodplains are generally depositional environments that slowly aggrade, and most of the sediment-bound nutrient load deposited on floodplains can be considered as lost from the river system.

Increases in nutrient loadings can affect riverine ecosystems and the ecosystems of downstream receiving waters in both direct and indirect ways. The most common impact is an increase in the abundance of nuisance plants and algae – including cyanobacteria. Blooms of cyanobacteria produce toxins that have ecological impacts and make the water unsafe for humans and stock. Scums produced by blooms are both unsightly and give off offensive odours. Excessive growth of attached algae alters the habitat for invertebrates and fish, and leads to shifts in food web structure. Abundant filamentous algae can clog water filtration systems. Overabundance of aquatic plants can obstruct waterways, thus affecting fish migration, and plants alter in-stream habitat conditions for most biota by changing hydraulic and light conditions. Indirect effects occur when algae or plant populations die and are decomposed by bacteria. The high biological oxygen demand this creates depletes dissolved oxygen and may lead to massive fish kills.

Nutrients in river ecosystems are continually cycling through different biogeochemical forms, not all of which are available for nuisance plant and algae growth. The most bioavailable form of phosphorus is orthophosphate, and the most bioavailable forms of nitrogen are nitrate and ammonia. Bioavailable phosphorus exists in both the dissolved form and attached to sediments. The proportion of sediment-bound phosphorus that can be considered bioavailable is poorly resolved, and certainly the desorption of phosphorus of sediment particles is affected by sediment mineralogy and water quality conditions. Anoxic conditions are particularly conducive to phosphorus release from sediments. Given that the availability of sediment-bound nutrients depends on local sediment and water conditions, choosing measures of biologically available nutrients for a broad-scale assessment is difficult. Harris (1994) suggests that all phosphorus entering a catchment from diffuse sources will be available for nuisance plants and algae, given enough time, whereas most phosphorus in sewage discharges may be immediately available for uptake, but only for a relatively short distance downstream. With increasing distance, a fraction of the dissolved phosphorus in sewerage effluent will adsorb to the sediment particles in the flow. Because of the difficulty in defining objective measures of bioavailable nutrients that apply to all rivers, most broad-scale assessments therefore use measures of total nutrient loads and concentrations, based on measurements of both dissolved and sediment-attached nutrient forms.

6.5.3 Modelling nutrient export and river transport

Because of the great variation in nutrient load generation across a catchment and between catchments, the complexity of nutrient delivery from the land to the river (including riparian processes), and the continual downstream changes in nutrient transport, transformation, and storage, detailed modelling of nutrient export and river transport is restricted to relatively small catchments.

Even in these catchments, large amounts of data are required to parameterise models that capture the complexities of the processes. These are seldom to hand, and at least some new data are commonly sought and collected – often to enable modelling studies in which reasonable model calibration is needed to ensure some reliability in a model's predictions. Many detailed models of sediment and nutrient export and transport exist, and while they differ in many regards, most include detailed modelling of rainfall–runoff processes and river flow routing at a daily or sub-daily time-step. Because of the high temporal variability in flow, detailed modelling of the hydrology is necessary to obtain reliable predictions of the temporal and spatial changes in nutrient concentrations along river systems. This approach to modelling nutrient transport was not possible for the NLWRA because insufficient data existed to allow such models to be developed and calibrated across the area assessed. In any case, such detailed modelling in unnecessary to provide broad-scale assessments of the spatial variations in natural and current average nutrient loadings.

Because long-term average nutrient loads are a function of total flow volume (dominated by floods), detailed modelling of hydrologic processes and variability is not required. An existing model for predicting long-term average nutrient loads without detailed hydrologic modelling is the Catchment Management Support System (CMSS) (Davis and Farley, 1997). CMSS predicts nutrient exports by classifying land parcels according to land use and other attributes, and assigning areal export rates to each land class. Where export data for the catchment being investigated do not exist, the export rates are typically based on values quoted in the literature for catchments judged to be similar. CMSS adds nutrient loads up through the river network and reduces loads in transport using an empirical 'assimilation' routine that represents biological nutrient uptake and sedimentation (parameterised by the hydraulic residence time and a rate constant for each river network link). Although CMSS could have been applied for the NLWRA, the detailed modelling in NLWRA theme 5 of erosion and sediment delivery, and of soil–plant–atmosphere nutrient balances and the associated losses of dissolved nutrients to waterways, presented an opportunity to develop a new model that used as inputs data from detailed spatial and process modelling of landscape nutrient processes. Hence, while the approach adopted for the NLWRA does not include detailed modelling of temporal hydrologic variability, it does capture the detailed spatial patterns of nutrient sources across the landscape, and capture the gross spatial patterns in hydrology.

6.5.4 The river nutrient model

A model was developed that sums the nutrient sources to each link of a river network, and accumulates the consequent nutrient loads through the network. The model explicitly considers the input and transport of both sediment-attached nutrients and dissolved nutrients. It assumes that the sediment-attached nutrient load is associated with the clay fraction of the sediment, which is transported entirely in suspension. The exchange of phosphorus between the suspended sediment and dissolved forms is also modelled, as is the loss of nitrogen to the atmosphere by denitrification. The river nutrient model is embedded in a river sediment model (NLWRA 2001a) because it requires the predictions of the suspended sediment load in each network link. The nutrient sources represented in the ANNEX model are listed below.

Nutrient Sources

The model includes the following nutrient source terms to each network link:

- sediment-attached nutrients from hillslope erosion
- sediment-attached nutrients from gully erosion
- sediment-attached nutrients from river channel bank erosion
- dissolved nutrients in surface runoff and sub-surface drainage
- point source nutrient discharges.

All of these have dimensions of $M\,T^{-1}$ (mass per unit time) and all are assessed on an average annual basis.

A full description of the model and the datasets required to parameterise the model for updated assessments is given in Wilkinson et al. (2004).

River Nutrient Model Calibration

Several components of the river nutrient model were calibrated separately in NLWRA 1. First, the hydrology regionalisations can be considered as a calibration against observed flows. Second, the sediment delivery through the river network was calibrated on the basis of comparisons with measured sediment loads. This calibration was achieved by adjusting the hillslope sediment delivery ratio and the particle size used to model sediment settling for floodplain deposition. These calibrations of the sediment model affect the nutrient budgeting, both by affecting the sediment-attached nutrient loads reaching the river and by affecting the nutrient load lost to the floodplain.

The nutrient model was separately calibrated by comparing the predicted loads to measured nutrient loads for a number of rivers across the country. Because the sediment-attached nutrients inputs were fixed by the sediment model calibration, the main calibration of the nutrient model involved scaling the dissolved nutrients inputs. Insufficient data were available to allow meaningful calibrations basin-by-basin, and so adjustments to the dissolved input loads were made at a national scale to match measured loads. Because the measured loads are generally (though not always) based on average low flow concentrations and mean annual flow volumes, it is likely that they substantially underestimate the true loads. The nutrient model was therefore not calibrated to exactly match the measured loads on average, but was adjusted to best match those loads based on long-term flow-weighted concentration data. This means that compared to the full available dataset of measured loads the model over-predicts total phosphorus loads by 1.8 times and total nitrogen loads by 2.7 times.

The performance of the model in predicting loads is indicated by comparisons of the modelled average annual nutrient loads to the measured average annual nutrient loads (Figure 17). The measured load data used for comparison were taken from the following sources:

- 83 Victorian sites from the Victorian Department of Natural Resources and Environment data warehouse web site [http://www.vicwaterdata.net]. These are average low flow nutrient concentrations and mean annual flow volumes

- 4 sites in Western Australian with 10 years of fortnightly concentration and flow data (Bormans, *pers. comm.*)

- data for phosphorus loads for 3 sites on the Murrumbidgee River in NSW from NSW Department of Land and Water Conservation. Load estimates were based on only one year of data

- data for nitrogen and phosphorus loads at one site on the Fitzroy river in Queensland from Queensland Environment Protection Agency. Load estimates were based on 5 years of data

- data for phosphorus loads for the three main rivers draining into the Peel–Harvey Inlet in Western Australia from Birch (1982)

- data for phosphorus and nitrogen loads for the Richmond River in NSW from Mckee et al*.* (2000)

- data for phosphorus loads for the Kalgan River in Western Australia from Bott (1993)

- data for phosphorus loads for the Darling River at Bourke in NSW from GHD (1992)
- data for phosphorus loads for the South Pine River in Queensland from Cosser (1989)
- data for phosphorus and nitrogen loads for the South Johnstone River in Queensland from Furnas et al*.* (1995).

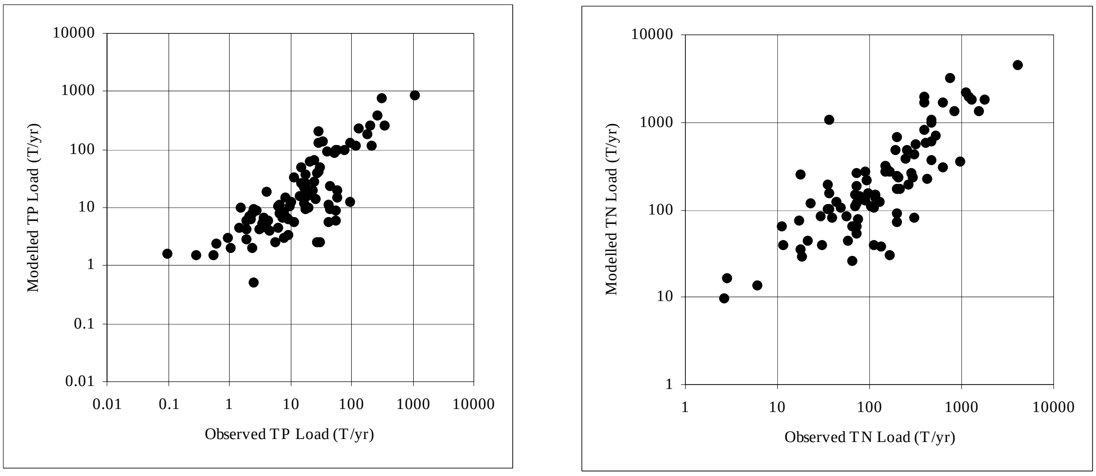

Figure 17 Comparison of modelled to observed average annual nutrient loads (t/yr).

Figure 17 shows reasonable agreement between the model and observations over four orders of magnitude. However, at any scale there is considerable scatter, particularly in the nitrogen loads. Because the largest determinant of total load is flow volume or the drainage area over which the load is generated, a more rigorous measure of model performance is a comparison of nutrient export rates (kg/ha/yr) (Figure 18).

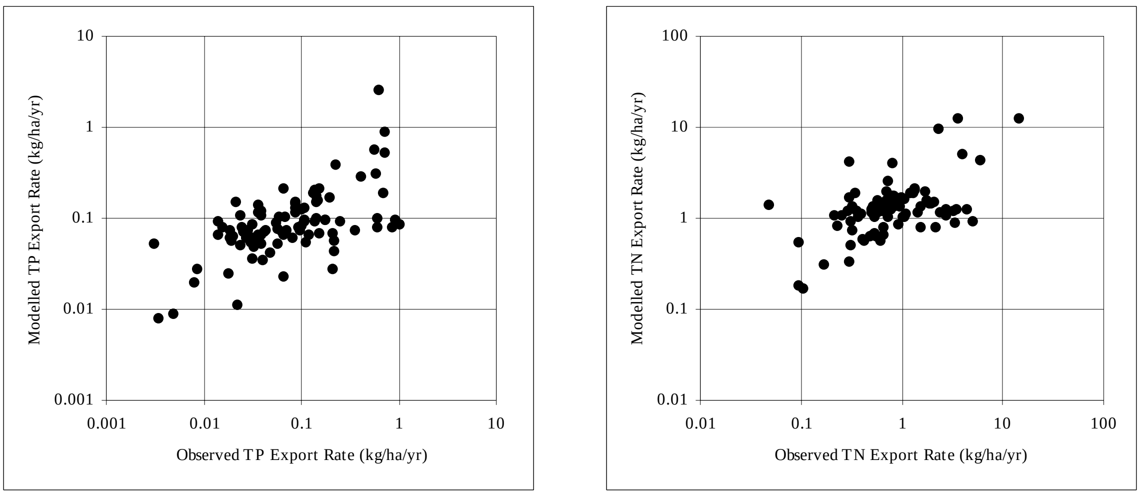

Figure 18 Comparisons of modelled to observed average annual nutrient export rates (kg/ha/yr).

Figure 18 shows greater scatter than the comparison of loads; however, after removing the large variability in catchment size, the model still provides reasonable predictions of the measured range of nutrient export rates. The general over-prediction of the model resulting from the calibration procedure referred to above can be seen in Figs 4.17 and 4.18. In spite of the scatter there is no apparent bias in the mode performance – that is, both low and high loads and export rates appear to be predicted equally well.

A third useful comparison is between the observed and modelled ratio of dissolved nutrient load to the total nutrient load (Figure 19). This was only possible for phosphorus, as none of the observed nitrogen load data discriminated between dissolved and sediment-attached nitrogen. For phosphorus, only the data from 69 of the Victorian sites provided dissolved and sediment-attached phosphorus loads. Across these sites the average ratio of dissolved phosphorus to total phosphorus is 0.25. For these same sites the average of the modelled ratio is 0.30. While this seems a reasonable comparison, it is clear from the data in Figure 19 that, on a site-by-site basis, the model does not predict this ratio well. There is reasonable agreement between observations and the model for about one-third of these sites; other sites are either substantially over- or under-predicted. One possible explanation of this poor agreement is that, given that mean annual flow and fine sediment load are both reasonably well calibrated, the assumption of a constant *K*s value across all rivers is unrealistic. *K*s controls the balance between the phosphorus loads carried in dissolved form and the load attached to the sediments. If data were available to model the spatial variations in *K*s, this might improve the calibration of the dissolved-to-total phosphorus ratio. The few data available for *K*s do show significant variation between sites (see Webster et al*.* 2001); however, the reasons for these variations are unclear and so do not allow systematic estimation of *K*s values for different rivers. A second possible explanation for the poor agreement of dissolved P to TP ratios is that the spatial patterns in the dissolved input term are incorrect, or that the efficiency of the delivery of this load from the landscape is spatially variable. In the calibration, a constant 'delivery ratio' was used for this dissolved input term.

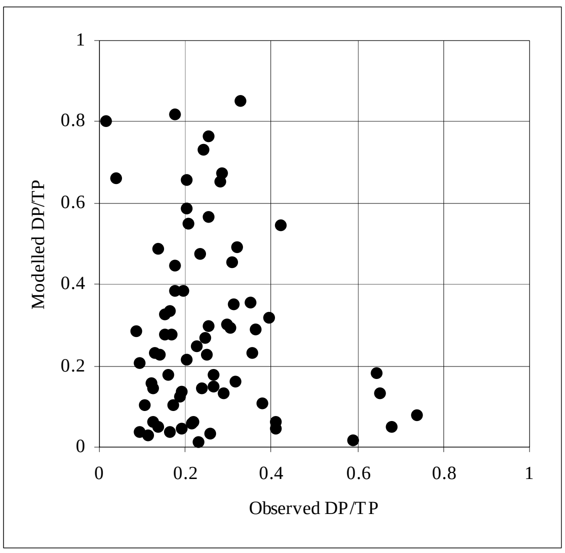

Figure 19 Comparison of modelled to observed ratios of dissolved to total phosphorus.

A fourth useful comparison is of the TN:TP ratios for the observed and modelled data (Figure 20). The average TN:TP ratio for the observed data is about 13.3, while the average modelled ratio for the sites where observations were available is 15.6. The relationships between TN and TP across the sites is similar for observed and the modelled data, although the observations show more scatter, with several observations plotting below the 'Redfield ratio' of TN:TP=6.8 by weight (Redfield 1958). The Redfield ratio is the ratio in which algae use nitrogen and phosphorus.

The similar TN:TP relationship between the observed and modelled data (Figure 20) is a good indicator of model performance. Although the modelled total phosphorus loads for the calibration sites are dominated by sediment phosphorus from erosion, the modelled total nitrogen loads for the calibration sites are dominated by dissolved nitrogen from surface and sub-surface runoff (mean ratio of dissolved N to sediment N of 0.76). The TN:TP ratios are therefore a result not only of the modelled spatial pattern of erosion, but also the modelled spatial pattern of dissolved nitrogen loss from landscape to rivers. The similar TN:TP relationship in the observed and modelled data therefore supports the reality of these spatial patterns. With respect to nitrogen ratios, it should be noted that while the average ratio of dissolved nitrogen to total nitrogen is 0.76 for the calibration sites, across all the rivers modelled this ratio has a mean of 0.60. At an AWRC basin scale, this ratio ranges from values as high as 0.97 in the Shannon River basin in south-west Western Australia, to as low as 0.27 in the Lake Torrens basin in South Australia and 0.28 in the Burdekin basin in Queensland.

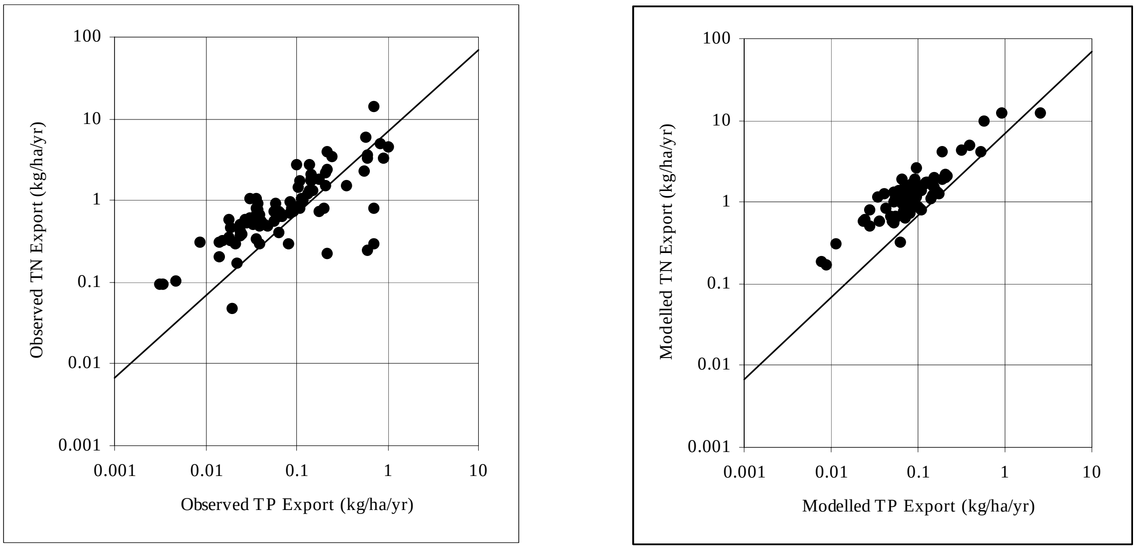

Figure 20 TN versus TP for the observed data and modelled data. The lines represent the Redfield ratio of TN:TP = 6.8 by weight.

Appendix A
River Sediment Budget Methods for the National Land and Water Resources Audit

Introduction

This document describes methods used in the NLWRA 1 to represent the erosion of sediment from riverbanks and the propagation of gully, hillslope, and riverbank sourced sediment through a river network, as reported in Prosser et al. (2001b) and Norris et al. (2001). Essentially it describes methods for constructing sediment budgets through river networks. The budgets include storage of sediment on floodplains, in the bed of the river, and in reservoirs. Calculations are made of the sediment output from each river reach and the contribution of sediment to the coast, or any other receiving body, from all sub-catchments. The budgets treat two types of sediment: suspended sediment and bedload. Spatial modelling was used to define river networks and their sub-catchments, import required data, implement the model, and compile the results. The model is referred to as the *SedNet* model: the *Sed*iment River *Net*work Model.

For this project, suspended sediment is characterised as fine-textured sediment carried at relatively uniform concentration through the water column during large flows. Its transport is generally limited by the supply of sediment to rivers rather than by the sediment transport capacity (Olive and Walker 1982, Williams 1989). The main process for net deposition of suspended sediment is overbank deposition on floodplains (e.g. Walling et al*.* 1992). The amount of deposition depends upon the residence time of water on the floodplain and the sediment concentration of flood flows. The velocities of suspended material within channels are relatively high so we can assume that its residence time is low, that there is negligible transient deposition of suspended sediment, and that steady-state conditions have been reached since the time when European settlement increased the supply of sediment. Suspended sediment is sourced from surface wash erosion of hillslopes, gully erosion, and riverbank erosion. The sediment budget is reported as mean annual values for either the current land use or for pre-European native vegetation cover.

Bedload is sediment transported in greatest proportions near the bed of a river. It may be transported by rolling, saltation, or, for short periods of time, by suspension. Transport occurs during periods of high flow, and over distances of hundreds to thousands of meters (Nicholas et al*.* 1995). Residence times of coarse sediment in river networks are relatively long, so there are transient depositions on the bed as the sediment works its way through the river network. Additionally, an increase in sediment supply from accelerated post-European erosion can cause the total supply of sediment in historical times to exceed the capacity of a river reach to transport sediment downstream. The excess sediment will be stored on the bed, and the river will have aggraded over historical times (Trimble 1981, Meade 1982). There has been a significant increase in supply of sand and fine gravel to rivers in historical times and deposition of this bedload has formed sand slugs: extensive, flat sheets of sand deposited over previously diverse benthic habitat (Nicholas et al*.* 1995, Rutherfurd 1996). The bedload budget aims to predict the formation of these sand slugs.

The basic unit of calculation for sediment transport in rivers is the link. A link in a river network is the stretch of river between any two stream junctions (or nodes). Each link has an internal sub-catchment, from which sediment is delivered by hillslope and gully erosion processes. The internal catchment area is the catchment area added to the link between its upper and lower limits (Figure 21). For the purpose of the model, the internal catchment area of first-order streams is the entire catchment area of the river link. Additional sediment is supplied from bank erosion along the link and from any tributaries to the link.

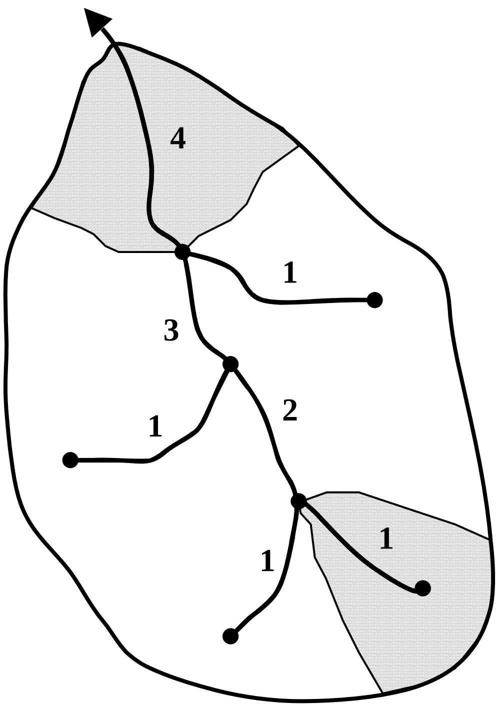

Figure 21 A river network showing links and nodes. The numbers are the Shreve magnitude of each link, and the shaded areas depict the internal catchment areas of a magnitude one and magnitude four link.

Sediment Budget for Bedload

A sediment mass budget for bedload was calculated for each river link (*x*) in a network. The mass balance was evaluated at the outlet of each link, with the aim of defining those links subject to net deposition (those where the historical supply of bedload exceeds the sediment transport capacity). The total load supplied at any time to the outlet of the link (from local gully and riverbank erosion and upstream tributaries) is compared with the sediment transport capacity at that point. If the load exceeds capacity, the excess is deposited and the yield to the link immediately downstream equals the sediment transport capacity. If the loading to the outlet is less than the sediment transport capacity there is no net deposition and the yield downstream equals the loading to the outlet.

Two forms of the coarse sediment model have been implemented: a steady-state bed material budget for post-European conditions and a more complex transient model that accounts for the long residence time of bedload within rivers (long in relation to the time since European settlement when erosion increased; Wilkinson et al. 2004). Wilkinson et al. (2006a) found that the transient model predicts the spatial location and extent of bed material accumulation within a river basin more accurately than does the steady-state model, which indicates that bed material takes decades to move through river networks. At the coarser modelling resolution used in the NLWRA, the difference in performance between the steady-state and transient models was much less important (Norris et al. 2001).

When the bedload model is conceptualised as limited by sediment transport capacity, it essentially models the river network as a set of valves of variable size. Upstream increases in supply of sediment to the river network will only be felt downstream if propagation of the increase is not prevented by low sediment transport capacity in any individual link. If a small-capacity valve limits downstream yield, then increased supply upstream is absorbed at that point through local deposition. Thus downstream propagation of increased bedload supply is relatively ineffective and is limited by the link of lowest sediment transport capacity along the route to the basin outlet. Increased source erosion will only result in increased delivery to the mouth for paths and sediment sizes that are supply limited. This may explain the ineffective delivery of sand and coarser sediment to the Australian coast and its insensitivity to upstream source changes (e.g. Roy 1977).

Sediment Budget for Suspended Load

Transient storage since erosion increased in historical times is not significant for suspended sediment. The residence time of suspended sediment carried within channels is considerably less than the period of increased erosion. The main areas of net accumulation of suspended sediment are floodplains and reservoirs, where sediment residence times are of the order of thousands of years. There is little widespread information available on the temporal dynamics of suspended sediment supply to rivers, so a steady-state budget was constructed to predict mean annual suspended sediment loads. This level of information is adequate for regional planning of land use change; it is also sufficient for catchment restoration and for identifying the spatial patterns of sediment sources, yields, and potential impacts.

The suspended sediment loads of Australian rivers, and rivers in general, are supply limited (Olive and Walker 1982, Williams 1989). That is, rivers have a very high capacity to transport suspended sediment and sediment yields are limited by the amount of sediment delivered to the streams. Consequently, if sediment delivery increases, sediment yields increase proportionally. Deposition is still a significant process, however. Plots of suspended sediment yield data against catchment area illustrate this (Figure 22). These plots show a reduction in sediment yield per unit area with increasing catchment area. Such a decrease in specific sediment yield is not satisfactorily explained by decreases in erosion intensity in large catchments (Richards 1993). The more likely explanation is deposition of sediment within river systems, and the most significant depositional areas are floodplains and reservoirs. Hence this, and description of the sediment sources, form the basis for our suspended sediment budget, which is aimed at predicting regional patterns of sediment loads.

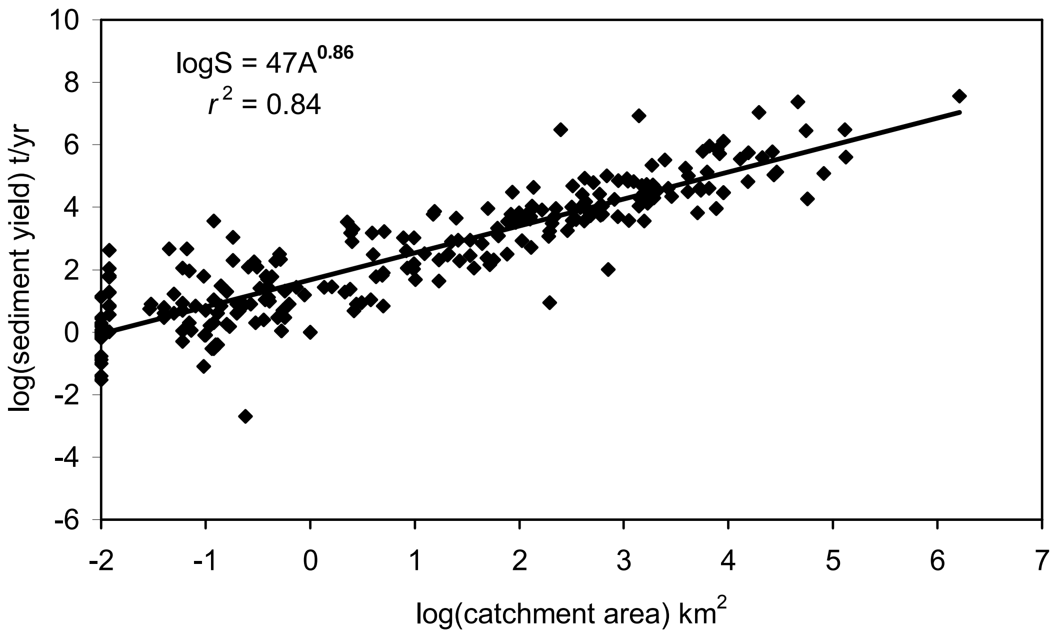

Figure 22 Sediment yields data for Australian catchments compiled by Wasson (1994). Only data for areas >0.01 km^2 are shown.

Suspended sediment is supplied to a river link from four sources: riverbank erosion, gully erosion, hillslope erosion, and tributary suspended sediment yield. Approximately half of the total sediment supplied from gully and riverbank erosion contributes to the sediment budget, the rest to the bed material budget. Sediment deposition on floodplains and in reservoirs is also accounted for in the suspended sediment budget. The areas of catchment contributing suspended sediment to a point in the river network can be determined using the SedNet model (Wilkinson et al. 2004) since the model accounts for spatial variation in sediment sources and also sediment sinks in the river network. This is important information for management actions aimed at improving water quality downstream.

Suspended Sediment Budget Under Natural Conditions

Across the diverse environments of Australia, strong differences occur naturally in suspended sediment loads. To assess the extent to which the current sediment loads reflect the natural ones, and to what extent they reflect accelerated sediment supply, we need to estimate the natural sediment loads. This is a fairly speculative process because there is limited knowledge of natural conditions, and no sediment load data other than for a few small catchments which remain relatively undisturbed.

The natural rate of hillslope erosion was estimated by interpolating to cleared lands the predicted rate under current conditions in reserves of native vegetation (Lu et al*.* 2001). The delivery of this sediment to streams was calculated using Equation 48, assuming an unchanged hillslope sediment delivery ratio. The natural rate of gully erosion is negligible compared to current rates and is not included in the analysis. In NLWRA the natural rate of sediment supply from riverbank erosion was also neglected. Subsequently, this has been revised and the current model accounts for the riverbank erosion that occurs even under healthy riparian vegetation. Reservoir deposition was also omitted from the natural sediment budget.

$$HF_X = SDR\sum_{j=1}^{j=n} A_j \qquad \textbf{Equation 4 48}$$

where A_j is the mean annual soil erosion rate predicted by USLE for each cell *j* in an internal catchment containing *n* cells.

Model Implementation and Validation

Most parameters required for the model have been found from empirical relationships among extensive datasets, for example hydrology, river width, and bank erosion. There are, however, three model parameters that cannot be set a priori. These are the sediment delivery ratio from hillslopes (Equation 48), sediment settling velocity for bedload (Equation 49), and sediment settling velocity for floodplain deposition. These three parameters can all be interpreted in terms of the physical processes of sediment transport and field observations. There are, however, no datasets to show spatial patterns of the variables. They were set as global constants across the assessment area, with the exception of hillslope sediment delivery ratio for which the assessment area was split into two geographic regions as explained below.

$$STC_x = \frac{86\, S_x^{1.3} \sum Q_x^{1.4}}{\omega\, w_x^{0.4}} \qquad \textbf{Equation 4 49}$$

where $\Sigma Q_x^{1.4}$ represents time integration of the flow record ($ML^{1.4}$/yr), and ω is the settling velocity of the bedload particles (m/s).

The aim of the bedload sediment budget was to predict the extent of sand slugs derived from massive increases in supply of sand and gravel to rivers from gully erosion and accelerated bank erosion. Observations of the sand slug deposits show that they are composed of coarse sand and fine gravel

(Rutherfurd 1996) so a mean particle size of 2 mm was used in Equation 49 to determine sediment transport capacity. Conversion of particle size to sediment settling velocity followed the procedures described by Richards (1982; p. 77–79). Other values were tested to examine the sensitivity of the choice made. It was found that an order of magnitude change in particle size was required to significantly alter patterns of deposition in river networks. Predictions based upon finer particle sizes resulted in the focus of deposition being pushed further downstream to reaches of very low sediment transport capacity. The opposite occurred using a coarser mean particle size. As well as matching field observations, the mean particle size used produced patterns of deposition similar to those mapped in the Glenelg Rivers, and conforming to our own field knowledge of the Murrumbidgee River. No comprehensive mapping of sand slugs has been published in Australia outside of the Glenelg River catchment (Rutherfurd 1999) and further testing of the bedload model across diverse rivers is needed.

The sediment delivery ratio from hillslopes and the particle settling velocity for floodplain deposition influence patterns of suspended sediment supply to rivers, suspended sediment yields, and deposition. Changes to the sediment delivery ratio change the total mass of sediment supplied to rivers and the mean annual loads and mean annual deposition rate of each river link rise proportionally. Changes to the sediment settling velocity influence the intensity of deposition through the river network. An increase in settling velocity increases the proportion of suspended load that is deposited on floodplains and decreases the sediment yield from the basin. In terms of the traditional plot of suspended sediment yield (log *S*) against catchment area (log *A*; Figure 22), reducing the hillslope sediment delivery ratio maintains the pattern of values but lowers all values on the plot. Reducing the sediment settling velocity increases the gradient of the regression line through the points and thus increases the sediment yield from the river basin. Changing the hillslope sediment delivery ratio also changes the ratio of sediment supplied from hillslopes to that supplied from channel processes (gully and bank erosion).

Five criteria were used to evaluate appropriate values for hillslope sediment delivery ratio and sediment settling velocity. These were:

1) calibration against observed suspended sediment loads of rivers

2) shape of regressions of log *S* against log *A*

3) mean annual rate of floodplain deposition

4) mean annual rate of reservoir infilling, and

5) ratio of hillslope to channel sediment sources.

Measurements of mean annual suspended sediment loads are scarce in Australia, but some data are available to calibrate the model and evaluate the bounds of suspended sediment yield. For this project we focussed on getting a good geographic spread of river load data, and focussed on larger catchments where errors might become more apparent (since the sediment budget is essentially built from the bottom up by summing all the source terms, allowing for losses along the way).

Wasson (1994) has produced regional plots of log *S* against log *A* for Australia using data available at the time. The data span a huge range in scales from 10^{-6} to 10^{6} km^2, and linear regression of the following form (Equation 50) was fitted to the data.

$$\log S = a \log A + b \qquad \textbf{Equation 50}$$

Only the data above 50 km^2 are relevant for comparison against the river model results. A feature of the data is the common observation that the value of *a* is usually less than 1, but generally greater than 0.8, implying storage of sediment as catchment scale grows (Walling 1983). The best regional relationship is for the southern uplands of Australia where the exponent = 0.94 (Wasson 1994), but even there the uncertainty on that value is high when only river scale data are considered.

Equation 51 predicts the mean annual rate of deposition on floodplains. It is conceivable that the suspended sediment model could predict the correct catchment sediment yields by using a gross overestimate of sediment supply and absorbing the error in floodplain deposition. This could occur by not applying a hillslope sediment delivery ratio for example. Such an error would result in high values of floodplain deposition. Floodplains accumulate sediment at rates of the order of 1–2 mm/yr or less, producing 1–2 m of aggradation in 1000 yr (e.g. Walling et al*.* 1992) and constraining model parameterisation.

$$D_x = \frac{Q_f}{Q}\left(TIF_x\right)\left(1 - \exp\left(-\left(vA_f / Q_f\right)\right)\right) \qquad \textbf{Equation 51}$$

where A_f is floodplain area ($A_f = wx$) and Q_f is floodplain discharge ($Q_f = Q - Q_b$) where Q_b is bankfull discharge.

A similar argument can be made for reservoir deposition. Large reservoirs are effective sediment traps, usually trapping >90 percent of incoming sediment load (Outhet 1991), but if the predicted loadings to reservoirs are overestimated then they will be predicted to fill with sediment within several decades. Observations of sediment deposits in Australian reservoirs suggest that they will be viable for centuries to thousands of years under current accumulation rates (Davis et al*.* 1997).

Radionuclides from atmospheric fallout have been used to partition suspended sediment in rivers between surface-derived materials, relatively rich in attached radionuclides, from subsurface-derived materials, relatively deficient in radionuclides (Wallbrink and Murray 1993). The most obvious surface sources of sediment are sheetwash and rill erosion of hillslopes. Gully and riverbank erosion supply predominantly sub-surface material. Several catchments in SE Australia have sediments dominated by sub-surface radionuclide signatures (values up to 95 percent), suggesting that gully and bank erosion are the primary sediment sources (Olley et al*.* 1993, Wallbrink et al*.* 1998, Prosser et al*.* 2001a).

The best match against the criteria given above was to use a sediment settling velocity of 1 x 10^{-6} m/s across the assessment area and a hillslope sediment delivery ratio of 0.05 or 0.1 depending upon region. A sediment delivery ratio of 0.1 was used for the Burdekin, Fitzroy, and north Queensland regions, and a value of 0.05 was used elsewhere. The settling velocity is equivalent to particles of 0.033 mm diameter, which is in the silt range and well within the bounds of materials found on floodplains. The higher value for sediment delivery ratio in Queensland was needed to produce the observed sediment yields from the Johnstone, Burdekin, and Fitzroy Rivers. A higher delivery ratio for those areas may be feasible given the intense long duration tropical storms that enhance sediment transport on hillslopes. A plot of observed and predicted river sediment yields is shown in Figure 23. The match is very good considering the limited parameterisation and vast geographical spread of the predictions. Figure 23 does not just represent catchment size, as there are strong differences in area specific sediment yield between the catchments shown.

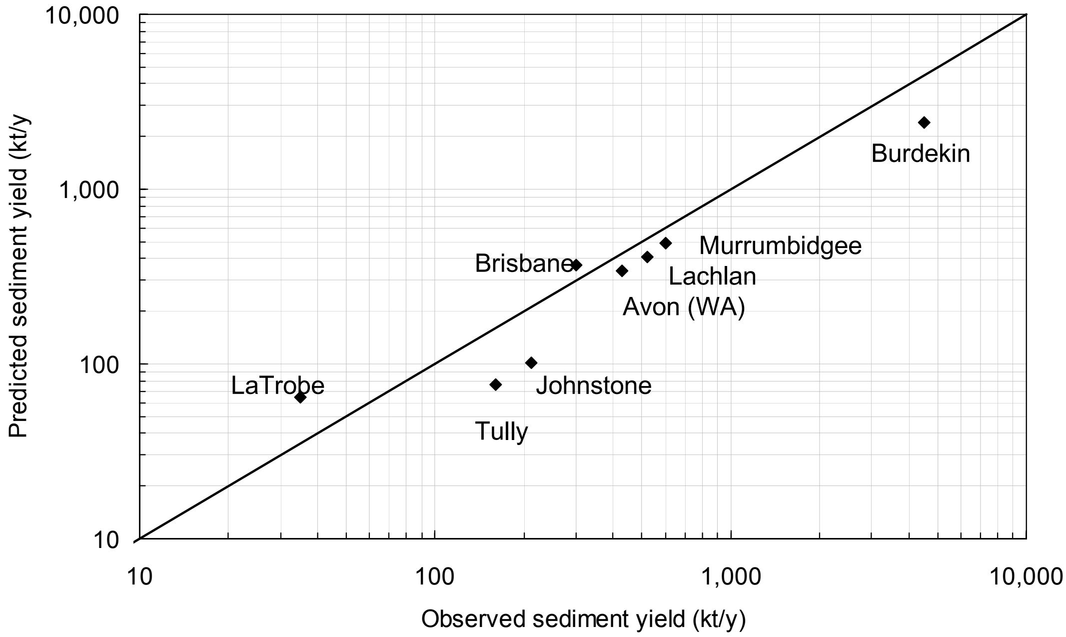

Figure 23 Observed versus predicted sediment loads for rivers spread across the assessment area.

Applying a sediment settling velocity of 1 x 10^{-6} across the assessment area produced a value of *a* from the model of 1.05 to 0.79. For the southern uplands, where monitoring data are best, the predicted value of *a* is 0.87 compared to that of *a* = 0.94 from Wasson (1994). These values are very close given the order of magnitude range of sediment yield in both measurement and prediction for any given area. Over 90 percent of river links were predicted to have floodplain deposition of less than 2 mm/yr and 98 percent had less than 3 mm/yr of deposition. Over 99 percent of the large dams were predicted to have capacities to store greater than 100 yr of sediment. Without the application of a low sediment delivery ratio, hillslopes dominate the supply of sediment to streams. Applying a value of 0.05 to south-eastern Australia resulted in >80 percent of sediment being delivered from gully and riverbank erosion, close to the independent estimates from radionuclide analysis. Other features of the model, which suggest that the results produce realistic values, include that near the mouths of catchments suspended sediment makes up approximately 90 percent of the load despite gully and bank erosion contributing evenly to bedload and suspended load.

Sand slugs have been mapped in the Glenelg River (Rutherfurd 1996). The model predicts this to be one the most impacted rivers for bedload, as found in the field. The coarseness of the river network and sediment supply terms in the model means that it will not predict the detailed distribution of slugs in the catchment (calculations are averaged over a link, for example). The broad location and extent of impacted reaches is relatively well predicted, however (Figure 24). Further comparison of the predicted location and extent of bed material accumulation against observations is contained in Wilkinson et al. (2006a).

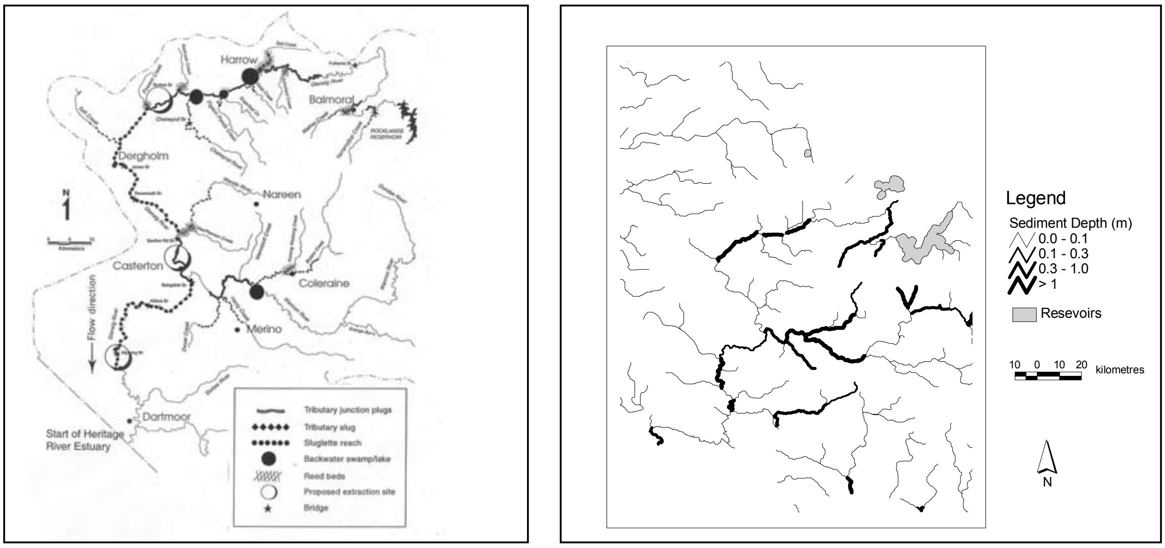

Figure 24 Observed versus predicted locations of sand slugs in the Glenelg River catchment of western Victoria. Observed map reproduced from Rutherfurd (1999).

Recent SedNet model developments

The SedNet model, including ANNEX nutrient budgets, is available as software at www.toolkit.net.au/sednet; guidelines for data preparation and model use are available in a user guide (Wilkinson et al. 2004). This provides the opportunity to re-run SedNet models of river catchments using higher resolution data than was available in the NLWRA, and also taking advantage of recent model developments, which include:

- Several hydrological parameters are required to model sediment nutrient budgets. The original NLWRA hydrology modeling (Prosser et al. 2001b, Norris et al. 2001) has been updated (Wilkinson et al. 2004), improving the capacity of the model to account for spatial variability in runoff (Wilkinson et al. 2006b).

- River bank erosion is now predicted as a function of stream power (DeRose et al. 2003, 2005), accounting for the presence of erodable soil in the riparian zone (Wilkinson et al. 2005) and giving improved spatial patterns across large river basins.

- The transient bed material model has been demonstrated to provide improved prediction of the spatial location and extent of bed material accumulation within river networks (Wilkinson et al. 2006a). The transient model is now implemented in the SedNet model software (Wilkinson et al. 2004).

Appendix B

Definitions

See http://www.fisheries.org/publications/bookpdf/aquaticintro for an exhaustive list of definitions of terms related to rivers and lakes.

Term	Definition
Bankfull flow	Maximum stream flow that can be accommodated within the channel without overtopping the banks and spreading onto the floodplain. Generally the level associated with 2- or 3-year stream flow events
Basin	A topographic area of a watershed or geological land area that slopes toward a common centre or depression where all surface sub-surface water drains
Connectivity	Water exchanged between the river channel and the associated floodplain, or the unimpeded passage between upstream and downstream sections of a river
Daily flow	Total discharge from midnight to midnight for a continuous recording
Flow duration	A measure of the overall difference between current and natural flow duration curves. This index was based on a measure developed by Young et al. (2000) to assess the overall hydrological deviation of a dam release option from the transparent dam flow
Geomorphology	Study of the origin of landforms, the processes that form them, and their material composition
Link	In drainage networks, stream reach of a particular order, or a stream at its origin
Stream macroinvertebrate fauna (or macroinvertebrates)	The larger invertebrates that live in streams; includes insect larvae, crustaceans, snails, and worms
Mean annual flow	Sum of total daily discharges for a year divided by the number of days in a year. Also, the total annual discharge divided by the number of seconds in a year
Mean annual rainfall	The arithmetic average of the total rainfall at a site, or an area, over several years
Median daily flow	The midpoint of daily flows for a defined period
Median overbank (excess) flow	The midpoint of flows exceeding bankfull over a defined period
Node (in river network)	Refers to the ending points of a line that is used in Geographic Information Systems as a reference point along a stream
Reach	Portion of a stream that extends downstream from the confluence of two streams or rivers to the next encountered confluence and where the stream power changes by 95 percent

Riparian vegetation	Vegetation growing on or near the banks of a stream or other water body that is more dependent on water than vegetation found further upslope
River network	The channels within a river basin
Seasonal amplitude	The change in amplitude of the seasonal pattern of monthly flows. It is defined as the average of two current:natural ratios, first, that of the highest monthly flows (QHMc:QHMn), and second, that of the lowest monthly flows (QLMc:QLMn) based on calendar month means
Seasonal period	The change in seasonal timing of flows. It can be defined as the difference from 1 of one-twelfth of the sum of the absolute values of the differences between current and natural of first, the numerical values of the months with the highest mean monthly flows (HM), and second, the numerical values of the months with the lowest mean monthly flows (LM)
Sediment	Fragmented material from weathered rocks and organic material that is suspended in, transported by, and eventually deposited by water
Stream power	Energy or ability of a stream to move substratum and scour stream banks that is based on gravity, slope, discharge, and water velocity
Suspended sediment	Sediments that are carried in suspension in the water column by turbulence and water velocity or by Brownian movement so that they are transported for a long time without settling to the bottom

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