Outcomes of the NRHP Urban Sub-Program

Report of a workshop held at Environment Australia, Canberra, 21st February 2001

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1 INTRODUCTION

The National River Health Program (NRHP) was established in 1993,with the aim of improving the management of Australia's rivers and floodplains for their long-term health and ecological sustainability. The Urban Sub-Program of the NRHP was established in 1994 to produce tools for measuring the health of urban streams and estuaries. The Sub-Program was delivered via a portfolio of eight research projects that were developed to meet the research priorities identified for urban streams and estuaries and complement other NRHP research projects being conducted on rural streams. The Water Services Association of Australia (WSAA) was contracted by Land & Water Australia (then LWRRDC) to manage the Sub-Program. The research projects commenced in 1996 and were completed in 2000.

WSAA convened a workshop, held on the 21st February 2001, to explore the knowledge gained and lessons learnt from the Sub-Program and to consider future directions for research and management of waterways (streams and estuaries) in urban areas. The workshop was attended by key researchers associated with the eight projects and representatives from agencies with an interest in the outcomes of the program (Appendix 1). This report is a summary of the findings and recommendations that emerged from the workshop.

A short synopsis of the eight Urban Sub-Program projects is presented in chapters 2-9, along with suggestions for 'value-adding' to the work already undertaken. The research projects include:

1. Decision support system for management of urban streams
2. River inVertebrate Prediction and Classification System (RIVPACS) for urban streams
3. Diatom Prediction and Classification System (DIPACS) for urban streams
4. Sediment chemistry-macroinvertebrate fauna relationships in urban streams
5. Classification of estuaries
6. Literature review of ecological health assessments in estuaries
7. Estuarine health assessment using benthic macrofauna
8. Estuarine eutrophication models

Future research directions that emerged from general discussion about the Sub-Program and individual projects are discussed in chapter 10.

2 BASIC DECISION SUPPORT SYSTEM FOR MANAGEMENT OF URBAN STREAMS

2.1 Project Synopsis

The aim of this project was to develop a classification system that would assist the management of streams in urban areas (Anderson 1999a). The resultant classification system is software based, and uses biologically important physical attributes to classify waterways in terms of asset value, potential for rehabilitation, physical and environmental condition, and the key constraints limiting rehabilitation. The project has been reported through four publications (Anderson 1999a, b, c and d), describing:

1. The development of the classification system for urban streams;
2. A software manual that accompanies the computer-based classification system;
3. The results of pilot studies undertaken for streams in Brisbane; and
4. A demonstration game – Let's Fix Urban Streams.

The classification system and computer model are based on a sampling strategy that included:

- Definition of homogeneous stream sections;
- Condition summaries based on length of stream;
- A variety of condition ratings and attributes;
- Single and aggregate ratings and formulae.

Five types of condition rating and data summaries for each of the homogeneous stream sections are stored in a simple database system. The ratings and summaries are described by Anderson (1999a and d) and are restated here. They include:

- Habitat condition and values ratings;
- Habitat classification indices;
- Depth parameters;
- Sediment particle size parameters;
- Waterway classification indices.

Habitat condition and value ratings expressed as percentages of the pristine condition (100% representing original condition and no loss of function; 0% representing total loss of original condition and function). The condition and value ratings include:

- Riparian vegetation condition rating based on width and community structure;
- An environs rating that refers to the overall condition of the stream side, including valley flat and floodplain areas, based on the extent of clearing and modification;
- Aquatic habitat rating based on percentage of canopy cover and the extent and diversity of bank and instream habitat;
- Bank stability based on active (rather than historic) processes. Bank stability is measured as the proportion of the stream section that is regarded as stable;
- Bed and bar stability, which is the proportion of the stream bed that is stable;
- Aquatic vegetation rating based on the proportion of the emerged bed that covered by emergent, submergent or floating vegetation;
- Conservation value rating based on the known occurrence of threatened species or their habitats;

- Overall condition rating, which is the average of the percentage rating for the various components listed above. The riparian vegetation and aquatic habitat ratings are given double weighting.

Habitat classification indices include a channel habitat diversity index and the riparian zone width. The channel diversity index is based on the proportion of the length of stream section classified as pool, riffle, run, glide, cascade etc. The riparian width of remnant native vegetation is included to provide information for assessing the rehabilitation potential of the section.

Depth parameters include pool depth and riffle/run depth. Sediment particle size parameters include pool sediment size, riffle/run sediment size, riffle sediment size and cascade sediment size.

Water classification indices include a modification index, an average buffer naturalness index and a buffer to bankfull ratio. The modification index is constructed using the extent of modification to five section components, including the bed, banks and three buffer zones: shoreline, middle and upper zones. The five scores are then combined to provide a five-digit index representing the modification to each of the five section components. The average naturalness index is another way of summarising modification scores for the five section components using percentage naturalness scores to reflect their relative importance to the stream section ecosystem. The buffer to bankfull ratio is used to classify stream sections on the basis of the remnant buffer width. The ratio is a useful measure for assessing a section's suitability for rehabilitation.

Indices that describe the potential for rehabilitation in each of the stream sections have also been included in the package. Indices include the pollution source index and suitability for natural design index. The pollution source index summarises the potential water pollution sources occurring in each section, or in areas immediately upstream, that may affect rehabilitation efforts in the section. The suitability for design index provides a subjective assessment of whether natural design concepts may be applied to the section.

Parameters of particular usefulness for assessing stream section condition and opportunities for rehabilitation were found to include:

- Buffer zone naturalness (upper, middle and streamside zones);
- Pollution source index;
- Suitability for design (e.g. pull out concrete channels). Constraints and opportunities are used for a constraints index;
- Riparian zone width;
- Pool depth.
- Modification index based on upper, middle, streamside, bank and bed ratings.

Operation of the package as a demonstration game is described by Anderson (1999d).

The Queensland Department of Natural Resources has become the biggest user of the package to date, especially for the development of Water Allocation Management Plans (WAMP). The package is also being trialed in some NSW catchments.

2.2 Opportunities for future development

The package may be applied to streams in both rural and urban areas to assist State of the Environment reporting, stressed rivers classifications and potentially impact assessment. The package would benefit from more use and evaluation via case studies. This would enable improvements to the ‘user friendly’ nature of the package. The package could also be simplified or modified to run as a stand alone (e.g. on CD-ROM) that would be a learning tool for students. Marketing of the package could also be undertaken to promote and encourage its uptake at a national level.

Ecological data could inform the model by providing biological constraints, as rehabilitation efforts may fail to restore target species, biological communities or processes if the limiting factors are not addressed. There is potential to add biological ratings but this has not been done yet; the modification index has been included to address this.

The opportunity also exists to link the model to the rehabilitation framework developed by Rutherfurd *et al*. (1999) for the rivers and riparian lands program coordinated by Land and Water Australia.

3 RIVPACS (RIVER INVERTEBRATE PREDICTION AND CLASSIFICATION SYSTEM) FOR URBAN STREAMS

3.1 Project Synopsis

The objective of the project was to create an urban AUSRIVAS model, consistent with the existing procedures established by the Monitoring River Health Initiative (Breen *et al*. 2000).

The construction of AUSRIVAS models are broadly based on the method of Wright *et al.* (1984), are summarised in Breen *et al*. (2000) and restated here. A series of minimally impaired sites are selected as reference sites. Reference sites are then classified into groups based on their invertebrate communities using cluster analysis. The environmental variables that best explain the separation of the reference site groups are identified using statistical methods such as discriminant function analysis. Environmental variables that may be influenced by human activity are generally discarded from the analysis. Any number of test sites may be selected and environmental variables that discriminate between reference site groups are measured and macroinvertebrates are sampled. The environmental variables are used to match test sites with reference sites and the probability of reference group membership estimated. The taxa that should occur at a test site in the absence of impairment (at whatever taxonomic level chosen e.g. family) are predicted. The difference between the invertebrate community structure observed (O) at a test site and that expected (E) can be used as an indicator of impact. Test sites equivalent to reference condition are expected to have an O/E ratio close to 1. Impacted sites are expected to have an O/E ratio significantly less than one.

The project used macroinvertebrate and environmental data compiled by existing CRC for Freshwater Ecology (CRCFE), Environment Protection Authority Victoria and Melbourne Water programs, in order to obtain a sufficient number of reference sites for modelling. Sampling sites in rural areas adjacent to Melbourne were used as references for urban (test) sites. While rural streams are subject to anthropogenic disturbances they were considered to represent reasonable health and rehabilitation targets for urban streams. The combined data set provided 31 reference and 31 test sites for evaluation.

The different sample sorting methods adopted by each program meant that some assessment and rationalisation was required before a consolidated data set was constructed for AUSRIVAS modelling. There were systematic differences between the laboratory and field-sorted data sets based on species level identification of invertebrates. Laboratory-sorted samples containing more taxa and a higher proportion of small and mobile species. Field sorted samples had fewer taxa, but an increased proportion of large and immobile species. While there were differences in the method pairs, ordinations analysis showed that lab-sorted and field-sorted produced similar community composition patterns (i.e. reference site groupings), suggesting that the difference between groups were larger than the differences between method pairs (Figure 1). The model developed with family data had a lower misclassification error than the species model.

O/E scores were negatively correlated with BOD and catchment imperviousness (Figure 2), which are considered to be indicators of water pollution and urban density. These relationships became even clearer when O/E was plotted against the artificial variable BODIMP, which is a product of BOD and catchment imperviousness (Breen e*t al*. 2000). Some sites had good O/E scores even though located in areas with high catchment imperviousness. These sites were found to have good bed substrate and intact native riparian vegetation that offers protection for macroinvertebrates. Two other outliers were of interest. One site had a high O/E score and a high BODIMP; the site was located downstream from an urban lake that acted as a water pollution control pond. Another site had a low O/E score and a low BODIMP; this site was located near industry (communities at this site where more impacted than expected apparently by factors not strongly correlated with BOD or impervious area, for example by heavy metals).

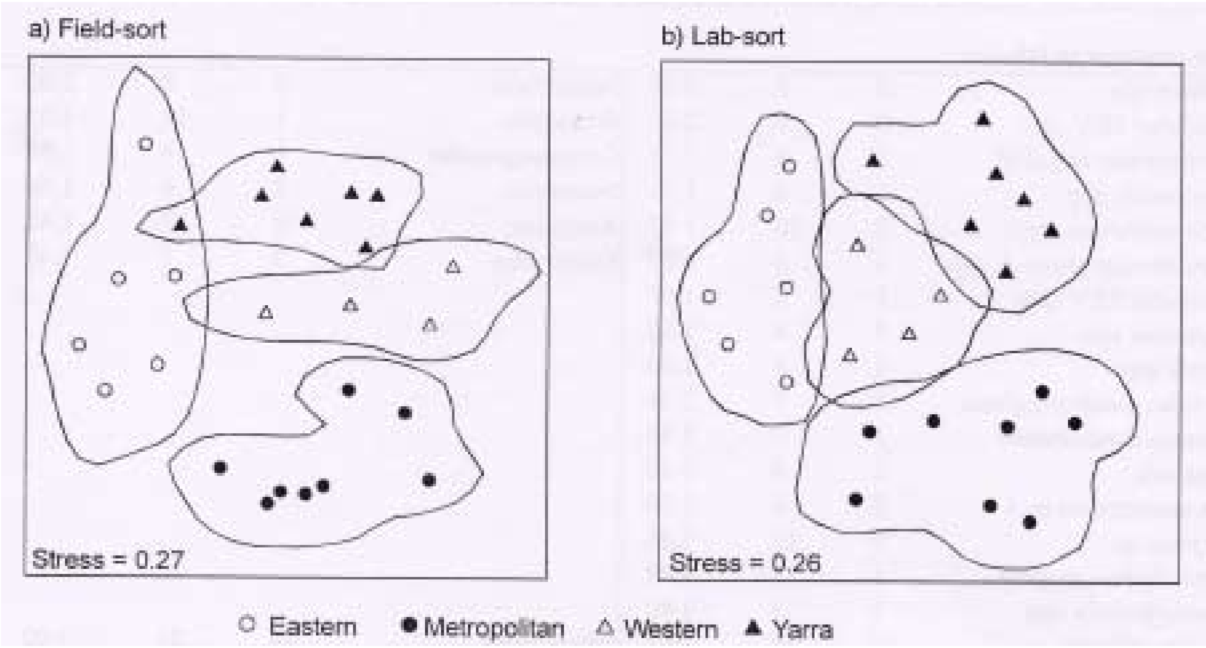

Figure 1: MDS ordinations of 25 sites used in the comparison of processing methods (from Breen *et al.* 2000).

There was no significant difference in the O/E vs BODIMP relationship for both family and species models. There were significant differences between the site ranks based on O/E and BODIMP, with the largest variation occurring in rankings for impacted sites. However there was general agreement between the rank ordering of the sites in good condition. Breen *et al.* (2000) suggested that the variability measured at heavily impacted sites may be due to their being in a constant state of recovery from a range of disturbances.

The project has demonstrated that it is possible to create urban AUSRIVAS models. The models developed using species and family level invertebrate data provided similar patterns for the grouping of reference sites and in terms of relationships with the key environmental variables BOD and catchment imperviousness. However, the results for the urban family model and the EPA Victoria family model were not significantly different. This suggests that the urban model was no more sensitive to urban impacts than the larger-scale EPA Victoria model. As the use of broader, regional models overcomes the difficulty in identifying reference sites for urban models, there is little advantage in the urban models over the regional model. However this study was based on existing data sets and reference sites were limited. With purpose designed sampling programs urban models may prove useful for surveillance monitoring.

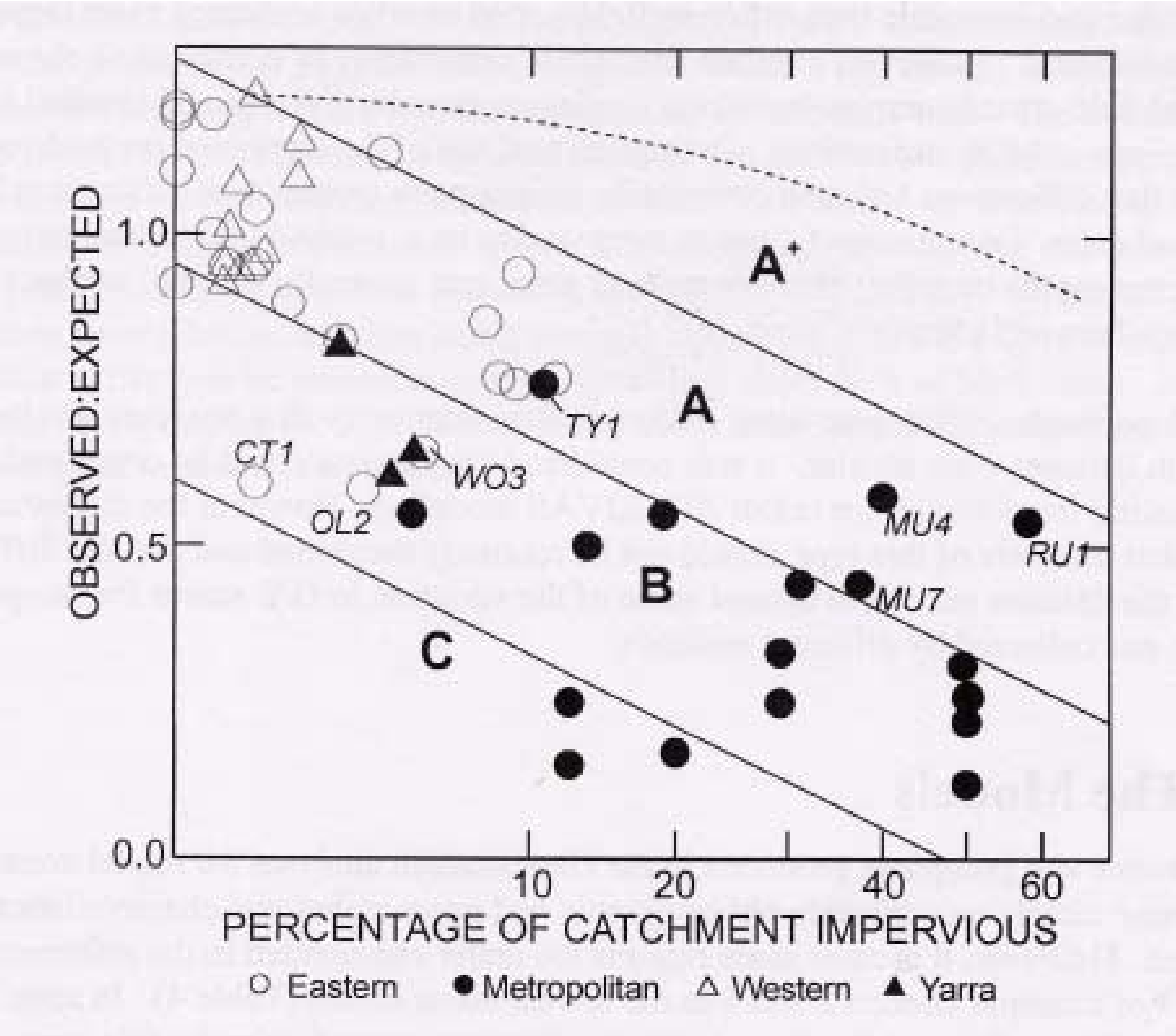

Figure 2: Scatter plots showing the relationship between urban model O/E ratios and BOD, catchment imperviousness and BODIMP (from Breen et al. **2000**)

An alternative approach for applying larger-scale AUSRIVAS models (based on pristine reference sites) to urban settings is the development of a relationship between community composition and catchment imperviousness. Catchment imperviousness is a general measure of catchment development and can be used as a covariate against which other ecological indicators can be set. It is assumed any increase in catchment imperviousness will result in some impairment of community composition. Test site O/E scores could then be evaluated against the O/E-catchment imperviousness relationship; catchment imperviousness becomes a 'handicapping' variable for the O/E scores (Figure 3). For a certain level of catchment imperviousness different sites tend to have varying O/E ratios. This pattern potentially allows the banding of O/E ratios against catchment imperviousness. Test sites can then be evaluated to determine if their O/E ratios are higher or lower than expected given a particular level of urban development. This approach avoids the problem of selection and availability of reference sites to construct urban models.

Conclusions

- The development of urban AUSRIVAS models is possible, however the difficulties in finding sufficient appropriate reference sites around major cities suggests that the use of regional models is more appropriate;
- Large-scale regional models can be adapted for urban settings by using catchment imperviousness as a handicapping variable for O/E scores.

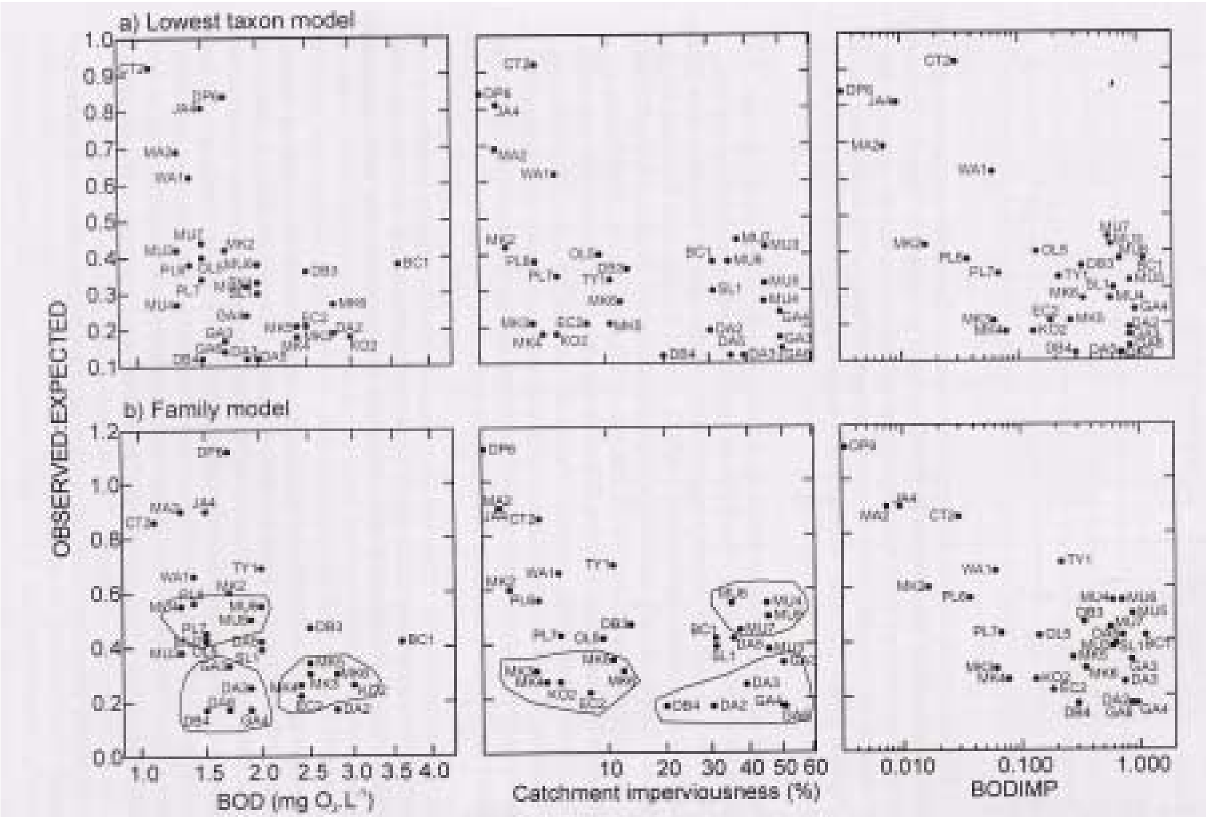

Figure 3: **Relationship between O/E ratios from the family model and catchment imperviousness for a selection of test and reference sites (from Breen *et al.* 2000).**

3.2 Further Development

The urban models developed for this project were found to provide a similar level of prediction as the regional model developed by the Victorian EPA. The urban models could be improved by the inclusion of more reference sites in the physiographic groups present in the Melbourne region. However the strong correlation between community composition and degree of catchment urbanisation among sparsely urbanised hinterland sites (Walsh *et al.* 2001), suggests that the use of such sites as reference sites in an 'urban' AUSRIVAS model is logically flawed.

The problem of appropriate reference condition may be avoided by developing models using a range of sites from severely degraded to pristine. These so-called "dirty water" models could then be used to predict changes in community composition resulting from improvement or degradation of key environmental variables. Dirty water models may also be linked to biogeochemical models. These models are not direct alternatives to AUSRIVAS models, but would be used as decision support tools for testing management options.

All the factors that drive invertebrate community structure should be considered when assessing stream health in urban areas, as this will provide more information for managers than AUSRIVAS scores alone (i.e. simply stating the distance from reference). As different stressors can have the same impact on invertebrate community structure, it is important that all potential stressors are identified. Variables such as BOD and catchment imperviousness should be included in programs that monitor stream condition around Melbourne.

The current project investigated the relationship between invertebrate populations and environmental variables such as BOD and catchment imperviousness. The inclusion of additional sites allows us to assess their potential for rehabilitation and identify the variables that should be managed in order to achieve rehabilitation objectives. Such an approach could also be used to measure recovery after intervention, for example by looking at the trajectory of recovery in relation to other urban sites.

4 DIATOM PREDICTION AND CLASSIFICATION SYSTEM (DIPACS) FOR URBAN STREAMS

4.1 Project Synopsis

The Swan-Canning estuary and nearby wetlands have a history of eutrophication and algal blooms in recent years (John 2000a). Settlement and development along the river system has resulted in pronounced changes to landuse and drainage and native vegetation has largely been replaced with alien species that decompose rapidly. Changes such as these have increased the organic and nutrient load entering local waterways. The region has a Mediterranean climate, with winter nutrient inputs carried by rainfall-runoff, followed by warm, still-water conditions conducive to the growth of algae. Diatom blooms are common in the lower Swan estuary, while blooms of dinoflagellates have been recorded in the upper Swan River. Blue-green algal blooms have been recorded in the Canning River (*Anabaena* spp.) and nearby wetlands (*Anabaena* sp. and *Microcystis* sp.). In February 2000, a freshwater blue-green algal bloom (*Microcystis aeroginosa flos aquae*) was recorded in the Swan River for the first time. The bloom coincided with heavy rainfall and decreased salinity in the estuary to levels tolerated by *Microcystis*.

The objectives of the project (John 2000a and b) were to:

- Develop a classification system for urban streams and drains based on water quality and stream condition variables;
- Develop a predictive model for biomonitoring urban streams based on diatom community composition;
- Produce a guide to aid the collection and identification of diatom species.

The classification system and model were developed using diatom and environmental data (up to 30 variables) collected from approximately 200 sites across 31 sub-catchments in rural, semi-rural and urban areas of the Swan-Canning estuary (John 1999a). While identifying potential impacted sites (labelled as ‘monitoring’ sites by John (1999a)) was relatively easy, identifying reference sites proved to be difficult, requiring analysis if historical water quality data and catchment condition, and site visits. Diatom communities were sampled using JJ Periphytometers (a chamber containing glass slides that is immersed in water to allow colonisation) in the summer of 1996, spring-summer of 1997 and autumn-winter of 1997. Scrapings from the JJ Periphytometers were used for Chl-a analysis and to make permanent slides of diatoms for future records.

All the sites were classified on the basis on the basis of seven environmental variables (alkalinity, NH3, TKN, TN, SRP, TP and colour) using the agglomerative hierarchical fusion method with flexible unweighted pair group mean average (UPGMA). Reference and impacted sites were separated by superimposing the UPGMA clusters on principal component analysis (PCA) results. Reference and impacted sites were found to be separated on the basis of nutrients (TKN, TN and TP). A third, intermediate group of sites, was noted in addition to reference and impacted sites, and was separated on the basis of electrical conductivity (brackish water sites).

Classification of the sites was also undertaken by comparing the diatom community structure at each site. Of the 200 diatom species recorded from all sites, 57 species were selected for the classification on the basis of their relative frequency (John 1999a). Multi-dimensional scaling

(MDS) was used to provide site ordinations based on diatom assemblages. The ordination resulted in the separation of three groups that were useful for setting management priorities: Pristine (reference), Intermediate and Impacted (Figure 4). The reference sites (mainly semi-rural sites that were least impacted by urbanisation) were all in the upper catchment areas (Figure 5) and cluster analysis and discriminant function analysis indicated that the diatom communities were separated on the basis of native vegetation and depth to water table. Impacted sites were generally in the lower catchment areas and their diatom communities were separated on the basis of alkalinity, electrical conductivity, groundwater salinity, catchment land use, riparian damage and colour. The separation between Reference and Impacted sites was not clear cut, and a group of sites that were a mixture of reference and impacted formed the Intermediate group.

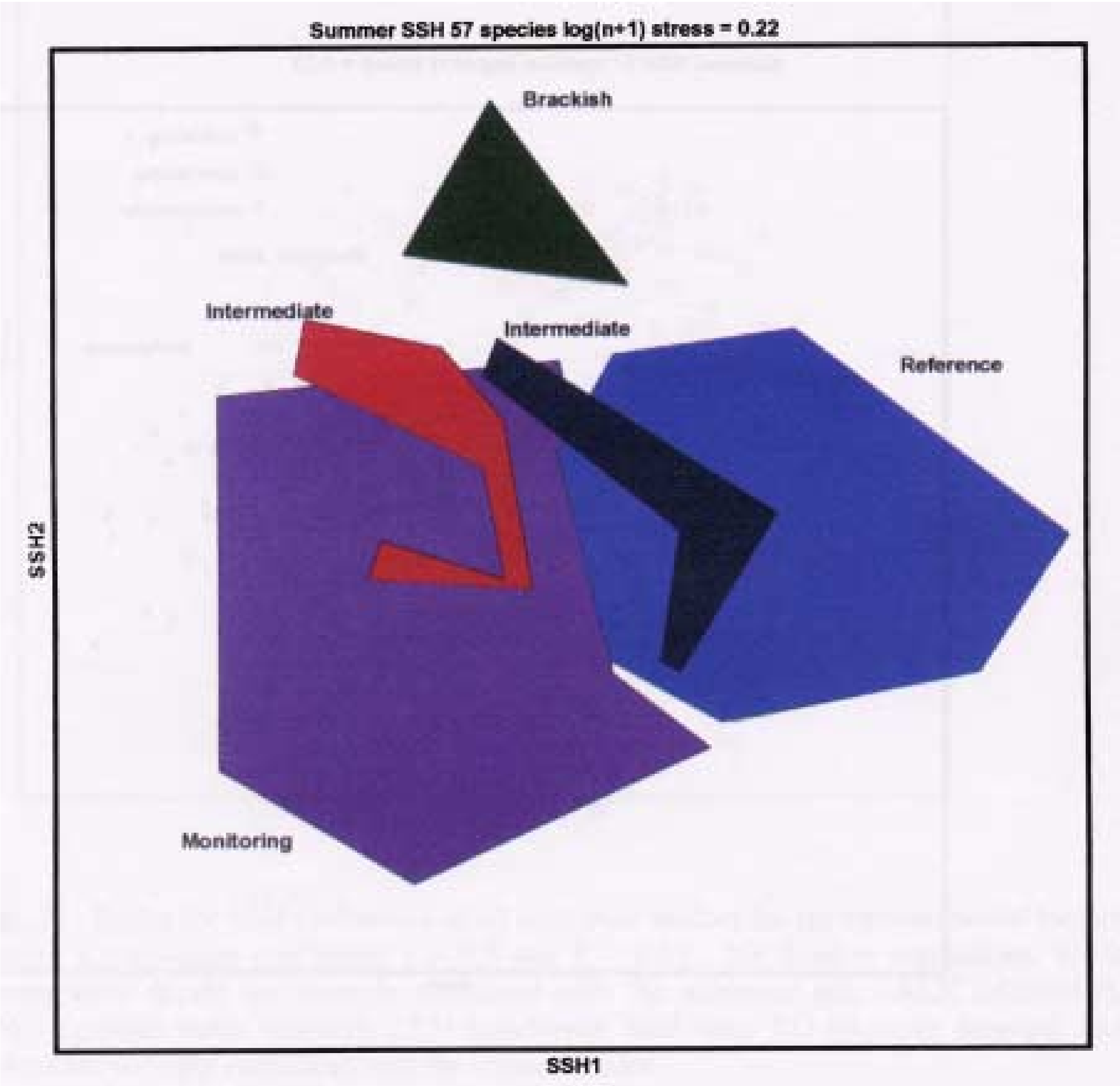

Figure 4: Ordination plot of 146 sites and 57 diatom species with UPGMA classification superimposed on the sites (from John 2000a).

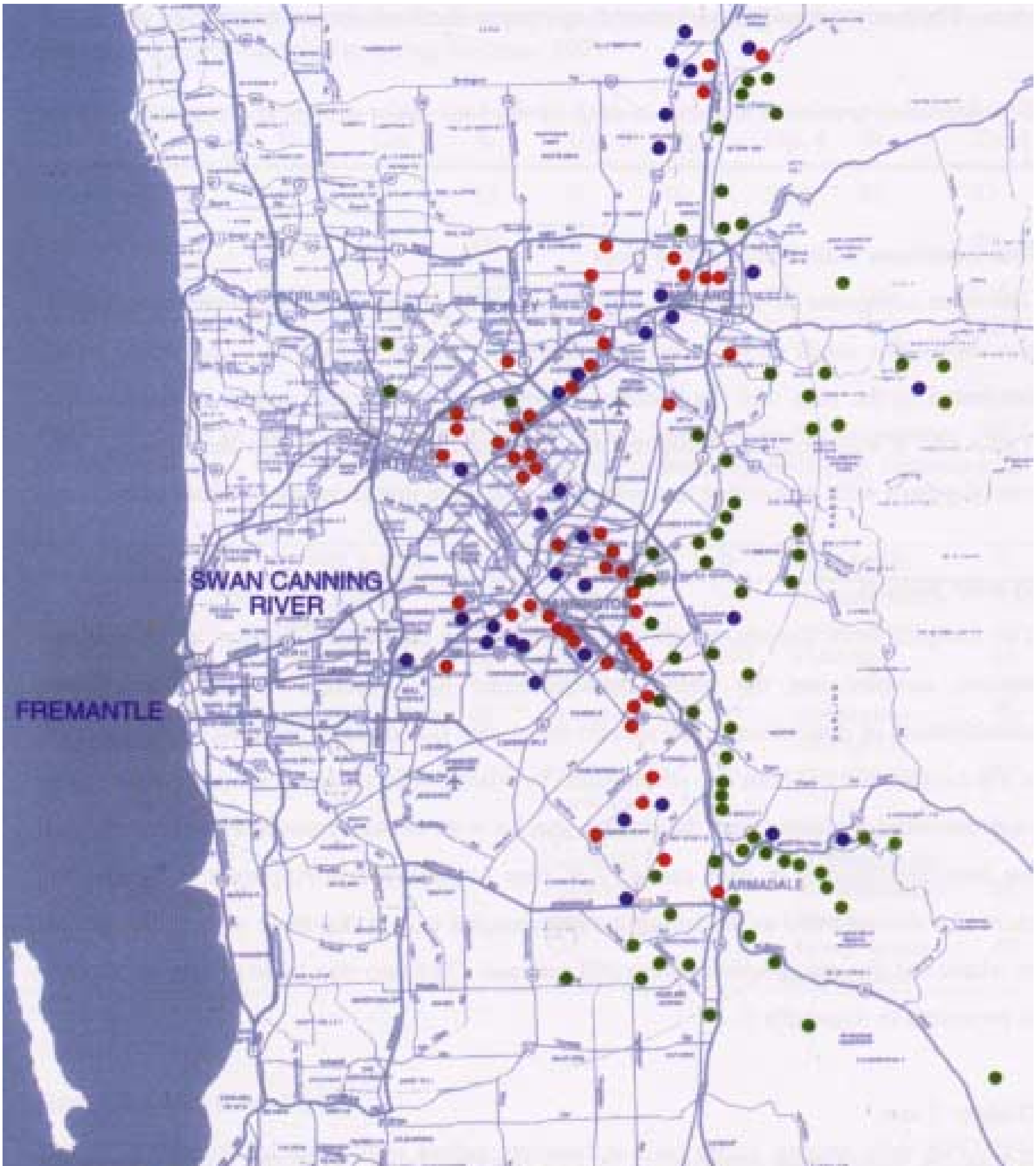

Figure 5: Final classification of sites in the Swan-Canning River (from John 2000a). Red = Impacted; Blue = Intermediate; Green = Reference.

Both the methods (based on environmental variables and diatom assemblages) showed similar classification patterns. This suggests that diatom assemblages respond to key environmental variables and are useful biomonitors of stream condition.

Two-way tables, which illustrate the distribution of diatom species by site and site groups by species, can be used to assess the health of a new ‘test’ site. Data on diatom assemblages at a test site may be entered into the ordinations and classified as ‘reference’ or ‘impacted’. Thus two-way tables may be used to provide a prediction of stream condition.

A diatom index was also developed to test whether a site was ‘reference’ or ‘impacted. The group classification obtained using diatom data was used in a discriminant function analysis of environmental variables. The index was derived from the following relationship:

$$\frac{(\Sigma Pr*Ar) +1}{(\Sigma Pu*Au) +1}$$

where A = abundance of reference (r) or unexpected (u) species
P = probability of reference (r) or unexpected (u) species

The index was tested successfully in the winter of 1997.

Another important and successful part of the project was the compilation of a comprehensive guide to diatoms (John 2000b) which has been widely recognised and sought after since its completion.

4.2 Further Development

The classification system and diatom index is transferable to streams in temperate areas such as Melbourne, but the models must be based on local species and environmental variables. The majority of diatom species (approximately 90%) found in areas with a Mediterranean climate are cosmopolitan, although Tasmania has a slightly higher proportion of endemic species. A sampling protocol would be helpful to those who intend to develop models for local conditions.

As some of the macroinvertebrate groups found in the eastern states are missing from Western Australian streams, the cosmopolitan nature of diatom species may be considered as beneficial when developing national sampling protocols.

Costs for sampling and analysis of diatoms is similar to that of invertebrates (sampling costs are expected to be lower but identification costs higher). If diatoms assemblages were adopted as biomonitors by agencies, this would create a ‘critical mass’ to drive sampling and identification costs down. Image recognition methods are being developed and refined, offering the potential for the automated processing of samples in the future. Genetic fluorescent tags are also an area of research that could aid identification of diatoms species.

5 SEDIMENT CHEMISTRY-MACROINVERTEBRATE FAUNA RELATIONSHIPS IN URBAN STREAMS

This project was undertaken to improve our understanding of the effects of sediment contamination on benthic invertebrates in urban waterways. The majority of toxicity studies reported in the scientific literature have focussed on the effects of individual toxicants on a relatively small number of species; investigation of the effects of multiple toxicants on stream communities has been little studied. The project built on the Streamwatch program then being conducted across Melbourne by Melbourne Water to explore the relationship (if any) between sediment toxicants and benthic macroinvertebrates (O'Connor *et al*. 1999). An improved understanding of this relationship will assist the development of an overall classification and prediction system based on macroinvertebrates.

Sediment toxicants were sampled at 44 urban and semi-urban sites across the Melbourne region on two occasions, in autumn 1996 and spring 1996. Macroinverterbates were sampled at 41 of these sites. The data collected was explored using multi-variate analysis to identify:

- The relationship between macroinvertebrate community structure and sediments toxicants; and
- Provide recommendations for the design of effective urban stream health assessment tools.

The toxicants selected for analysis included:

- Heavy metals;
- Phenols;
- Polycyclic aromatic hydrocarbons;
- Benzene, toluene, ethylbenzene and xylene (BTEX);
- Petroleum hydrocarbons;
- Organochlorine pesticides;
- Polychlorinated biphenyls (PCB's);
- Oil and grease; and
- Pthalate esters.

Principal components analysis was used to explore the relationship between sediment toxicant and habitat data. The spatial and temporal trends in invertebrate data were explored using MSD ordinations on Bray-Curtis sample dissimilarity matrices. Environmental data were correlated with ordinations by principal axis correlation using the software package PATN (Belbin 1994).

While the distribution of toxicants was patchy, sediment contamination was generally highest in streams in the inner and western suburbs, the areas with the highest level of urban or industrial development. There were temporal trends for oil & grease (highest in spring) and pthalate esters (highest in autumn). Only heavy metals showed any relationship with the macroinvertebrate communities. No relationship between the various organic compound and invertebrate communities was evident.

The pattern of invertebrate community structure was similar to that recorded by the CRCFE (e.g. west to east gradient), even though different data sets were used. The gradients were related to topography, urbanisation and toxicant point sources.

Conclusion drawn from the study include:

- Invertebrates are a useful measure of stream health, but invertebrate-environmental relationships are complex and relating any impact to a specific pollutant is very difficult;
- Future investigations or management should focus on the effects of heavy metals on biota, as metals are persistent in the environment and were the only toxicants to show any relationship with invertebrate community structure;
- Sources of toxicant contamination should be stopped before undertaking rehabilitation;
- Metals are persistent in the environment; you don't need lots of temporal data for assessment
- Future investigations should consider bioturbation and its role in remobilising contaminants from the sediments.

5.1 General Discussion

Future monitoring should focus on metals as these are persistent (cf volatile organic compounds). A suite of metals such as mercury and zinc were implicated as key contaminants in this study. Their source was likely to be runoff from road surfaces and industrial areas. Recent work by the CRCFE suggests that lead levels can be useful for separating urban sites.

This study examined broad relationships between multiple contaminants and invertebrates. Toxicity identification and evaluation (TIE) methods could be used to evaluate responses to more specific contaminants and point sources. However, TIE is often constrained to a few species of convenience. It is not clear if these species are representative of stream conditions; their use for toxicity assessments could lead to order of magnitude uncertainties. Identifying local species for use in TIE could reduce this uncertainty.

The CRCFE and others are investigating the species that respond to toxicants (e.g. lead). While it is possible to detect toxicity associated with various chemicals, the ecological impacts that might ensue are less clear and this is an area for future research. Separating toxicity impacts from other factors (e.g. flow and pollution effects) is also an important area of future research. For example, diatom communities could be grown on glass slides and then placed in reference and test streams to help identify species affected by or tolerant of toxicants.

6 PHYSICAL CLASSIFICATION OF ESTUARIES

The objective of this project (Digby *et al.* 1999) was:

> *"To develop a national classification of Australian estuaries based on easily quantifiable biologically important physical characteristics, to enable valid comparisons between biotic communities of different estuaries".*

The underlying rationale of the project was to assist future ecological health assessments by identifying physically comparable environments in the same region; this will enable the comparison of biota from impacted and unimpacted estuaries.

Spatial, geographic, morphologic and climate data from 780 estuaries was compiled into the Australian Estuary Database (AED). This built on the Australian Estuarine Inventory compiled in the 1980's (Bucher and Saenger 1989). Statistical analysis was used to develop a physical classification model that explained as much of the variability as possible in the mangrove and saltmarsh areas in the estuaries. Mangroves and saltmarshes were selected, as they are present in most estuaries, they are easily measured, and are relatively stable on a seasonal basis and inter-annually.

An expert panel defined inputs to the model. Initially 29 attributes were considered for inclusion in the model. However, it was realised that this could result in millions of classes and there were less than 800 estuaries in Australia. Further iterations resulted in a final classification approach that placed estuaries into five groups on the basis of climate (tropical, tropical savanna, hot dry, subtropical, temperate). These classes were further divided into 15 groups based on tidal range (low, medium, high); of the 15 potential groups, 11 contained estuaries. Estuaries with a mid- or high-tide range were further sub-divided on the basis of their intertidal proportion. Additional variables were also considered, including:

- Mouth type;
- Mouth constriction;
- Main drainage line and type;
- Embayments;
- Fluvial flow and intertidal proportion.

General linear modelling was used to develop relationships between mangrove and saltmarsh areas and the independent variables (climate zone, fluvial flow, intertidal proportion, tidal range and estuary morphology). The final classification resulted 23 classes, 21 of which contained estuaries. The final classification model explained 44.5% and 42.5% of the variation in mangrove and saltmarsh proportions respectively and, therefore, provided a methodology with which to identify biologically important physical factors. The final classification scheme was applied to 623 of the 780 estuaries around Australia, placing estuaries into 11 of a possible 15 categories, explained predominantly by the Bureau of Meteorology's 1998 climatic zones and tidal range.

The classification system is now being adopted by agencies across Australia, including:

- Queensland EPA;
- Queensland Parks & Wildlife Service;
- Australian Institute of Marine Sciences;

- Great Barrier Reef Marine Park Authority;
- Queensland Fisheries;
- NSW Marine Parks Authority;
- National Land and Water Resources Audit;
- Australian Geographic Sciences Organisation
- Victorian EPA.

However, future maintenance will be a problem as there is no custodian for any new data collected by the agencies. Avenues for future development are a web-based system and applying the method to develop a classification system for New Zealand estuaries.

6.1 General Discussion

This work was consistent with the National Land and Water Audit, as the Digby estuary database was used for both. The classification was based on intertidal communities; unfortunately there was not sufficient information with which to assess its suitability for subtidal communities. It would be useful to develop a model based on subtidal communities and compare the two models.

It is hoped that the classification system will be a useful tool for management by enabling the comparison of impacted and unimpacted estuaries. However, there was some concern that the unique nature of most estuaries (e.g. due to founder effects etc.) may confound the practical application of reference versus impact comparisons. The development of a model based on subtidal communities could help to clarify this issue.

7 LITERATURE REVIEW OF ECOLOGICAL HEALTH ASSESSMENTS IN ESTUARIES

A synopsis of this project was not presented at the workshop. The abstract reported by Deeley and Paling (1999) is restated here.

No single environmental indicator will unambiguously define the interactions between ecosystem form and function, resilience and stability of biological communities and response of the estuarine system to anthropogenic stress. It is necessary to evaluate a broad range of potential measures simultaneously, in order to define appropriate ecological health indicators to underpin the management effort. There is however, no certainty in the selection and evaluation process and even with the best efforts, type I (false positive) and type II (false negative) errors are likely and both may prove expensive. Increased confidence in the selected indicator suite can flow from an evaluation of the monotonicity of correlated indicators, especially when assessments show consistent patterns arising from physico-chemical measures and measures of biotic community structure for various trophic groups.

Physico-chemical indicators of ecosystem processes have provided reliable information in the past, but problems have arisen from attempts to relate these measures to biological endpoints, particularly for estuaries with large interannual variability. In the absence of biological data for estuarine ecosystems experiencing extreme heterogeneity of climate influence, such as estuaries in the cyclone belt, physico-chemical indicators, or socio-economic indicators of anthropogenic influence may be the only option. Paleolimnological investigations may also provide additional insight, but the degree of taxonomic resolution required and the cost of stable isotope analysis may require considerable resources.

An evaluation of available historical data can better define temporal and spatial heterogeneity of systems and define normal behaviour and normal variability. Iterative refinement of the optimum indicator suite will require considerable ongoing research, monitoring and evaluation. Unfortunately, estuaries by their nature are ‘slow systems’ with decadal time constants for iterative loops of management measures and assessment of their success. It is a relatively simpler task to define indicators, which describe the status quo (e.g. degree of eutrophication), but it is much more difficult to develop a predictive capacity.

Autotrophic protistans (periphyton, phytoplankton), appear to be useful for describing nutrient enrichment, salinity and pH profiles, but complicating factors such as the nature of coupling of secondary predation need to be identified. Autecology of local indicator species also needs to be defined. Zooplankton appear to be limited as environmental indicators, but may be useful as elements of biotic indices across trophic groups. One of the major impediments to using planktonic organisms for inferring the condition of estuarine health is the considerable vertical, horizontal and temporal heterogeneity displayed by these organisms in both disturbed and undisturbed systems.

More recently, benthic macroinvertebrates have been successfully used to describe the nature and magnitude of organic enrichment of estuaries. Community structure, biomass and relative abundance of functional groups and indicator species have also been developed and used as environmental indicators.

Measures of community structure have problems because of a lack of information about interaction governing diversity and evenness of biotic communities and stability and resilience of the ecosystem. Species richness, diversity indices and measures of biomass have probably been the most widely used indicators in the majority of published works, but generally without appropriate critical analysis of their utility.

A myriad of biotic indices (ratios of functional groups) within and across trophic levels have been described in the international literature. There are problems in defining weightings for elements contributing to biotic indices and the loss of valuable information during these types of data reduction limit their potential. Detailed autecology of members of functional groups are required for biotic indices and this type of information is potentially available for some cosmopolitan species, but generally lacking for endemic species which may describe important nuances for the local environment.

As with biotic indices, there is a range of combined metrics described in the literature. Metrics generally combine physico-chemical elements, and may include some biological information. Many of the problems with the biotic indices apply equally to metrics, but when calibrated for a particular local situation, they offer considerable discriminatory power.

For Australian estuaries, physico-chemical measures of catchment and estuarine processes and socio-economic measures of anthropogenic influence may be of use. If assumptions about the linearity of interactions between the diversity of biotic communities and the stability and resilience of ecosystem function are valid, then conventional measures of community structure will also provide useful insights.

A hierarchy of environmental indicators is required for Australian estuaries, which provide for assessment of current status, a measure of diagnostic precision and a robust predictive capacity (‘early warning’). Of the range of potential indicators evaluated in this review, some core indicators have been used successfully by managers, some will require further development and others will need considerable additional research before links between stress and response have been established.

The ongoing selection, evaluation and refinement of environmental indicators for assessing ecological health of Australian estuaries, needs to proceed as a close partnership between land and waterway managers and scientific specialists.

8 ESTUARINE HEALTH ASSESSMENT USING BENTHIC MACROFAUNA

There is no commonly accepted definition of an estuary. However, broad-scale estuary health assessment requires a definition of the type of estuary that is to be monitored. The objectives of this project were, therefore, to:

- Determine if a RIVPACS type predictive model can be made of south-eastern Australian estuaries;
- Assess the usefulness of such a model for monitoring national estuarine health; and
- Assess other methods for monitoring national estuarine health.

This study was based on coastal-plain estuaries in Victoria. This excluded Port Phillip Bay and WesternPort, which were considered to be marine embayments rather than estuaries.

Estuaries are naturally stressed environments and it can be hard to separate human-induced effects from natural effects. For example, estuaries may vary naturally from well mixed, to poorly mixed, stratified systems that are affected by hypoxia events and sediment contaminant release.

A RIVPACS type model was constructed using data collected from 58 of 100 sites sampled in 29 estuaries across Victoria. Rare taxa (those that occurred at less than 4 sites) were excluded from the data set. Many of the excluded taxa occurred in near marine and near freshwater habitats and this may have reduced the importance of salinity gradients in the model. The model was tested by examining O/E scores from 6 estuary sites (Table 1). The low number of expected species meant that O/E scores had a large range.

Table 1: Summary for Upper Derwent Recovery

Site	Predicted	Expected	Observed	0/E
Curalo Lagoon, body	11	6.83	5	0.73
Tamago River, lower	7	4.01	3	0.75
Merriman Creek, upper	11	6.45	6	0.93
Lake Tyers, mid	18	13.08	14	1.07
Wingan Inlet, lower	7	4.49	5	1.11
Lake Yambuk, upper	11	7.35	9	1.22

The project looked at temporal variation (four seasons) at 11 sites in four estuaries and found that there was a large temporal variation in taxa. This suggests that variability in the O/E ratios could be reduced by changing the sampling procedure. By collecting samples at the time when the maximum numbers of taxa are present numbers of expected taxa would be higher, or, even better, would be sampling over a number of seasons as is done in the AUSRIVAS procedures.

While a working RIVPACS type model was developed with data collected in the pilot study, The model was considered to be of limited use for estuarine health assessment due to the low number of taxa predicted to occur at test sites. Further method and model development to increase the predicted taxa is required if a national protocol based on RIVPACS models is to be developed.

Multivariate analysis of K-dominance curves was recommended as an alternative to identify stressed sites. This method is independent of species composition and sampling methods and so can be applied widely K-dominance curves are plots of cumulative percentage abundance versus increasing species ranked by decreasing number of individuals, and are used to provide a graphical representation of species richness and species evenness (Moverley and Hirst 1999). While the ordination of multivariate data can be used to identify any differences between test and reference sites, it cannot be assumed that stress at test sites is due to anthropogenic impacts; this requires additional investigation.

Multivariate analysis of K-dominance curves for the southeast Australia reference sites found that 63% were healthy, 17% were unhealthy and 20% were indeterminate or questionable. The method was shown to be useful for detecting changes in the health of the upper Derwent Estuary in Tasmania following improvements to wastewater discharged from a paper mill (Table 2).

Table 2: Summary for Upper Derwent Recovery

	% Healthy	% Questionable	% Unhealthy
Reference	63	20	17
1990	9	34	57
1995	60	25	15
1998	76	20	4

8.1 General Discussion

At depths greater than approximately 0.5 m, estuaries that are not mixed by tidal or wind movement are naturally anoxic. Sampling of shallow sub-littoral biota rather than deep benthic biota could increase the number of estuaries that could be included in predictive models of estuarine health. While this may incur additional sampling and identification costs, the costs may be minimised if a rapid assessment method that included sub-sampling is developed. The costs for complete processing of samples per site currently vary, depending on the amount of organic matter found in a sample. Samples may take from hours (free of organic matter) up to weeks (considerable organic matter) to sort the average being a day.

9 ESTUARINE EUTROPHICATION MODELS

This study was undertaken to:

> 'Develop and demonstrate a widely applicable model of eutrophication processes in urban estuaries which can predict the potential for phytoplankton blooms and bloom types on the basis of environmental data'.

Various approaches to modelling estuarine eutrophication were reviewed, including both load-response models and complex process-based simulation models (Parslow *et al*. 1999a). The project focussed on relatively simple process-based models, which incorporate the core variables and processes controlling the cycling of nutrients and bloom development in estuaries.

A Simple Estuarine Eutrophication Model (SEEM) was developed, based on an inverse model that estimated physical exchanges in estuaries based on salinity data (Parslow *et al*. 1999b). The inverse model divides an estuary into length-wise columns and vertical layers (2D model), and has so far been applied to test cases with one layer (well mixed) and two layers (stratified or partially stratified). SEEM allows for two-way exchanges both horizontally and vertically, to represent both advection and diffusion (Figure 6).

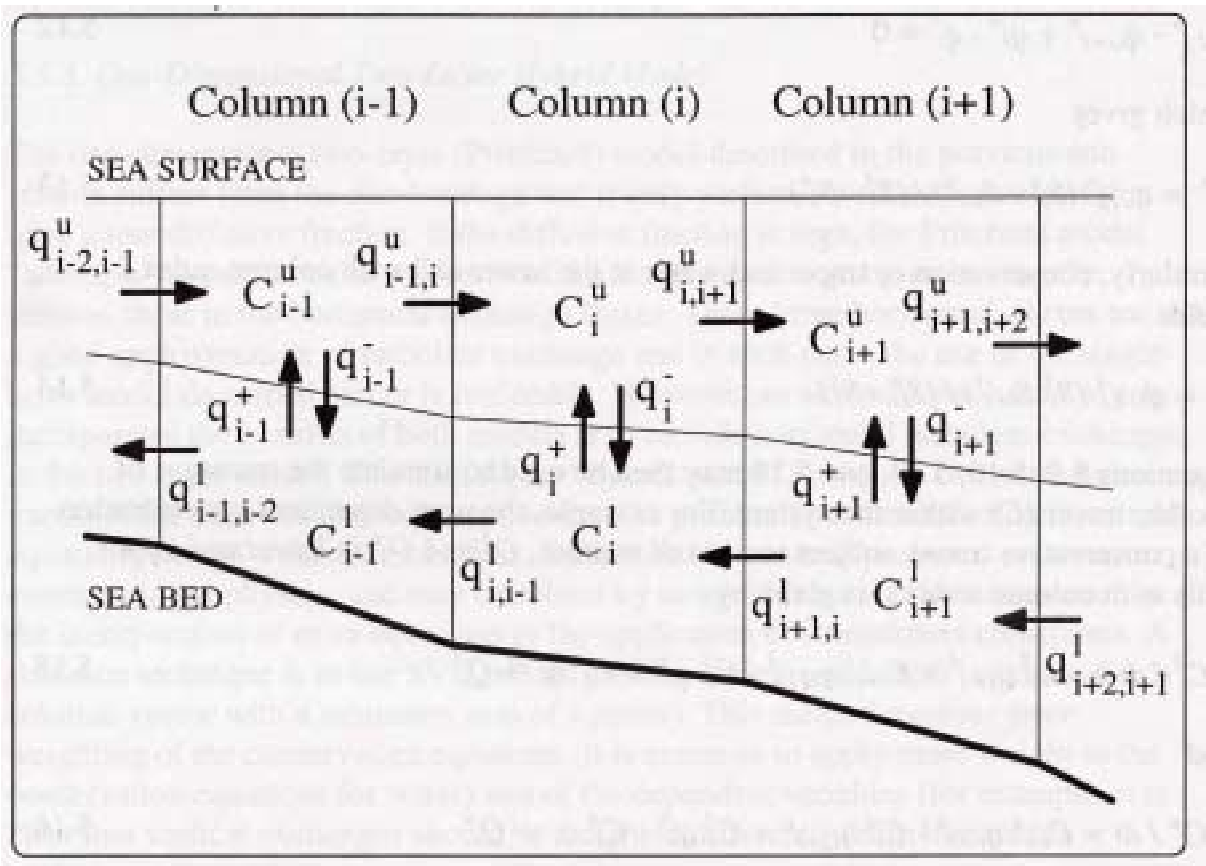

Figure 6: Structure of interior cells of a two-layer (Pritchard) model (from Parslow *et al.* 1999a)

The SEEM process model describes the interaction of nitrogen cycling and algal biomass (Figure 7) in terms of four pools:

1. Phytoplankton N;
2. Dissolved inorganic N (DIN);
3. Labile organic N; and
4. Refractory organic N.

Core processes include phytoplankton uptake of nutrients and growth, phytoplankton mortality, production and breakdown of labile and refractor organic N, and sediment respiration and denitrification. These processes may be modelled using a steady-state single-box model or a dynamic spatially-resolved model. For estuaries with long-estuary salinity gradients, the one-box model with a single flushing time is not valid. A spatially resolved model was therefore adopted for SEEM, using salinity distribution to infer flushing (Parslow *et al*. 1999a). SEEM has been developed as a java-based, user-friendly PC application (Parslow *et al*. 1999b).

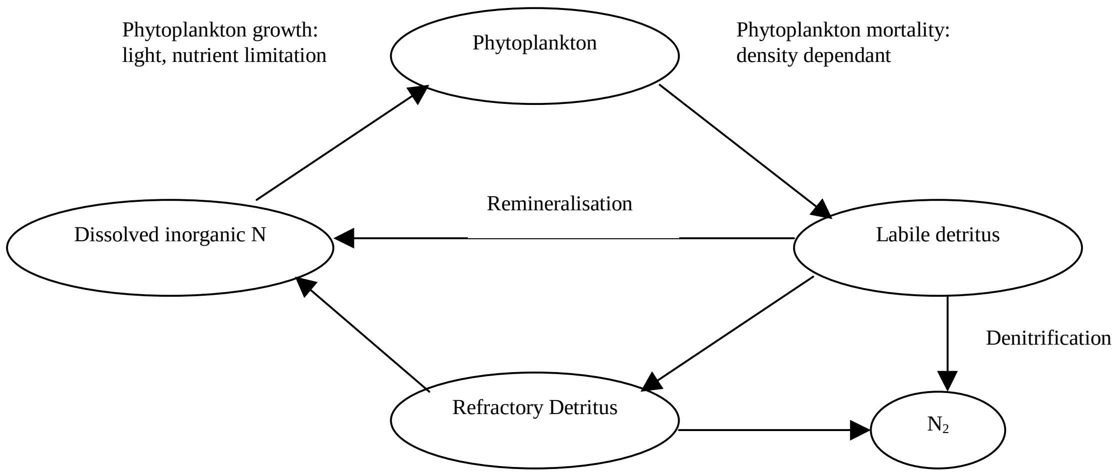

Figure 7: Conceptual basis of the SEEM process model

SEEM has been applied to three test cases, the Derwent, Brunswick and Hardy estuaries, representing different morphological, runoff and stratification environments. SEEM is also being applied to a major 3-year study of the Huon River in Tasmania.

Derwent Estuary, Tasmania

- Saltwedge estuary with stratification at high flows.
- Temperate, marine nitrate and phosphate inputs in winter
- STP loads along the estuary, but impact varies depending on location and depth of discharge. Loads discharged at the top of the estuary or into the bottom layer (and transported upstream) have the biggest effect on algal blooms

Brunswick Estuary, New South Wales
- Vertically mixed tidal estuary
- Large STP loads
- N load unutilised
- N or P limitation
- P load disappears (due to adsorption?)
- Impact of STP location – larger impacts for upstream location

Hardy Inlet, Western Australia
- Two layered system; mixed lagoon and channel
- Principal loads from the catchment
- DIN and DIP co-limitation
- An unidentified P source – possibly the bottom sediments

Huon River, Tasmania
- Salt wedge estuary, marine inputs in winter
- Fish farms loads in summer
- Summer algal blooms
- Modelled response to increased fish farming being used to set fish-farm quotas for the Huon River.

Overall, SEEM has proved to be very easy to use and quick to set up and apply. It is based on a small number of parameters and is easily tuned for application to different estuaries (as the model processes have been simplified, parameters must be tuned for each application). SEEM captures key processes associated with transport, phytoplankton response to light and nutrients; it doesn't capture the interactions between N and P, and the simplified sediment model cannot deal with seasonal or long-term exchanges between sediment and water column.

National Land and Water Audit
The National Land and Water Audit has adopted SERM (Simple Estuarine Response Model). SERM includes lagoon, salt wedge and tidal estuaries and is process based to account for N, P and C cycling. SERM also includes seagrass and benthic components. The model is robust and should not require fine tuning. It is currently being tested against focus estuaries and will eventually be available on the World Wide Web for easy access. This will allow those interested to test scenarios for any estuary of interest.

9.1 General Discussion

Both SEEM and SERM can be applied to lakes in addition to estuaries.

Stream and estuary systems may switch between N and P limitation. Updating the model to account for this requires the quantification of nutrient loads and knowledge of the processes associated with load sources and sinks. The inclusion of such a capability in the model could also be used to identify sources and sinks not previously accounted.

The National Land and Water Audit is attempting to include both physical and ecological classification as the basis for assessing estuary health. The difficulty so far has been in demonstrating ecological impacts and this is an area where information on biota such as invertebrates and diatoms collected by the Urban Sub-Program could be helpful.

10 FUTURE DIRECTIONS FOR URBAN WATERWAY RESEARCH

Over $1 m has been invested in the Urban Sub-Program. The maximum benefit from this investment will be realised when we have:

- Reviewed what we have learnt from the 8 projects that have been completed;
- Identified the best way to apply what we have learnt from the program; and
- Identified knowledge gaps and opportunities for further research or development that will build on the products and lessons learnt to date.

The seminar considered the use and limitations of models in managing ecosystem health. The aim of the monitoring is to assist decision making about the management of waterways in urban areas. Reference sites are selected, not as management goals, but as standards against which to judge change.

Biological Models

The concept of ecosystem health is multidimensional and requires a multivariate approach, as stream and estuary 'health' is a result of a number of environmental factors all acting together. Once we understand the processes that affect river health, we can then look for surrogates for future monitoring. Models such as AUSRIVAS provide a cost-effective measure of ecosystem status. AUSRIVAS was originally intended to have components based on fish, habitat, ecosystem respiration and primary productivity. However, only the invertebrate component has been developed to date. This begs the question 'are invertebrates enough to assess the health of streams in urban areas'? Developing AUSRIVAS further to enable the assessment of habitat conditions would be useful. This would also be consistent with geomorphological models and restoration protocols.

Biological models may inform us of the relative condition of a stream or estuary system, but by themselves may offer little for managers in terms of identifying the issues and priorities required to address an issue (e.g. rehabilitate a degraded stream; protect areas with a high conservation status). Managers need to identify the potential sources of stress and then find ways to fix or prevent environmental problems. An understanding of stream processes is required in order to develop a predictive capability that will help managers. This will be important for identifying what management issue to tackle first (e.g. flow, water quality or habitat?). The maintenance of stream habitat is important, as it is a precondition for good river health; a stream without physical habitat is generally considered to be degraded. If good quality habitat is present but stream biota are representative of degraded conditions, then this suggests water quality or flow problems. However, there are examples of streams with relatively little habitat, but good water quality and healthy invertebrate populations. Having good quality water also increases the likelihood of success for stream rehabilitation that involves habitat reintroduction. Managers will want to know where and on what to invest in order to improve environmental conditions in waterways (e.g. ensure good water quality in the upper catchments, then look to improve habitat condition). The availability of both biological and habitat indicators will be useful for assigning priority to stream management works.

The findings of the urban AUSRIVAS project found that there is no reason why pristine sites could not be used as reference sites for models to assess urban degradation. However, it is entirely appropriate that target conditions for restoration work might be less than pristine, and the use of catchment imperviousness can inform the distance from pristine that might be appropriate. Streams flowing through urban areas are usually stressed systems and the adoption of pristine reference sites as targets for rehabilitation may be unrealistic, as it is generally impossible to return urban streams to their prior condition. Setting rehabilitation targets therefore becomes a management decision, where reference is defined in relation to a set of values or quality goals that are the target for restoration (e.g. based on ‘least impacted’ sites in the area of interest). For example, organisations such as Melbourne Water might seek to improve or maintain waterways in an ecologically ‘healthy’ state, safe for human contact, with no visual pollution and a pleasing visual aspect or amenity.

Assessment models such as AUSRIVAS do not argue that reference sites are management end points; reference sites are standards against which to judge change (e.g. to river health) rather than whether or not rehabilitation targets have been met. It should also be remembered that reference sites are themselves dynamic, rather than static. In an urban stream context:

- Pristine sites, such as those used in regional AUSRIVAS or diatom models, may be used to assess the level of urban degradation;
- Reference sites should be drawn from areas with similar physiographic characteristics as the test sites. This is because only a limited number of variables can be expected to be operating equally across both reference and test sites (e.g. geographic location, altitude, slope, geology, rainfall, relative distance from mouth);
- Targets for rehabilitation may be based on aspects of ‘least impacted’ sites, or other values defined by management or the community; and
- Reference sites can be considered as a origin point for test sites, but not as rehabilitation end points or management targets

Catchment imperviousness may be used as a handicapping measure, and a covariate against which other ecological variables can be set (Walsh *et al*. 2001). For example, deviation from the O/E-catchment imperviousness relationship can be used to identify sites that are more or less disturbed for a given level of imperviousness (Breen *et al.* 2000). This offers the potential to set ecological targets (equivalent to least impacted sites for a given level of imperviousness) against which to quantify change. If this approach proves successful, it could overcome the difficulty in defining reference sites for systems where ‘pristine’ is unrealistic.

There was considerable discussion about the value of reference sites in setting management objectives and performance measures, especially for urban streams and estuaries. A counter-argument was put that managers should set objectives that are reasonable and acceptable to the community, even though these may be far from the “reference” condition. The counter-argument recognises that there is no single best measure of the health of streams and estuaries. Health can be measured along many dimensions, covering water quality, sediment quality, habitat, biogeochemical function and biological community structures. At some point, managers and the public have to start prioritising among these measures, and even to consider trade-offs among them.

Process-based models such as SEEM and SERM try to predict the response of multiple indicators covering water quality, sediment quality and simple ecological attributes such as macroalgae or seagrass, to management actions. These models can be used to address multi-objective management approaches. However, there is a need to couple these models to the diagnostic approaches based on community structure of benthic invertebrates, diatoms, etc.

Dirty Water Models
The problem of appropriate reference sites can be averted for small scale models derived for individual cities using the 'dirty water' model approach for scenario analysis. For example, if catchment imperviousness is increased from 5 to 35%, then we expect to lose 'these' biota. If we decrease nitrogen load by 50%, we expect to gain 'these' species. Catchment imperviousness can also be used to prioritise management works. For example, rehabilitation at stream site with a very high proportion of catchment imperviousness is unlikely to be successful or will be very expensive. A site with a low proportion of catchment imperviousness but in poor condition may be a prime candidate for rehabilitation. Dirty water models will be helpful for cost-benefit analysis of rehabilitation objectives.

Dirty water models show promise as a tool to assist managers and should be developed further. Dirty water models may also provide a way to link process models of water quality and habitat (like SEEM and SERM) to ecological health indicators based on the community composition of macroinvertebrates or diatoms. Such an approach would combine the diagnostic power of the ecological community-based approaches, with the predictive power of the process-based models.

Future Monitoring
Monitoring data has already been used to identify ecological patterns, and waterway attributes that can be managed. A commitment to long-term monitoring and evaluation should be seen as an investment in the future, as in addition to building models we can use it to look for anthropogenic effects and to detect long-term trends in condition. Habitat assessment and other environmental variables, in addition to biological monitoring (e.g. macroinvertebrates and diatoms), will useful in this regard. BOD and catchment imperviousness have already emerged as useful indicators of general urbanisation effects. Current work by the CRCFE suggests that sediment heavy metal concentration may also be a useful predictor.

Impervious area data should be collected for areas where AUSRIVAS and diatom models currently exist and the relationship between O/E ratios and impervious area explored more widely. It is likely this relationship will be influenced by climate, with the effects of impervious area being greater in high rainfall areas.

Future monitoring designs should be based on specific objectives or to answer key management questions. Monitoring programs are reduced in value if the data collected is not evaluated in light of the objectives for which the monitoring program was designed. When designing monitoring programs, we should also look at the information needs of available decision support systems that might assist management and consider this in design (e.g. habitat data, impervious area and biological data). The needs of estuary monitoring should also be kept in mind when designing monitoring programs for streams, as should the information needs of basic process models.

The proportion of catchment imperviousness is a key indicator of urban effects and a useful tool for identifying potential sites for rehabilitation (the higher the level of imperviousness, the less chance of rehabilitation success) (Walsh 2000). Impervious area and drainage efficiency should be included in future monitoring programs, along with measures of water quality, habitat and biota.

Image analysis
Biological monitoring programs are often limited by the costs associated with the sorting and identification of stream and estuary biota. Any means that reduces the costs of biological monitoring and evaluation programs is likely to be met favourably by the water industry. Image recognition systems are being developed to help reduce the time and effort required of trained staff for processing samples and recording data. For example, Macquarie University has developed an image recognition system for terrestrial invertebrates. A similar version for aquatic invertebrates would be useful.

Image recognition systems for identifying plankton have been developed (e.g. the Video Plankton Recorder from Woods Hole Institute, Massachusetts) but have not been adopted commercially in Australia due to the high initial outlays required. Current systems also require substantial taxonomic backup and so the benefits of adopting them are not yet clear. For example, image recognition may provide family level diatom identifications, but more information is likely to be gained from species level identification. Image capture and identification training at the species level is likely to be very difficult.

However, if (when?) set up costs are reduced and such systems are adopted, analysis costs are likely to decrease markedly. An indication of the relative sampling costs using image recognition might be gained from comparing the costs of current methods with that of the Macquarie University terrestrial invertebrate system.

Assessment Frameworks
Assessment frameworks have been developed overseas (e.g. USEPA) and similar systems could be developed for Australia. This could be in the form of a hierarchical protocol:

- Large magnitude impacts by expert opinion;
- Medium magnitude impacts by AUSRIVAS or diatoms;

Synthesis of Knowledge and Opportunities for Future Research
The Urban Sub-Program did not have any formal knowledge exchange process. Projects were conducted in parallel and there was little opportunity for collaboration between projects. Synthesis from the projects would help deliver the key messages and identify ways to promote the findings of the sub-program. Synthesis across the projects could also be used to develop standard approaches to monitoring and assessing the condition of waterways in urban areas. A challenge will be to achieve integration across a range of projects, programs and organisations. The benefits of this integration could be large (e.g. an agreement of estuary classification across Australia and standardised sampling strategies), but the process must be funded and will require careful management.

Waterway managers would also welcome the integration of the various models that have been used to classify and measure the health of urban streams and estuaries, especially in terms of developing definitions of what constitutes a healthy waterway that may be applied in a management context. Important components to consider will include ecology, hydrology, catchment imperviousness and habitat measures. For example, the CRCCH is currently developing a Decision Support System (DSS) that predicts changes in water quality and flow with changes in urban land use and stormwater mitigation practices. Ecological input to the DSS is being developed in collaboration with the CRCFE. The DSS is intended as a tool with which to identify the options available to address management issues. This and similar DSS's will be valuable resources and knowledge exchange tools. The opportunity exists for further collaborative efforts that aim to synthesise existing information and develop new, integrated prediction tools for management.

11 CONCLUSIONS

The key outcomes from the workshop are:

- Monitoring programs for urban streams and estuaries should be designed to provide data to facilitate decision making on the management of the stream or estuary.
- Monitoring should be designed with the ultimate intent of supporting process models. This is of particular importance in estuaries. In the first instance, monitoring need not provide a comprehensive data set but should be sufficient for the process models.
- Large-scale regional models based on pristine sites appear adequate at detecting varying levels of degradation in urban streams. The use of urban hinterland sites as references for the development of small-scale models to assess stream health at local sites is likely to be inappropriate, as low levels of catchment imperviousness can result in the impacts at the proposed reference sites that the models were designed to detect. Less-than-pristine targets for rehabilitation in urban settings may be set by reducing target O/E scores from such models with increasing levels of catchment imperviousness.
- When comparing urban streams, urbanisation, expressed as the “level of catchment imperviousness”, may be used as a basis of providing a valid comparison between sites.
- The aim of urban stream management should be to reach “best practice” rather than achieve “pristine conditions”.

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APPENDIX 1 WORKSHOP ATTENDEES

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Barry Ball	Brisbane City Council
Gary Bickford	EPA New South Wales
Peter Breen	CRC for Freshwater Ecology
Peter Cottingham	CRC for Freshwater Ecology
Bruce Gray	Environment Australia
Brian Hall	River Info
Neil Hughes	Environment Australia
Bob Humphries	Water Corporation
Jacob John	Curtin University of Technology
Peter Komidar	Environment Australia
John Langford	Water Services Association of Australia
John Moverley	Museum of Victoria
Nick O'Connor	ECOS Consulting
John Parslow	CSIRO Marine Research
Graham Rooney	Melbourne Water
Chris Walsh	CRC for Freshwater Ecology
Michael Whelan	Southern Cross University
Tony Wong	CRC for Catchment Hydrology