Use of invertebrate predictive models, the reference condition and causal criteria for ecological assessment of river condition

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Susan J. Nichols BAppSci(Hons)

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Institute for Applied Ecology, University of Canberra

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Abstract

This thesis presents my most significant contributions to the science of ecological assessment of river condition. The thesis traces the development of ecological assessment and shows where my work has made a significant contribution to knowledge of ecological assessment. I demonstrate the value of bioassessment and the 'reference condition approach' by describing applications and evaluation of the Australian River Assessment System (AUSRIVAS), which has been the national standard method of biological assessing river health for over a decade. AUSRIVAS includes a standardized invertebrate sampling method, the reference condition approach, predictive models, and software for assessing river health. However, new methods to aid the synthesis of ecological studies are imperative if the increasing body of scientific research is to improve management and outcomes for freshwater systems. My most recent work has contributed to establishing a new causal–criteria analysis method, 'Eco Evidence', for assessing evidence for and against environmental cause–effect hypotheses.

This thesis reviews bioassessment and AUSRIVAS predictive modelling, the reference condition approach, and the origins of Eco Evidence to provide background and context for my research. I have arranged the nine research articles that comprise the body of this thesis in three categories: 1) AUSRIVAS sampling method evaluation; 2) applications of AUSRIVAS; and 3) the synthesis of multiple studies for environmental causal assessment using Eco Evidence. In addition, the final chapter outlines problems encountered and future directions for the work.

A major contribution of my research has been to demonstrate the utility of the reference condition approach for (i) predicting reference (that is pre-impoundment) biota in the Cotter River (ACT); (ii) establishing reference biota within Kosciuszko National Park (Australia); and (iii) using the reference condition approach to assess the condition of Portuguese streams. This body of work is highly relevant to river managers wanting to apply the reference condition approach and (a) understand the consequences of sample variability on bioassessment results; (b) allocate resources appropriately for the level of replication required to detect an ecological response; and (c) avoid method-related bias where studies cross multiple jurisdictions that use different sampling methods. This research highlights the significance of standardized sampling of fixed sites (both test and reference) over long periods and demonstrates the value of the reference condition approach when assessing the biological

response to flow regulation. When applied within a robust study design and an adaptive management framework, the bioassessment program coped with changing questions and unforeseen events, such as extended drought. Application of AUSRIVAS has shown that management actions maintained the ecological resilience of the Cotter River, enabling it to recover when higher river flows returned after the drought.

This thesis also describes the recently published Eco Evidence method for systematic review of environmental science literature and draws together some lessons learned about the application of causal analysis to define ecosystem response to flow. The Eco Evidence method was adapted from epidemiological techniques for attributing causation. Such causal assessment can be necessary to inform management actions aiming to improve environmental condition. This work is highly relevant to researchers and environmental practitioners that require a method for quantifying and combining scientific evidence from multiple studies. The Eco Evidence weighting system for individual studies is a major advancement in environmental causal assessment. This research effort is part of a worldwide trend towards facilitating greater use of evidence-based methods in environmental assessment and management.

My research has contributed to advancing the understanding of ecological assessment that uses invertebrate predictive models, the reference condition approach and causal criteria analysis. Rigorous bioassessment studies and the reference condition approach when applied within the context of adaptive management, long-term assessment, and a framework for causal assessment, can provide the ecological evidence to inform current and future river management.

Acknowledgments

First, I would like to acknowledge Professor Richard Norris for his contribution as my mentor for 17 years. Under Richard's mentorship and training my knowledge of freshwater ecology and biological assessment of river health grew. Richard encouraged my research and publications as part of the overall output of his research group. This thesis therefore reflects the history of the ecological assessment research I have participated in through work on a great variety of projects over the years. Those projects yielded not only a wealth of experience but also a wonderful network of colleagues and friends. I am grateful to Richard for opening up a world of opportunities. His untimely passing was a great loss to us all.

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[1] Ann Milligan (nee Ann Petch) is a former agricultural scientist and hydrogeologist who has been editing and writing reports and articles about freshwater ecology since 1998.

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Introduction

Humans rely heavily on freshwater resources and managing these resources requires an understanding and integration of the physical, chemical, and biological interactions that define aquatic systems (Dodds and Whiles 2010). The demands for water from industry, agriculture, and power generation that our lifestyle requires are threatening the quality and security of water resources in terms of the economic, cultural, aesthetic, scientific and educational values (Malmqvist and Rundle 2002; Dudgeon et al. 2006). Exposure to the effects of water abstraction, pollution and habitat degradation can damage the biological communities that inhabit aquatic ecosystems (Malmqvist and Rundle 2002; Dudgeon et al. 2006). Thus, the measurement of aquatic communities can signal declining ecological conditions, or in the case of restoration, can also quantify the ecological success of management activities (Hellawell 1986).

My scientific research has been in the fields of freshwater ecology and assessment of river condition. The aim of the work has been to advance the understanding of ecological assessment, particularly aspects of assessment that use invertebrate predictive models, the reference condition approach and causal criteria analysis. This thesis presents my most significant contributions to a body of research that has advanced the understanding of ecological assessment of river condition in Australia, and in the international arena. I demonstrate how the scientific principles of study design, statistical inference, and aquatic invertebrate ecology underpin the methods for biological assessment of river condition (Chapters 3 and 4) (Nichols and Norris 2006; Nichols et al. 2006b) and describe various applications of the Australian River Assessment System (AUSRIVAS) methods (Chapters 5–9) (Nichols et al. 2006a; Feio et al. 2009; Nichols et al. 2010b; Norris and Nichols 2011; White et al. 2012). As the body of research on stream-bioassessment has grown and expanded the knowledge base of freshwater ecology, extensive associated biological datasets have developed. This has presented opportunities to analyse the collected data in ways to provide further insights into ecological processes, such as the role of disturbance. Chapter 7 (Nichols et al. 2010b) emphasizes the importance of long-term ecological studies for capturing the ecological effects of, and recovery from, disturbances in our changing environment (which includes assessment of climate-related effects on stream biota). Chapters 8 and 9 (Norris and Nichols 2011; White et al. 2012) also demonstrate how bioassessment and continued ecological research on environmental flow manipulations has been combined with appropriate study-design principles to achieve desired ecological outcomes, within an adaptive management framework. In Chapters 10 and 11 (Norris et al. 2012; Webb et al. 2012) I present my most recent research, which expands the theme of applying scientific principles and knowledge of study-design fundamentals to ecological assessment. These two chapters introduce a new causal criteria analysis method, 'Eco Evidence', to assess the evidence for and against environmental cause–effect hypotheses. These final two papers take the thesis beyond field-based studies and into desktop research drawing evidence from multiple studies within the largely underutilized pool of published scientific literature. Eco Evidence analysis has the potential to facilitate better use of the extensive research already published about ecological health, and change the way of doing environmental assessment in the future.

The structure of this thesis is as follows. Chapter 1 is a review of bioassessment and predictive modelling, the 'reference condition approach', and the origins of the Eco Evidence framework, to provide the research context. Chapter 2 is an overview of the contemporary relevance of each research output included in this thesis and the original and scholarly contribution they each make to knowledge in the disciplines of freshwater ecology and applied science. I briefly outline the principal significance of the findings and highlight the links between each published paper. I then present each paper as separate chapter (Chapters 3–11). The concluding chapter (Chapter 12) outlines problems encountered and proposed future directions for work in ecological assessment.

Chapter 1: Literature review

1.1 Bioassessment of river health: a review

The concept of 'river health' was explored at length in the 1999 special issue of the journal 'Freshwater Biology' (Boulton 1999; Karr 1999; Norris and Thoms 1999), and although the actual term was not defined, it is generally considered to convey an understanding that incorporates the river's ecological integrity (resilience) and invokes community concern about human impacts on rivers (Boulton 1999). River health is a value-laden term that means different things to different people depending on what they value about the river (often in terms of the goods and services that rivers provide) (Fig. 1). The ecological values associated with river health are based on maintenance of ecological integrity (ecosystem functioning) and resilience, and sometimes ecological values and human values will overlap to define river health (Fig. 1). However, ecosystem health cannot be measured directly, and therefore, surrogate measurements and observations are used to indicate the system's capacity to support key features, functions and processes (Davies et al. 2010).

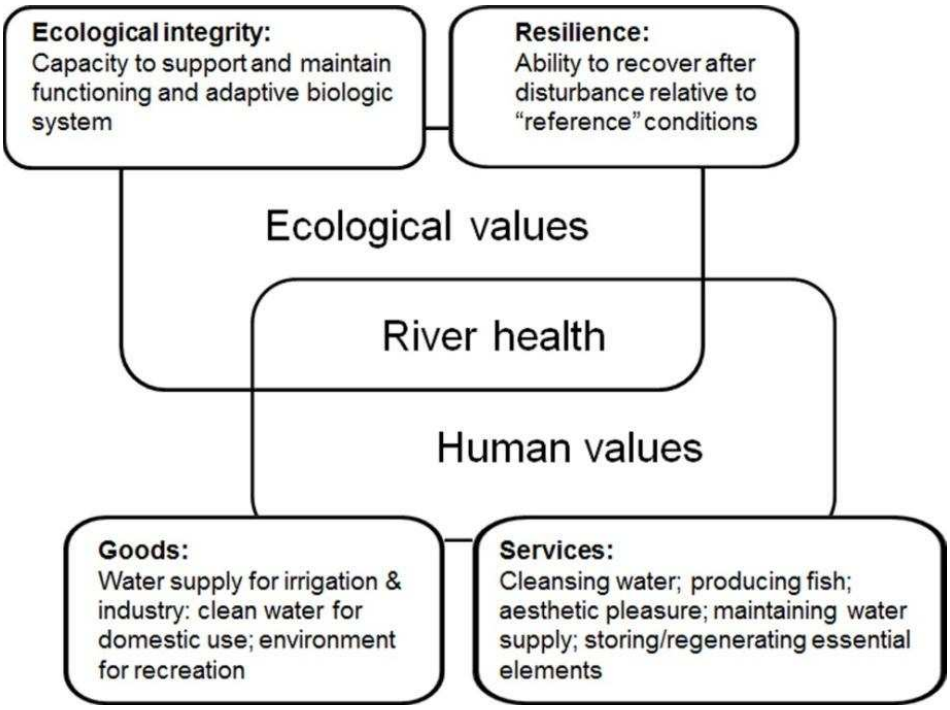


Figure 1. Representation of the concept of river health (redrawn from Boulton 1999).

Human activities often result in changes to river health, such as decline in water quality and physical degradation of the channel. The traditional method for assessing water quality used measurements of physical and chemical characteristics of water, mainly for the purpose of effluent discharge compliance and protecting humans (Norris and Thoms 1999). However, some biological characteristics (such as biochemical oxygen demand and algal growth) have a long history of use for assessing the effects of effluent discharge. Decades ago, the biological components of ecosystems were acknowledged as an essential part of ecosystem health assessment (Patrick 1949; Hynes 1960) and it was recognized that biota could provide a basis for setting environmental management objectives (Reynoldson and Metcalfe-Smith 1992). Traditional water quality assessment that relied on episodic or isolated observations was not well suited to large-scale management and protection of aquatic ecosystems (Norris and Thoms 1999).

Many people have reviewed and documented the evolution of bioassessment of freshwater systems and acknowledge the long tradition of using freshwater invertebrates for bioassessment (Hellawell 1986; Metcalfe 1989; Reynoldson and Metcalfe-Smith 1992; Rosenberg and Resh 1993; Bailey et al. 2004; Bonada et al. 2006). A definition of biological assessment is:

"*…the systematic use of biological responses to evaluate changes in the environment with the intent to use this information in a quality control program*" (Matthews et al. 1982 p. 129).

Aquatic invertebrates are commonly selected for bioassessment because they are present in most aquatic habitats, relatively easy to sample (compared to other biota), are a diverse group, long-lived and sedentary (thus representative of the location sampled), and respond strongly (often predictably) to disturbances in river systems (Reynoldson and Metcalfe-Smith 1992; Rosenberg and Resh 1993). Thus, aquatic invertebrate sampling can be used to monitor continuous or intermittent, and single or multiple, stressors and pollutants of the water they inhabit because their response is an integrative measure of exposure to environmental variability (Hellawell 1986; Rosenberg and Resh 1993).

However, the evolution of bioassessment has its origins in a variety of approaches and biota (Fig. 2). Early use of aquatic biota for waterway assessment focused on organic pollution and was based on biological response (of plankton and periphyton) to sewage contamination (e.g. the Saprobic System, Kolkwitz and Marsson 1909) and early attempts were made to classify systems based on their invertebrate communities (e.g. Forbes and Richardson 1913;

Thienemann 1925). The focus shifted to diversity indices with concentration on patterns of species richness and abundance (Patrick 1949; Simpson 1949) and they were widely adopted (Wilhm and Dorris 1968). During 1950–1970 numerous diversity and biotic indices were developed (see reviews in Washington 1984; Rosenberg and Resh 1993). They were appealing because they attempted to reduce data to a single number to understand the state of the community (Washington 1984). The underlying premise of using diversity indices was that undisturbed systems will be characterized by high taxa richness and evenness (Reynoldson and Metcalfe-Smith 1992). One major perceived advantage over the Saprobic approach was that diversity indices did not require (the often unavailable) information on the sensitivity of species to any particular pollutant. However, not using such sensitivity information when it was available was also considered a disadvantage. In addition, diversity indices do not account for naturally low diversity and any single number index should always be interpreted in the light of other biological evidence (Norris and Norris 1995).

Many of the numerous biotic indices, including the ‘Biological Monitoring Working Party’ (BMWP) score (Armitage et al. 1983), and later the Australian ‘Stream Invertebrate Grade Number – Average Level’ (SIGNAL) score (Chessman 1995), have their roots in the Trent Biotic Index (see overview by Reynoldson and Metcalfe-Smith 1992). Indices like the BMWP index and SIGNAL incorporate the sensitivities of invertebrates to pollution, and sum the sensitivity values for all taxa collected to arrive at a site score. Again, the US Hilsenhoff's Biotic Index (Hilsenhoff 1987) was originally developed to assess the response of stream-dwelling invertebrate assemblages to organic pollution.

The inclusion of evaluations of sensitive taxonomic groups, functional feeding groups (Merritt and Cummins 1984), and regional patterns of species assemblages emerged with the introduction of stream ecosystem concepts such as the River Continuum Concept (RCC) (Vannote et al. 1980), which made predictions regarding progressive downstream changes in functional processes. The RCC had limitations when applied globally because of regional differences. For example, in Australia the evergreen eucalypt forests produce litter-falls in summer, and disturbance from highly variable flow regimes appears more important in structuring biological communities (Boulton and Brock 1999). However, the RCC generated hypotheses to test outside North America (see Winterbourn et al. 1981; Greathouse and Pringle 2006). The development of RCC was not as a biological assessment approach. However, it created interest in functional measures of ecological process that could enhance diagnostic capability and provide an explanation of the cause of any observed structural change in biological assemblages.

The 1970s began an era of statistical rigour in the design of bioassessment and also ‘rapid’ assessment protocols (Green 1979; Karr 1981; Plafkin et al. 1989; Underwood 1993; Underwood 1994; Downes et al. 2002). Natural systems can exhibit change because of natural processes (Underwood 1994) and bioassessment must account for such variation to distinguish correctly between impairment and underlying natural variation (Downes et al. 2002). To assess whether a human activity has caused an observed change requires data from the location(s) before and after the commencement of the activity and from related ‘control’ location(s) (the Before-After, Control-Impact or BACI design, Green 1979). BACI and related designs (such as BACIP and MBACI, Faith et al. 1995; Carey and Keough 2002) incorporate fundamentals of study design that are applicable to current day bioassessment studies. Further, much of the work involving community similarity indices (e.g. Bray–Curtis index) and multivariate analysis techniques (e.g. Green 1971; Green 1974) was used in the predictive modelling approaches that were to become the next generation of bioassessment. These approaches incorporated comparisons between suspected stressed communities and a reference or control assemblage (Field et al. 1982). The approach linked the structure of the biological dataset to structure in associated environment data to determine if biota at undisturbed sites could be predicted from the site’s physical and chemical characteristics (Wright et al. 1984) — the predecessor to the reference condition approach (later defined by Reynoldson et al. 1997).

The main disadvantages of using invertebrates for routine bioassessment in their early use (before the development of rapid assessment techniques) were (from Reynoldson and Metcalfe-Smith 1992) that:

- identification to species (the taxonomic level generally used in early bioassessment) required specialist verification;
- invertebrate responses to human disturbances (other than sewage pollution) were not well documented;
- spatial heterogeneity in species distribution required appropriate sample replication that was time-consuming; and

- sample processing in quantitative studies was time-consuming and costly, which was prohibitive and precluded rapid results (Reynoldson and Metcalfe-Smith 1992).

When rapid assessment techniques were developed, they enabled the provision of qualitative and timely information, with savings in time and effort compared with previously used sampling-intensive, quantitative methods; and they often used family-level data rather than species (Metzeling et al. 2003). The United Kingdom, Australia and North America were the first regions to widely adopt these rapid bioassessment techniques. Two divergent forms of rapid assessment developed; they were the ‘multi-metric’ (e.g. Barbour et al. 1999) and the ‘multivariate’ techniques (Reynoldson et al 1997). Multi-metrics are attractive because they are generally easy to calculate and provide single value to compare to a target value. Multivariate techniques compare test sites to reference groups, which require the complexity of initial model construction but are usually incorporated into computer software for ease of use (Reynoldson et al. 1997). Examples of the multivariate techniques are the River Invertebrate Prediction and Classification System in the (RIVPACS; Wright 1995; Wright et al. 1998), the Australian River Assessment System (AUSRIVAS; Davies 2000; Simpson and Norris 2000) and the North American BEnthic Assessment of SedimenT (BEAST and CABIN; Hawkins et al. 2000; Reynoldson et al. 2000; Rosenberg et al. 2000). Both the rapid multi-metric approaches (e.g. Barbour et al. 1999) and the rapid multivariate techniques define and use a regional reference condition when assessing a site (Reynoldson et al. 1997).

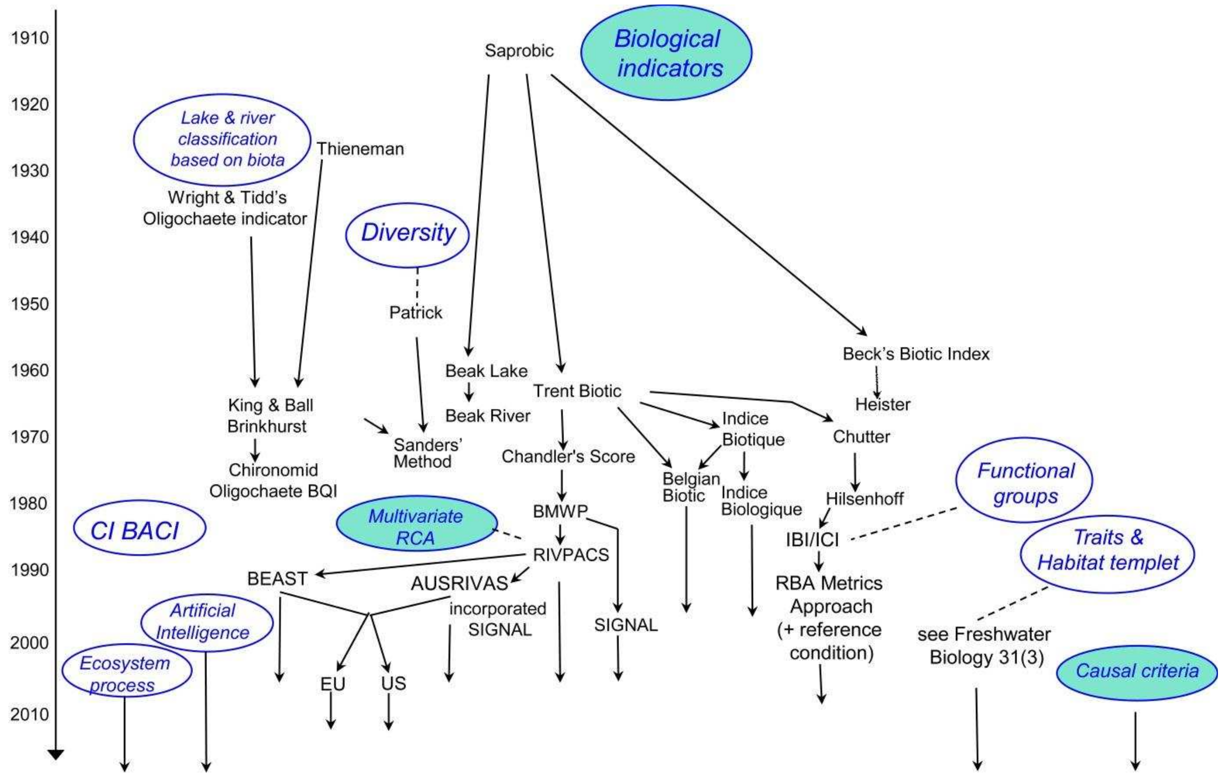
The reference condition definition given by Reynoldson et al. (1997) is:

“*…the condition that is representative of a group of minimally disturbed sites organized by selected physical, chemical, and biological characteristics”* (Reynoldson et al. 1997, p. 834).

An advantage of using the Reynoldson et al. definition of the reference condition, as used in the multivariate techniques, is that after grouping reference sites (using classification based on biota), independent data (e.g. physical and chemical attributes) are used to match test sites to reference-site groups for assessment (Reynoldson et al. 1997). This is important for bioassessment because it allows comparisons to sites where stream attributes are expected to yield similar invertebrate communities in the absence of disturbance, and that can account for naturally low diversity. The multivariate predictive-modelling approach does not assume that test sites exactly match a single reference site but instead calculates the probability of a test-site belonging to groups of reference sites. By 1997, design trends in freshwater studies suggested that reliance on information from only one or few reference sites as 'controls' was becoming less common (Reynoldson et al. 1997).

A European water-quality initiative adopted in 2000 (Water-Framework-Directive 2000) meant that European Union countries were required to establish consistent methods to assess the water quality of streams (Poquet et al. 2009; Feio and Poquet 2011). They too have chosen to use rapid bioassessment.

With this evolution of bioassessment around the world, the reference condition approach and the multivariate predictive modelling techniques have emerged as widely applicable in many contexts in numerous countries for the assessment of river health (Fig. 2).



Saprobic: (Kolkwitz and Marsson 1909); **Thienemann**: (Thienemann 1925); **Diversity**: (Patrick 1949); **Wright and Tidd**: (Wright 1955); **Beck's Biotic Index**: (Beck 1955), also see (Terrell and Perfetti 1996)**; King and Ball** (King and Ball 1964); **Brinkhurst** (Brinkhurst 1966**)**; **Trent Biotic Index**: (Woodiwiss 1964); **Beak**: (Beak 1965); **Indice Biotique**: (Tuffery and Verneaux 1968); **Chutter**: (Chutter 1972); **Chironomid and Oligochaete BQI**: (Milbrink 1973); **Sanders' method** (Sanders 1968); **Chandler's Score**: (Cook 1976); **Hilsenhoff**: (Hilsenhoff and Hine 1977); **CI, BACI**: (Green 1979); **BMWP** (National_Water_Council 1977); **Belgian Biotic**: (De Pauw et al. 1979); **IBI**: (Karr 1981); **Indice Biologique**: (Verneaux et al. 1982); **RIVPACS**: (Wright et al. 1984; Wright 1995); **RBA**: (Plafkin et al. 1989; Barbour et al. 1999); **Artificial Intelligence - Bayesian and Neural Networks**: (Walley and Fontama 2000 and references within); **Traits and habitat templet**: (Southwood 1977; Townsend and Hildrew 1994; and Freshwater Biology special issue 31(3)); **Ecosystem processes**: Benthic metabolism (Bunn et al. 1999); **SIGNAL**: (Chessman 1995); **AUSRIVAS**: (Simpson and Norris 2000); **BEAST**: (Reynoldson et al. 2000); **US**, United States: (Hawkins et al. 2000); **EU**, European Union countries: (Davy-Bowker et al. 2006; Feio et al. 2009); **Causal Criteria and Causal Assessment**: (Cormier et al. 2002; Suter et al. 2002; Norris et al. 2005; Norton et al. 2008; Norris et al. 2012).

Figure 2. The evolution of bioassessment in freshwater systems, showing the introduction of multivariate bioassessment techniques (e.g. RIVPACS) and the reference condition approach (RCA) in the 1980s and causal criteria analysis in early 2000 (redrawn and modified from Bailey et al. 2004 p. 9).

1.2 Predictive modelling bioassessment: an Australian perspective

In Australia during the early 1990s, large-scale environmental events and environmental policy resulted in funding for the national bioassessment initiative because of recognition that biological data were needed to make informed management decisions regarding the nation's rivers. Until then, management in Australia was slow to adopt the use of aquatic biota for assessing water quality (Norris and Georges 1986; Norris and Norris 1995). Once biological indicators were included in policy and water quality guidelines (such as ANZECC 1992), development of bioassessment was rapid. Aquatic invertebrate sampling became part of a national river assessment strategy in 1994 sponsored by the Australian government (the National River Health Program and the Monitoring River Health Initiative, see Davies 1994). The national sampling initiative was prompted by a 1992 prime ministerial statement on the environment following a persistent and extensive algal bloom in the Darling River in 1991 (Boulton 1999; Davies 2000). The initiative developed the national biological assessment protocol already mentioned—the Australian River Assessment System (AUSRIVAS) — which incorporated biological, physical and chemical assessment of stream condition. As with the US national program (Barbour et al. 1999), rapid collection, compilation, analysis, and interpretation of data were considered crucial to facilitate management decisions and the actions needed to control and/or mitigate impairment (Davies 2000).

Although the bioassessment protocol needed to produce rapid results and be cost-effective, the rationale needed a basis in sound theoretical concepts in ecology (Bonada et al. 2006). AUSRIVAS methods are conceptually based on the notion that the 'valley rules the stream' (Fig. 3), meaning that catchment character (or even broader scale controls) influences a river by controls on hydrology, sediment delivery and chemistry (Hynes 1975). In turn, the characteristics of a river reach control the occurrence of aquatic invertebrates, thus resulting in correlations between invertebrate composition and habitat characteristics (Southwood 1977; Townsend and Hildrew 1994). These theories, including the ecological niche concept (where niches are defined by habitat gradients and functional relationships among species) (Hutchinson 1957) are central to the AUSRIVAS approach to assessing river health (Norris and Thoms 1999). AUSRIVAS uses habitat variables (with attributes ranging from small to broad scale) to predict the occurrence of stream biota.

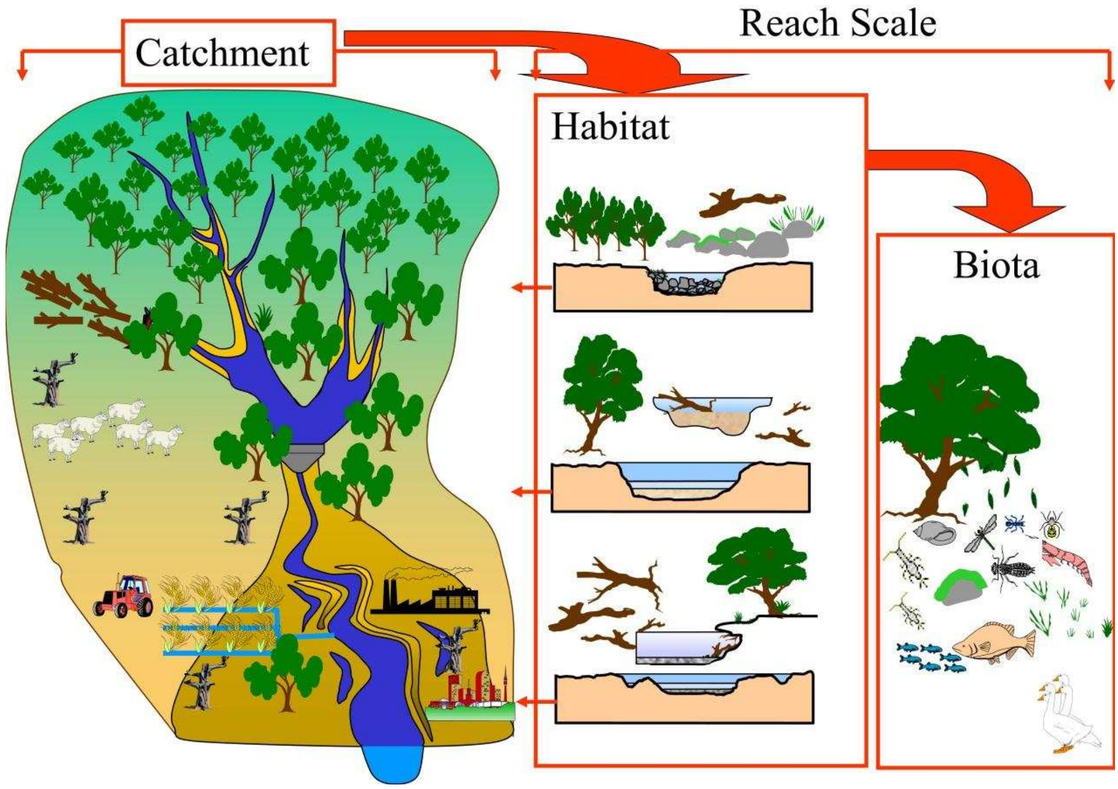


Figure 3. The concept that broad-scale catchment characteristics influence characteristics at reach scale (riparian vegetation, geomorphology) and together influence biota (from Norris et al. 2001b).

The importance of reference condition

The reference condition approach is also central to AUSRIVAS because knowledge of undisturbed 'control' conditions is essential in bioassessment (Bonada et al. 2006). Using an array of reference sites, the reference condition approach characterizes the biological condition of a region (defined by the spatial extent of the reference-site sampling regime). A test site (a site of unknown condition) is then compared to an appropriate subset of the reference sites (that is, sites with similar environmental characteristics) or (in the case of AUSRIVAS) to all the reference sites but using the probability of group membership as weightings (Reynoldson et al. 1997; Bailey et al. 2004).

Specific (and often subjective) *a priori* criteria define a reference site. These criteria might include being free from pollutants and free from human disturbances like land clearing (Davies 1994; Yates and Bailey 2010). For bioassessment based on predictive modelling, the reference condition should represent the range of the biotic variability among reference sites within the region under assessment. It should be the best available condition (Reynoldson et al. 1997) and is an alternative to control sites (that is, sites matched as closely as possible with the test sites, but not exposed to human disturbances) so as to provide benchmarks for the assessment of river condition.

A management target condition is not necessarily the same as reference condition, but possibly somewhere between the current condition and the reference condition (Fig. 4).

"Reference and target condition are different concepts and should not be confused: Reference condition provides a benchmark from which you can determine how far, and in what direction, the river's condition has changed …Target condition represents your management goal" (Whittington 2002 p. 4).

The target condition may be the 'best attainable' condition agreed to by stakeholders under a given scenario of human activity. In a management and economic sense the difference between the best attainable and reference condition is the ecological cost of accepting that target condition (Fig. 4).

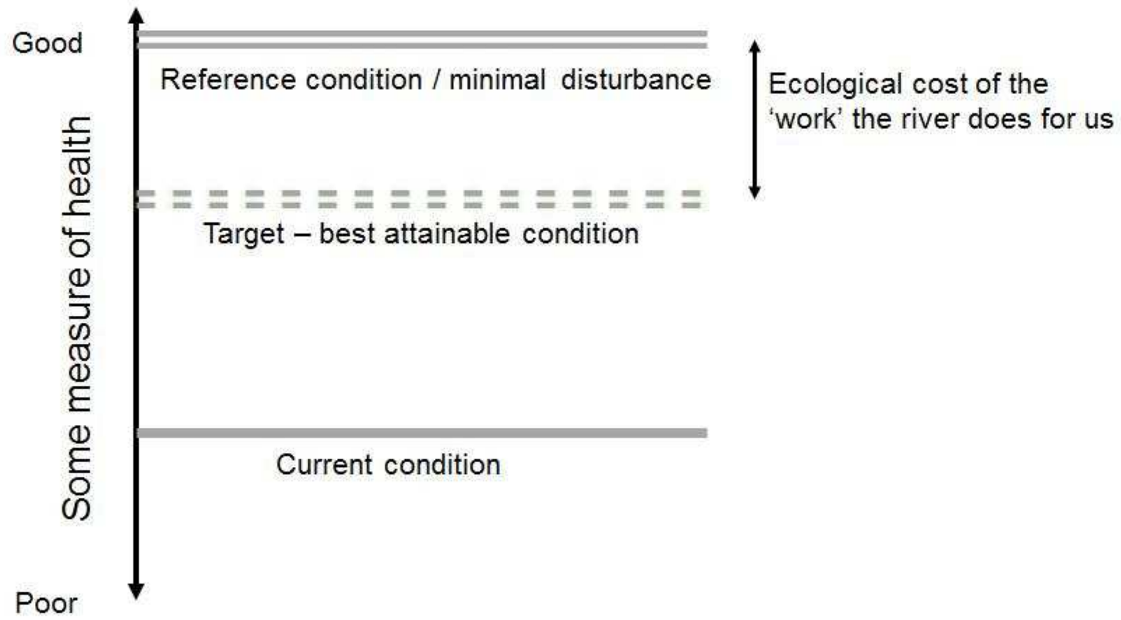


Figure 4. In many cases, target condition will not be the same as reference condition but rather somewhere between target condition and reference condition (redrawn from O'Connor and Nichols 2006).

Development of AUSRIVAS based on benthic invertebrates

The initial development steps for AUSRIVAS, summarized in Davies (2000) included:

- a preliminary evaluation of the proposed predictive modelling and reference condition approach to bioassessment by a technical working group (Davies 1994);
- the production of a single bioassessment manual describing the process of developing a predictive model for Australian conditions (Coysh et al. 2000);
- contracting a government agency in each of the eight states and territories that would use the manual to develop RIVPACS-type predictive models (Wright 1995);
- contracting a team from the Cooperative Research Centre for Freshwater Ecology (based at University of Canberra) to assist the state and territory agencies with statistical training and analysis, and to develop the predictive models and software platform; and
- a technical advisory committee to review ongoing issues.

Two conditions were fundamental to the successful development of a national bioassessment approach such as AUSRIVAS (Davies 2000). First, an ongoing collaborative and integrated approach needed a stable custodial environment. Second, the continued confidence in the bioassessment tool required a way to minimize the risk associated with the application of rapid bioassessment in unsuitable situations. Confidence might lapse if users were not aware of the tool's limitations. Davies (2000) identified important factors for the success of AUSRIVAS that included the development of an integrated river assessment 'package' that would need to overcome constraints such as, available funding, available expertise, and the adoption by management.

By mid-2000, the intended AUSRIVAS package was to include:

- documentation of the history and explanation of the components;
- biological sampling protocols that incorporated physical and chemical assessment of condition and a manual (initially for invertebrates but with the intent to include additional biological indicators, such as algae, fish and measurement of functional attributes such as benthic metabolism);
- invertebrate taxonomic keys and coding system;
- a habitat assessment protocol;

- software and user manual; and
- training and accreditation materials.

Delivery of most of the above items was completed, creating AUSRIVAS, a bioassessment system that uses aquatic invertebrate data to assess river health (eWaterCRC 2012b). It includes:

- standardized invertebrate sampling and processing methods;
- state and regional predictive models;
- software to output observed / expected (O/E) scores, expected taxa with probability of occurrence, SIGNAL, and impairment bands; and
- user training and accreditation for AUSRIVAS methods.

The one major omission from the intended AUSRIVAS package has been the use of other biological indicators, and so AUSRIVAS remains an assessment method using only aquatic invertebrates.

Protocols using alternative biota have been trialled in Australia: for example, diatoms (John 2004), fish (Davies et al. 2008) and benthic metabolism (Fellows et al. 2006), but multijurisdictional standardized methods have not been widely adopted. A choice of biological assemblage in a bioassessment program would provide greater opportunity to select an appropriate indicator given the characteristics of the area under assessment and the study objectives (Resh 2008). For example, if a small area is under assessment then it would be important to use an indicator with limited mobility and, if related to the study objectives, an indicator sensitive to relevant nutrients and herbicides.

Australia's large size and associated factors were important issues to consider when modifying the RIVPACS method for use in Australia (Turak et al. 1999). Technical challenges were realized at the time of AUSRIVAS development, such as modelling methods that could incorporate spatial and temporal variability into the bioassessment process, and the ability to develop predictive bioassessment for the highly variable streams of the semi-arid and arid regions of Australia (Davies 2000). A comprehensive account of the development of RIVPACS-style predictive modelling and reference condition approach for bioassessment is provided in a book authored by the participants of a 1997 workshop held in Oxford, UK (Wright et al. 2000).

The Oxford workshop also identified challenges facing the further development of bioassessment, which included identifying and reducing bias from sample variability (related to temporal variability and sampling method bias) (Humphrey et al. 2000), assessing the precision and accuracy of indices (and the uncertainty in 'quality banding') (Clarke 2000) and variation in reference sites through time. My research has addressed and tested some of these issues and the outcomes are described later in this thesis (see Chapters 3, 4 and 7, Nichols and Norris 2006; Nichols et al. 2006b; Nichols et al. 2010b). However, other issues, such as alternatives to the use of reference sites where suitable reference sites are unavailable (Reynoldson and Wright 2000), the use of alternative modelling approaches, and improving the links between structural and functional components of aquatic ecosystems (Johnson 2000; Walley and Fontama 2000) remain as challenges for bioassessment in Australia (Chessman et al. 2010).

1.3 AUSRIVAS methods as used for my research outputs

Bioassessment assumes that the samples collected will be representative of the invertebrate assemblage present in a river reach for the purpose of distinguishing between disturbed and undisturbed sites (Diamond et al. 1996; Metzeling et al. 2003; Gillies et al. 2009). The major components of the AUSRIVAS sampling protocol are standardized to reduce sampling bias, to ensure accuracy of site assessments and to make valid sample comparisons (Humphrey et al. 2000). The research outputs presented in Chapters 3 and 4 have tested and analysed some of the assumptions associated with standardized sampling methods and the accuracy of results when comparing test and reference sites (Nichols and Norris 2006; Nichols et al. 2006b).

The sampling protocol: habitats, sub-sampling and season

Within the AUSRIVAS protocol, separate samples are collected from specific habitats (Nichols et al. 2000). The justification given for sampling single habitats is that invertebrate samples from different habitats (e.g. riffle and stream edge) have a distinctive faunal composition (Parsons and Norris 1996). Samples collected from the same habitat at different sites were more similar than samples from different habitats at the same site (Parsons and Norris 1996). Habitat–effort bias would result if all habitats are not present at all sites (a problem particularly in regions with many differing stream types) or where habitats are not present in the same proportions at all the sampled sites within a region (this could have been accounted for by sampling in proportion to available habitat, however, it is not the established

AUSRIVAS sampling protocol, Chessman et al. 2007). Thus, sampling all habitats at a site to produce a composite sample could produce a level of variability that could confound detection of biological damage. However, the results from other bioassessment studies that sampled multiple habitats for composite samples have not produced consistent findings to support a consensus for composite or single habitat sampling (Hewlett 2000; Ostermiller and Hawkins 2004; Gerth and Herlihy 2006). The degree to which multi-habitat sampling will affect bioassessment results will depend on the distribution and variability of habitats within the region under assessment and the influence of spatial scale of sampling sites (Chessman et al. 2007). Standardized sampling of specific habitats is considered an important component of the AUSRIVAS bioassessment technique because it reduces between-site sample variability and thus maximizes the potential to detect adverse effects from human activities (Parsons and Norris 1996). Further, a particular human activity can affect one habitat but not another (Roy et al. 2003), which is a reason the AUSRIVAS sampling protocol recommends sampling two major habitats when assessing river condition.

For AUSRIVAS assessments, the collection of one invertebrate sample involves sampling 10 metres of a single habitat. Invertebrates collected using a D-framed pond-net require sorting from the associated detritus, plant matter and sediment. Sorting and counting all the animals from the large samples that are usually collected is time consuming and costly, and would negate one of the 'rapid' components of the bioassessment technique. Consequently, the sample is sub-sampled using fixed counts of approximately 200 animals. The decision about how many animals to count and identify from a sample remains a contentious issue for those developing a bioassessment program (Cao and Hawkins 2011), and it necessitates compromises. The use of small fixed-count samples to estimate differences in taxon richness among sites could lead to misleading estimates among sites (Cao et al. 2002; Cao and Hawkins 2005; Cao et al. 2007). Confident use of any biological assessment method depends on understanding the random variation and error that can influence the site assessments based on the samples. I have tested the effects of invertebrate sampling error and variability on bioassessment results and the findings are presented as articles in Chapters 3 and 4.

There is natural seasonal variation in invertebrate assemblages. In AUSRIVAS, both season-specific sampling and associated predictive models account for the variation. The collection of invertebrate samples is restricted to particular seasons to ensure sample comparability (particularly if data are used to build predictive models or if analysed using the season-specific predictive models) (Coysh et al. 2000; Simpson and Norris 2000). The models predict the taxa present in a sample from a particular region, season and habitat.

Predictive model development

Building of AUSRIVAS predictive models requires invertebrate and environmental data collected from reference sites. Generally, the AUSRIVAS models use invertebrate data at family-level taxonomic resolution, except Chironomidae (to sub-family), Oligochaeta (class) and Acarina (to order) (Parsons and Norris 1996; Simpson and Norris 2000). However, there are exceptions, with some species and genus-level models developed in an attempt to improve sensitivity in detecting ecological disturbance (Marchant et al. 1997; Lamche and Fukuda 2008). The assumption is that species or genera within families may differ in respect to water quality tolerances, flow preferences, and food requirements and thus, will respond differentially to human disturbances (Lenat and Resh 2001). In many cases, testing this assumption is not possible because of the lack of species-specific knowledge. Further, species models for bioassessment may perform no better than family models in regions characterized by low species diversity within families (Bailey et al. 2001; Chessman et al. 2007; Heino and Soininen 2007).

AUSRIVAS models were developed by adapting the approach originally described by the authors of the RIVPACS models (Wright et al. 1984; Wright 1995). The method of model building begins by using cluster analysis to classify groups of reference sites based on their aquatic invertebrate assemblages. A set of potential 'predictor variables', not influenced by human disturbance, are selected from the environmental dataset. The current models use predictor variables acquired from maps (e.g. location, altitude, stream order, distance from source, catchment area) or measured directly in the field (e.g. channel characteristics). A subset of predictor variables is selected by using discriminant function analysis to determine which environmental variables best discriminate between the naturally occurring groups established from the reference site invertebrate-assemblage data (Simpson and Norris 2000). The probability of a test site belonging to each of the reference site groupings from the faunal classification is calculated (Coysh et al. 2000). The frequency of occurrence of each taxon in each of the reference-site groups is then determined. Multiplying a taxon's frequency of occurrence in a classification group by the probability of a test site belonging to that group and summing the results for all of the groups in the model, gives the probability of a taxon occurring at that test site (Table 1). Most bioassessment predictive modelling currently involves this method of site classification, followed by discriminant function modelling to predict the probability of a new (test) site's membership to reference site groups using the subset of environmental predictor variables (Van Sickle et al. 2006). However, unlike AUSRIVAS, not all methods calculate taxon occurrence by using the probabilities from all groups, but rather use the most similar group (e.g. Reynoldson et al. 2000).

Table 1. Example AUSRIVAS calculation of the probability of a taxon occurring at a test site for this four-group model. Combined probability that taxon X will occur at Site Y = 76.5 % (from Coysh et al. 2000).

Classification group	Probability that test site Y belongs to group	Frequency of taxon X in group (%)	Contribution to probability that taxon X will occur at site Y (%)
A	0.50	90	45.00
B	0.30	70	21.00
C	0.15	60	9.00
D	0.05	30	1.50
			Total 76.50

1.4 Using predictive models

The AUSRIVAS models predict the number of taxa expected at a site by summing the individual probabilities of occurrence for all the taxa predicted to have >50% probability of occurrence (Simpson and Norris 2000). The number of taxa actually observed (O) at each test site is then compared with the number expected (E) based on the reference site data, to produce an O/E score. How much the observed invertebrate assemblage (O) deviates from that expected (E) is a measure of the severity of any environmental degradation at the test site. To aid interpretation, the AUSRIVAS O/E scores are assigned to quality bands that represent different levels of biological condition (Coysh et al. 2000). For example, sites with O/E scores in band A are similar to reference condition, whereas sites with O/E values in band B or lower are considered impaired (Table 2).

Table 2. Example of AUSRIVAS bands of biological condition for the ACT-autumn-riffle model, showing O/E range, band descriptions and interpretations (adapted from Coysh et al. 2000).

Band	O/E value	Band description	O/E interpretations
X	>1.12	**More biologically diverse than reference** O/E value greater than 90th percentile of reference sites used to create the model.	More taxa found than expected. This could mean that the site is potential rich in biodiversity. Alternatively, the site may have mild organic enrichment that initially could increase the number of taxa (e.g. by favouring certain suspension – deposition feeders) because of increased food resources resulting from the increase in nutrients (Rosenberg and Resh 1993). Likewise, a continuous irrigation flow in a normally intermittent stream may result in more taxa than expected. Thus, a test site falling in band X requires further consideration before a conclusion is drawn.
A	0.88–1.12	**Similar to reference** O/E within range of central 80% of reference sites used to create the model.	Most/all of the expected taxa are found. This indicates water quality and/or habitat condition similar to reference sites.
B	0.64–0.87	**Significantly impaired** O/E below 10th percentile of reference sites used to create the model. The band width is equal to band A.	Fewer families than expected. It is possible that the water quality and/or habitat quality are impaired, resulting in loss of expected taxa.
C	0.40–0.63	**Severely impaired** O/E value below band B. Band B width is equal to band A.	Many fewer families than expected. Poor water quality and/or habitat quality resulting in loss of expected invertebrate diversity.
D	0–0.39	**Extremely impaired** O/E value below band C down to zero.	Few of the expected families and only the hardy, and pollution tolerant taxa remain. Extremely poor water quality and/or habitat quality resulting in severe impairment.

Effects of sample variability, sub-sampling method and scale on the biological assessment results

The important questions for management agencies and the users of bioassessment outputs relate most directly to the conclusions regarding river condition that are drawn from the biological assessment. The AUSRIVAS predictive modelling approach uses the invertebrate data to make comparisons between sites, and the confidence placed in those comparisons will depend on ensuring that differences are unrelated to sampling error (Clarke 2000; Humphrey et al. 2000; Bailey et al. 2004; Ostermiller and Hawkins 2004). Confident use of any biological assessment method depends on having some idea of random variation and error that can influence the site assessment (Clarke 2000; Clarke et al. 2002).

Various sources of error in the observed taxa and the values of biotic indices can arise from sampling variation and from sample processing errors (Clarke 2000; Humphrey et al. 2000; Clarke et al. 2002), such as:

- sampling variation resulting from spatial heterogeneity within stream habitat;
- effects of environmental stress resulting in taxa being absent from samples because of natural rather than human-induced stress;
- sample processing error (such as poor recovery of small and cryptic taxa related to method of collection);
- identification errors; and
- natural temporal variation.

Variability and sources of error are always issues of concern for those developing and designing monitoring and assessment programs (Herlihy et al. 2008; Cao and Hawkins 2011; Solimini et al. 2011). Thus, rigorous assessment of data comparability should be a standard aspect of quality assurance when developing and applying biological indices (Cao and Hawkins 2011). Studies that use AUSRIVAS sampling often rely on a single invertebrate collection (without replication) sampled from 10 metres of specific habitat at a site (e.g. stream edge and/or riffle), which is then compared with the reference condition assemblage by using the predictive models (Nichols et al. 2000; Turak et al. 2004). Chapters 3 and 4 of this thesis address concerns regarding the effects of sample variability on the AUSRIVAS results (Nichols and Norris 2006; Nichols et al. 2006b).

Over 10,000 river reaches nationwide have now been sampled using AUSRIVAS methods and assessments made by AUSRIVAS model outputs (Norris et al. 2001b; Davies 2007). Studies range in scale from broad national and statewide scale (Smith et al. 1999; Turak et al. 1999; Metzeling et al. 2006) to targeted assessment at smaller scales (Sloane and Norris 2003; Peat and Norris 2007; Harrison et al. 2008). Chapter 5 presents a case study (Nichols et al. 2006a) that exemplifies the use of AUSRIVAS in a catchment of 482 km^2 and critiques the AUSRIVAS method for assessing the effects of river regulation. Chapter 6 describes another AUSRIVAS application where the methods were transferable for the broad-scale assessment of streams in Portugal.

Long-term assessment and reference site stability

Research into freshwater ecology and assessment of the biological health of rivers in Australia, and elsewhere, has contributed to building extensive biological-assessment datasets. Long-term biological datasets are particularly valuable for assessing ecological responses to environmental change (Jackson and Fureder 2006; Mazor et al. 2009). Defining what constitutes a long-term study is done by considering the sampling regime needed to detect trends, which will depend on the selected ecological measurement (Peterson et al. 2011). For example, the assessment duration needed to detect an ecological response (e.g. reproductive success) in short-lived species will differ from that needed for long-lived species. Likewise, whether wishing to detect annual or longer-term trends in river condition, a sampling program needs to consider sampling frequency, season, and flow-dependence to detect variation beyond 'normal' for the selected indicator (Souchon et al. 2008). Regardless of whether a change is the result of human activities or natural, without long-term data it is difficult to assess where the system is positioned along a trajectory toward recovery (Peterson et al. 2011).

The reference condition approach and predictive modelling are now common methods used in freshwater assessment (such as the AUSRIVAS, RIVPACS, BEAST approaches). These techniques use the invertebrate assemblage sampled at reference sites to define the reference condition. To maintain the relevance and integrity of bioassessment results for continued use over many years, the reference condition should remain stable through time, or the temporal change (drift) should not exceed the spatial variation within a defined reference condition. Long-term assessment of reference sites can validate assumptions of stability. Chapter 7

(Nichols et al. 2010b) describes a research study that shows how the reference-condition approach maintained the integrity of a bioassessment program through time.

Assessing ecological outcomes in variable environments

Programs to assess river condition and inform management actions need underpinning by rigorous science designed within an 'adaptive management' framework, which can then guide management actions and be responsive in a changing environment (Bunn et al. 2010). The term 'adaptive management' has been used for over 30 years to describe an experimental approach to natural resource management (Holling 1978; Walters 1986), and the approach is widely advocated (Poff et al. 2003; King et al. 2010; Kingsford et al. 2011). Adaptive management aims to test hypotheses and is considered excellent in principle, yet difficult to implement (Lee 1999).

Key elements of the adaptive management approach include:

- explicit definition of the desired management outcomes;
- development of management strategies to achieve those objectives;
- implementation of strategies in a comparative experimental framework to improve understanding of system responses to management actions;
- assessment to evaluate the success and limitations of alternate strategies; and
- iterative modification of management strategies to improve management outcomes (Keith et al. 2011).

Adaptive management is a process that recognizes inherent uncertainties of dynamic and unpredictable ecosystems but tests these uncertainties (Kingsford et al. 2011). The process should allow ongoing reflection, learning and adaptation to improve management and ecological outcomes. However, for management to adopt this approach the changes need to result in detectable outcomes within a reasonable timeframe and have low political risks (Sutherland 2006). Integrating decision-making with best available scientific knowledge in a collaboration between scientists, managers, and other stakeholders to link defined (and agreed) ecological values and objectives will help to overcome barriers to adoption (Poff et al. 2003; King et al. 2010). Chapters 8 and 9 (Norris and Nichols 2011; White et al. 2012) demonstrate the use of AUSRIVAS and principles of good study design within an adaptive management framework.

1.5 The need for new methods to aid the synthesis of ecological assessment studies

Well-designed bioassessment studies can do much to inform management actions. However, relating the observed biological responses to a range of drivers, such as possible flow regimes, often requires more. Drawing evidence from other research findings and multiple studies within the largely underutilized pool of published scientific literature is a relatively new way of assembling the rigorous science needed to inform management actions. Sound decision-making for river management and rehabilitation requires an understanding of the cause–effect relationships between environmental stressors and ecological response. While river management activities are numerous, studies that assess their effectiveness in achieving the desired ecological outcomes are less common (Palmer et al. 2005; Souchon et al. 2008), despite the existence of many biological assessment tools.

Demonstrating a causal relationship between management actions and a response in river systems is challenging. Common difficulties include natural variability, restricted opportunity to perform rigorous assessment experiments (such as those that use before-after-control-impact (BACI) designs, especially where 'before' data are not available), lack of true replication and randomization, and the presence of uncontrolled confounding factors (Beyers 1998; Downes et al. 2002; Norris et al. 2005). In many situations faced by environmental managers these problems cannot be rectified by simply applying the best possible study design because the information for a rigorous study design is missing (e.g. where 'before' data are not available because the putative impacts have already occurred) (Downes et al. 2002). Furthermore, the results of a single field-study are unlikely to be generally applicable (or are at the wrong spatial or temporal scale), and may not improve confidence to reach similar conclusions at new locations (such as different river systems) or under different environmental conditions. These difficulties can weaken the ability to infer with confidence that an observed biological impairment or recovery is caused by human activities (Downes et al. 2002). Bioassessment can demonstrate impairment or improved conditions but demonstrating a causal link to management actions may need to be based on argument (Beyers 1998).

Nevertheless, environmental managers worldwide are increasingly required by 'evidence-based policy' (Tomlinson and Davis 2010) to employ 'best available science' (Ryder et al.

2010; Murphy and Weiland 2011) to underpin 'evidence-based practice' (Pullin and Stewart 2006). Water reform policy in Australia (SEWPaC 2012b), the Clean Water Act in the US (USEPA 2012a), and in Europe the Water-Framework-Directive (EU_Water_Framework_Directive 2012) are all examples that advocate the integration of best research evidence into management decisions. The assembly of evidence from existing science publications for use in support of evidence-based practice could be cost effective. Such an approach was developed for environmental causal assessment (Nichols et al. 2011; Norris et al. 2012) by adapting 'causal criteria analysis' techniques used by epidemiologists (described in detail below and Chapters 10 and 11). This new method has the potential to aid the transfer of research into practice and improve management outcomes for freshwater systems and the environment more generally.

Evidence-based medical practice

Evidence-based practice is well established in other disciplines, particularly medicine and public health (Straus et al. 2005b). Evidence-based practice in medicine is defined as:

''*…the integration of best research evidence with clinical expertise and patient values*'' to achieve best possible patient management (Straus et al. 2005b, p. 1).

The process of evidence-based practice in medicine usually comprises five steps (Straus et al. 2005b; Thyer and Myers 2011):

1. convert the need for information into an answerable question;
2. assemble the best evidence to answer that question;
3. critically evaluate the validity, impact, and applicability of that evidence;
4. integrate relevant evidence with expertise and client values and circumstances; and
5. evaluate the expertise in conducting steps 1–4 above, and consider refinement.

Evidence-based practice in public health has achieved improvements to patient outcomes (see Pullin et al. 2009). The term 'evidence-based medicine' was formally introduced in 1992 (Evidence-Based Medicine Working Group, see Brownson et al. 2011) with its origins in the work of Cochrane (Cochrane 1972). Initiatives like the Cochrane Collaboration (http://www.cochrane.org) have used systematic literature reviews of interventions to drive an ‘effectiveness revolution’ through the incorporation of best scientific evidence into clinical practice (Stevens and Milne 1997).

Causal criteria approaches are used in epidemiology to assess the evidence for causation where lack of experimental data or the reliance on few studies would otherwise provide weak evidence and reduce the ability to draw inferences about cause and effect in patient health (Hill 1965; Weed 1997; Tugwell and Haynes 2006). The causal criteria are a ‘checklist’ against which hypothesized cause–effect associations are assessed. Such criteria have their philosophical basis in the Henle and Koch postulates for inferring causes of disease (Evans 1976). A well-known case study using causal criteria was the 1964 report prepared for the US Surgeon General on the health effects of smoking (USDHEW 1964). That landmark use of systematic criteria built a convincing argument to establish the link between tobacco smoking and lung cancer. The best-known set of epidemiological causal criteria was that developed by Hill (1965), which includes strength of association, consistency of association, specificity of association, temporality, biological gradient, biological plausibility, coherence, experimental evidence, and analogy. Alone, these different ‘levels’ of evidence are considered weak evidence but combined and used with multiple ‘lines of evidence’ they build a stronger argument for a causal link (Downes et al. 2002; Norris et al. 2005).

Causal criteria assessments seek to identify cause–effect relationships that can then be managed and its use is most common in medicine and public health, where systematic review of multiple studies in the research literature provides the ‘evidence’ (Greenhalgh 2010). Meta-analysis is another cross-study synthesis technique familiar to both medical and environmental sciences (Osenberg et al. 1999) but causal criteria is more about how data and information are assembled in the first instance (Downes et al. 2002).

Growing interest has emerged in applying the principles of epidemiology to assemble evidence and build convincing arguments regarding ecological questions of cause and effect (Fox 1991; Beyers 1998; Downes et al. 2002; Suter et al. 2002; Adams 2005; Plowright et al. 2008; Suter et al. 2010). However, such evidence-based practice has not been adopted as a paradigm in environmental science or management despite recent calls in conservation biology to implement such evidence-based methods (Pullin et al. 2009). Novel (and robust) methods are required to assess cause–effect hypotheses concerning river rehabilitation and the effects of human activities, particularly if legally challenged or where management decisions to balance river health with economic or social considerations are contested.

Presenting environmental evidence to reduce uncertainty

The premise underlying the causal criteria approach is to present evidence in such a way as to reduce inferential uncertainty. Environmental study designs lie on a gradient of inferential uncertainty. For example, an 'after impact only' design is likely to provide less convincing evidence than a BACI design (Green 1979; Eberhardt and Thomas 1991; Downes et al. 2002). More weight in an argument should be given to evidence from study designs that reduce our uncertainty (that is studies that have controls, randomization, before data, and replication) because they increase our ability to detect variation outside what is considered normal. With the additional support from multiple lines of evidence, confidence builds in our ability to make informed environmental management decisions (Reynoldson et al. 2002; Suter et al. 2002).

In freshwater ecology, such multiple lines of evidence could come from different biological assemblages such as invertebrates, fish, and algae or relate to different effects within a single group such as invertebrate community structure, biomagnification in invertebrate tissues, and laboratory toxicity tests (Grapentine et al. 2002). The evidence for causal analysis could be sourced from field-based studies and experiments, laboratory trials, in peer reviewed journals and unpublished studies. The multiple lines and levels of evidence approach (Norris et al. 2005) is more than a 'weight of evidence' derived from multiple effects and multiple studies (Burton et al. 2002; Chapman et al. 2002) because the evidence must be shown to be plausible and must consider evidence both for and against the hypothesis, thus showing that an alternative hypothesis is unlikely (Downes et al. 2002).

Environmental use of causal criteria

The first use of causal criteria (levels of evidence) and multiple lines of evidence (different effects) in a freshwater situation was in the assessment of the risks of toxicants (Menzie et al. 1996; Gilbertson 1997; Suter et al. 2002). Relatively few ecological case studies that apply causal criteria exist and the criteria employed among them lack consistency and clarity in their definitions (Lowell et al. 2000; Cormier et al. 2002; Downes et al. 2002; Norton et al. 2002; Collier 2003; Fabricius and De'Ath 2004; Norris et al. 2005; Burkhardt-Holm and Scheurer 2007; Haake et al. 2010; Wiseman et al. 2010). The lack of a consistent and standardized method for causal criteria analysis may be a contributing factor in environmental science lagging behind other disciplines (such as medicine) in the systematic assessment of published evidence to improve management outcomes. Standardized, transparent and defensible methods would assist in the application of causal criteria in synthesizing the different pieces of evidence to reduce inferential uncertainty and increase confidence in conclusions.

If managers of freshwater ecosystems, and the environment more generally, wish to make better use of the extensive published research to guide evidence-based decisions they need to cope with a continuously increasing rate of research publications. Results of topic searches of Web of Science using the string 'freshwater' or 'rivers' and 'assessment' and another using 'pollution', indicates the expansion in research publications over recent years (it is acknowledged that all search results may not be entirely relevant to the topic) (Fig. 5). Rather than helping an investigator to make decisions, the sheer volume of literature is becoming a problem (Antezana et al. 2009). Tools to aid the synthesis of information are imperative if the ever increasing scientific research (Horsburgh et al. 2009) is to be used to improve management and outcomes for freshwater systems.

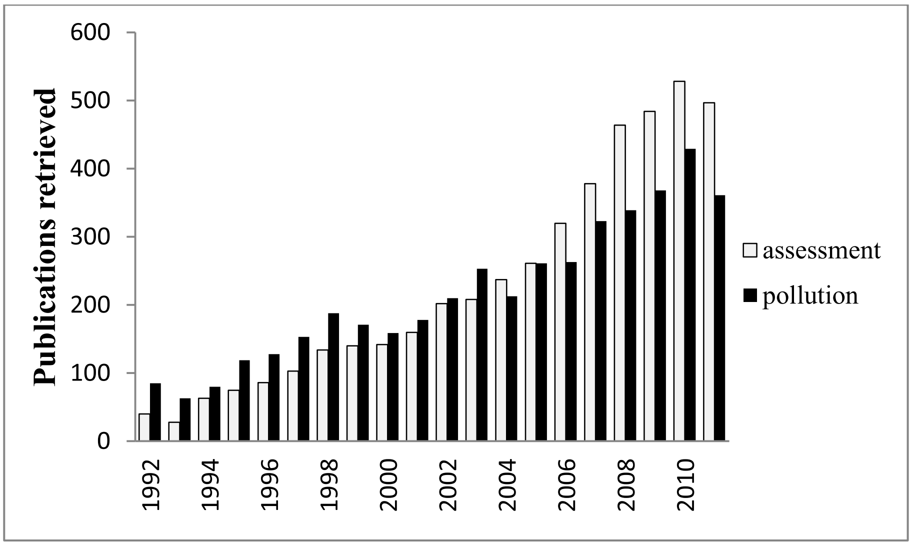


Figure 5. Results of Web of Science searches on the string 'freshwater' or 'rivers' and 'assessment' and 'freshwater' or 'rivers' and 'pollution'.

Eco Evidence tools

A causal criteria approach for environmental causal assessment was developed by adapting the techniques used by epidemiologists (Hill 1965), who faced similar difficulties in attributing causation. Named 'Eco Evidence' (Norris et al. 2012, Chapter 10), the method is a development derived from previous work (Norris et al. 2005; Nichols et al. 2011) and incorporates concepts suggested in the published literature (Beyers 1998; Downes et al. 2002; Suter et al. 2002; Adams 2005; Suter et al. 2010). This quantitative method is generally applicable to a range of environmental studies and combines information 'across criteria' to assess the overall level of support for a given cause–effect association.

As in evidence-based medicine, the Eco Evidence process begins by asking a 'quantifiable' and answerable question (Fig. 6). Only after 1) developing the question, 2) setting the context for the question, 3) building a conceptual model of plausible causal mechanisms, and 4) identifying the relevant cause–effect hypotheses to investigate, is the 'evidence' searched for and extracted (5). These steps are analogous to the steps in evidence-based medicine and include iterative steps to refine the question and conceptual model by incorporating new knowledge gained through the review process (Eco Evidence Step 6). Eco Evidence then provides a method to weight and synthesize the evidence (Steps 7 and 8). It is here that this research has made a major contribution to advancing environmental causal assessment (see Chapter 10).

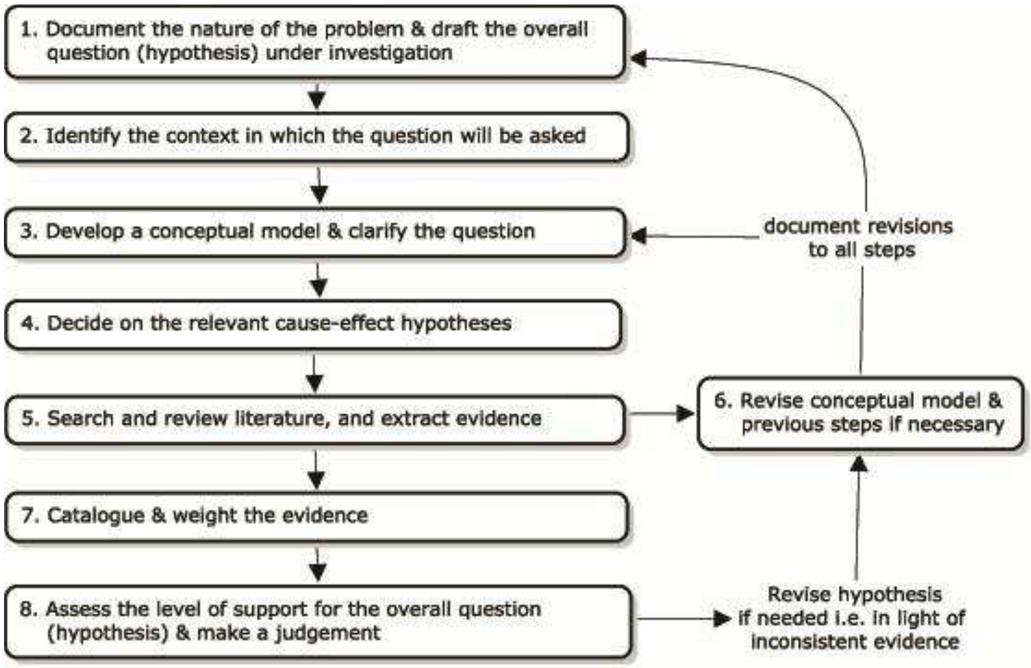


Figure 6. Steps in the Eco Evidence framework to format, extract, compile and evaluate evidence (adapted from Nichols et al. 2011).

The evidence used within the Eco Evidence process comes from scientific studies, primarily peer-reviewed literature, but could use other evidence such as unpublished studies and reports. Eco Evidence employs a set of causal criteria to evaluate the strength of evidence and weights evidence according to study-design features that provide the greatest ability to draw inferences (also similar in concept to evidence-based medicine, Greenhalgh 2010). Environmental studies with stronger designs (e.g. BACI or gradient response designs) and more replication contribute more to the assessment of causality, but weaker evidence is not discarded. The outputs of the analysis are a guide to the strength of support for or against the causal hypotheses under investigation.

The evidence integral to evidence-based practice involves far more than engaging with relevant research (Thyer and Myers 2011). Factors such as the expertise required to interpret the science, the context in which decisions are made, and the ecological values held by stakeholders will all influence the integration of scientific evidence into the decision making process (Fig. 7). However, the evidence will not inform practice or decision making unless it forms a compelling argument that will reduce uncertainty and provide the confidence to make the 'right' decision given competing or conflicting options. Without transparency and defensibility, anecdote may easily outweigh experimental or empirical evidence (Brownson et al. 2006; Brownson et al. 2011). Eco Evidence provides transparency through documentation of all steps, from the conceptual model of cause–effect linkages through to the synthesis, nature and source of the evidence. Basing the argument on sound, justified, scientific evidence and a repeatable analysis achieves defensibility.

I am not suggesting that the causal criteria assessment approach is an alternative to conducting well designed monitoring and bioassessment studies. Rather, it is complementary where both may be used to guide evidence-based actions that have previously been effective or to establish confidence regarding cause–effect linkages in different situations. The research presented in Chapters 10 and 11 describes the Eco Evidence method and draws together some lessons learned about the application of causal analysis for the systematic review of environmental literature.

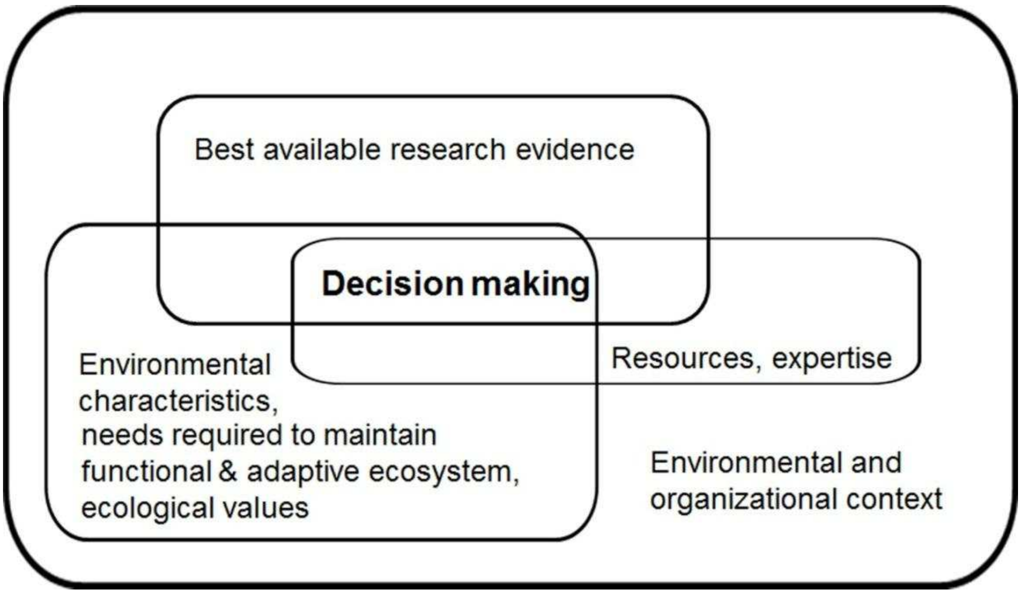


Figure 7. Domains that influence evidence-based decision making (redrawn and modified for environmental context from Brownson et al. 2011).

Chapter 2: Contribution and significance of published papers

I worked extensively with the Cooperative Research Centre for Freshwater Ecology and University of Canberra teams that researched and developed AUSRIVAS. That work contributed towards the development and testing of standardized methods, predictive models, software, and user training for a method now used as the national standard to assess the biological health of Australian rivers (eWaterCRC 2012b). However, new methods to aid the synthesis of ecological studies are imperative if the increasing body of scientific research is to improve management and outcomes for freshwater systems. My further research, in association with eWater CRC, focused on adapting causal criteria as used in epidemiology, for use in assessing evidence from multiple ecological studies for environmental causal assessment (eWaterCRC 2012a). This thesis therefore, reflects my most significant contributions to the field of ecological assessment as part of the output of the research group under supervision of Professor Richard Norris. Nine of my peer-reviewed research articles comprise the body of this thesis.

These papers, published in ERA ranked journals and one book chapter, make a significant scholarly contribution to knowledge in the field of ecological assessment. Two publications (Chapters 3 and 4) evaluate the effect of different AUSRIVAS sampling methods on bioassessment results. Five of the articles (Chapters 5–9) describe the research associated with various applications of the AUSRIVAS method. Another two publications (Chapters 10 and 11) document the new Eco Evidence method and its application for synthesizing the research outputs from multiple studies to assist causal assessment. I have arranged these articles in three categories: 1) AUSRIVAS sampling method evaluation; 2) applications of AUSRIVAS; and 3) the synthesis of multiple studies for environmental assessment. The following sections outline the principal significance of the findings of each paper and identify links between my published research outputs.

2.1 AUSRIVAS sampling method evaluation (Chapters 3 and 4)

Sample variability influences on the precision of predictive bioassessment (Chapter 3)

This journal paper (Nichols et al. 2006b), published in Hydrobiologia (impact factor 1.964), makes a major contribution to understanding the consequences of sample variability on invertebrate-based bioassessment results. This research is highly relevant to researchers and practitioners designing studies to detect river condition impairment, or recovery from disturbance or rehabilitation. A common misconception in the use of AUSRIVAS is that its use negates the need for replication and robust study design, or that the method always advocates the collection of a single sample per site regardless of the study objectives. This misconception may have resulted from its use as a broad-scale assessment tool, where in fact the sampling of many sites achieved the desired replication (e.g. the First National Assessment of River Health, SEWPaC 2012a).

The specific objectives addressed in this study were to:

1. analyse the effects of sampled area on the AUSRIVAS bioassessment of an individual site;
2. assess the site-scale variability of bioassessment results and the confidence that can be placed in the standard, single collection from reference and impaired sites; and
3. for regional assessments (such as catchment-scale studies), where the sites provide the level of replication, a statistical sub-sampling and solver algorithm was developed to determine the number of sites required to detect impairment based on sample variability.

As discussed in Section 1.4, the standard AUSRIVAS sample is a 200 animal sub-sample of each invertebrate collection from 10 metres of specific habitat. This study found that the error from size of the area sampled was negligible compared to the variability resulting from sub-sampling. If assessing at the site scale (river reach), investigators should consider replicating both the number of samples collected and number of sub-samples for more accurate results. For example, collect three samples with two 200-animal sub-samples from each or two samples per site with three sub-samples taken.

However, when assessing river condition at broader scales, such as for detection of land-use impacts, many single-site samples should be combined for the assessment. Replication within a 'treatment' (e.g. a land-use such as agriculture), where sites are the desired level of replication, avoids the need for within-site replication. If assessing at broad, regional scales (catchment or multiple catchments) the distribution of sites should encompass as much spatial variability as possible. For such studies, replicated sampling within sites may be a waste of resources.

These results are important to environmental managers in terms of allocating resources for bioassessment and the level of replication required for detecting an ecological response. My use of replicated sampling and statistical analysis described in this paper demonstrated that sample replication should be maximized at the spatial scale required for reporting (e.g. at the river-reach, catchment or larger regions). As a rule, land-use studies at broad (regional or catchment) scale should maximize replicate river reaches and river-reach (site) scale assessments should maximize replication of sample locations within reaches.

River condition assessment may depend on the sub-sampling method: field live-sort versus laboratory sub-sampling of invertebrates for bioassessment (Chapter 4)

The aim of this study (Nichols and Norris 2006), published in Hydrobiologia (impact factor 1.964), was to test the effect of live-sorting and laboratory-sorting on the bioassessment results from AUSRIVAS predictive models.

Within regions (such as states and territories), the AUSRIVAS sampling methods are standardized but they can vary between regions. The original bioassessment manual allowed enough flexibility for the different states and territories to determine their own specific needs when developing their bioassessment protocol (Davies 1994). For example, standard invertebrate sample collection methods, site selection, taxonomic resolution and data analysis were used (Davies 1994; Coysh et al. 2000; Ransom et al. 2000) but some states used different sorting strategies to sub-sample ~200 invertebrates from each sample (Nichols et al. 2000; Turak et al. 2004). The two different sub-sampling methods involved live-sorting 200 animals in the field and laboratory-sorting 200 animals using a microscope. Many across-border studies use two different sub-sampling techniques to take advantage of the most appropriate predictive model to assess site condition. This raised the question: do data collected in these different ways produce comparable site assessments.

In this study (Nichols and Norris 2006), live-sort and lab-sort predictive models were developed for the NSW South Coast region to compare same-site assessments. The results showed a distinct method bias in invertebrate samples and this paper makes a significant contribution in documenting the consequences of deviating from a standardized sampling protocol. The samples collected from the same site using different sub-sampling methods were not comparable in their invertebrate composition or richness. The live-sort sub-samples tended to have more of the large and conspicuous invertebrates and often fewer of the small and cryptic animals that were more likely to be found in lab-sort samples (where a microscope was used). The sub-sampling methods need standardization within and among studies if the raw biological data are to be comparable.

Consequently, the taxonomic differences in composition mean that the live-sort data will not provide a valid site assessment if used in a predictive model created from lab-sort reference data (and vice versa). The mismatched model would not predict the correct reference assemblage for comparison with the observed assemblage. However, when data and models were matched, the different sub-sampling methods resulted in few method-related differences in condition assessments (as described in Table 1, Chapter 4). Matching of data and models means that live-sort data are used in a predictive model created from live-sort data and lab-sort data are used in a predictive model created from lab-sort data. Some sites received a different condition assessment from the two methods and the paper discusses the role of seasonal differences in invertebrate assemblages to explain the disparity.

The major finding that data from the same site derived through different sampling methods resulted in few method-related differences in assessment when using an 'appropriate model', confirms the value of using model-derived outputs in site assessments. This finding reveals the importance of standardizing results when using different sub-sampling methods across jurisdictional boundaries, such as for national State of the Environment reporting.

2.2 Applications of AUSRIVAS (Chapters 5–9)

Ecological effects of serial impoundment on the Cotter River, Australia (Chapter 5)

This research article (Nichols et al. 2006a), published in the journal Hydrobiologia (impact factor 1.964), quantified the physical, chemical and biological effects associated with serial impoundments (three dams) of the Cotter River, in the Australian Capital Territory. The major contribution of this research was the use of AUSRIVAS models to predict pre-dam biota.

Most previous applications of AUSRIVAS were for broad-scale surveillance of river condition. This paper is highly relevant to river managers wanting to apply AUSRIVAS because it demonstrates the method's utility to provide an assessment of river condition as a specific response to flow regulation.

The use of AUSRIVAS within an interdisciplinary framework demonstrated an approach where the interpretation of physical, chemical and biological data provided a holistic evaluation of the downstream effects of the dams. Compared to predicted reference conditions, the benthic invertebrate samples from the sites directly below the dams indicated impairment but were similar to reference condition within 4 km downstream of Bendora Dam (the middle impoundment). If the study had examined the selected chemical indicators alone, it would have found no effects of regulation. If the study had used geomorphological indicators alone, the Cotter River would have appeared to show little downstream recovery from the channel-contracting effects of impoundment. Data collected here provided an ecological-condition baseline for the Cotter River, which was useful also as a benchmark for later studies designed to compare the effects of environmental flows implemented in subsequent years (Norris and Nichols 2011).

Water quality assessment of Portuguese streams: regional or national predictive models? (Chapter 6)

AUSRIVAS biological assessment methods were adapted to develop predictive models to assess the condition of Portuguese streams (Feio et al. 2009). This paper, published in Ecological Indicators (impact factor 3.058), made a significant contribution in demonstrating the applicability and transferability of AUSRIVAS predictive modelling and data-analysis methods to streams in Portugal, where a wide variety of landscape types occur in a small area. The location and type of landscape influence a river's geomorphology and hydrology, and in turn play an important role in determining the invertebrate assemblage observed in a river.

Macroinvertebrate communities are prone to temporal and spatial variation in natural and modified streams (Linke et al. 1999; Beche and Resh 2007). Many factors contribute to the variation in distribution and abundance of benthic macroinvertebrates, such as differences in substrate size and composition (Quinn and Hickey 1990), stream size and position in the catchment, e.g. upland and lowland (Minshall et al. 1985), in-stream and riparian vegetation (Read and Barmuta 1999; Quinn et al. 2004), temperature (Jacobsen et al. 1997) and flow-related variables (Boulton et al. 1992; Scarsbrook 2002; Armitage 2006; Poff and Zimmerman 2010). Dependent on the spatial extent of sampled sites, these variables may show up as environmental gradients in invertebrate datasets (Marchant et al. 1999) and potentially be used to predict invertebrate assemblages.

This work (Feio et al. 2009) was important in determining that the regional, rather than national scale, was generally most appropriate for developing invertebrate predictive models for assessment of water quality in the Portuguese territory. Ultimately, the aim in model development is to sample sufficient reference sites from each river type within a region so that accurate comparisons can be made (Hawkins and Vinson 2000). For these Portuguese models, the variation in environmental variables used to predict invertebrate assemblages was better accounted for by the gradients within regions rather than at national scale.

This work also contributed to further research on the evaluation of seasonal patterns of Portuguese macroinvertebrate communities, and to the application of predictive modelling to other aquatic communities in Portugal, such as diatoms and macrophytes (Aguiar et al. 2011; Feio and Doledec 2012; Mendes et al. 2012; Serra et al. 2012; Almeida and Feio in press).

Using the reference condition maintains the integrity of a bioassessment program in a changing climate (Chapter 7)

This paper (Nichols et al. 2010b), published in The Journal of the North American Benthological Society (impact factor 2.974), is highly relevant to natural resource managers, because it highlights the significance of standardized sampling of fixed sites (both test and reference) over long periods. This research used an invertebrate dataset collected over 15-years from test-sites potentially affected by ski-resort activities and from a sub-set of the reference sites originally used to create an AUSRIVAS-type predictive model for Kosciuszko National Park, Australia. This research investigated whether external influences, such as climate variability, inhibited clear interpretation of the predictive model results.

Analysis showed that the invertebrate assemblage composition at reference sites remained stable through time. Thus, the original predictive model was able to accurately predict the aquatic invertebrate assemblage for comparison with current test-site data, and use it to assess stream condition. Furthermore, the biological indicators detected the effects of extreme climate-related events (such as severe drought and extensive bushfire) on streams in Kosciuszko National Park. This paper highlights the importance of continued, standardized bioassessment through time. Combined with an appropriate study design, the research distinguished the ecological effects of human activities from those attributed to climate-related influences. Pre-event data provided by long-term assessment will be equally important for assessing the effects of, and recovery from, future events that threaten river health.

Environmental flows: achieving ecological outcomes in variable environments (Chapter 8)

This book chapter (Norris and Nichols 2011) draws together extensive biological assessment research on the Cotter River, ACT. It demonstrates the value of AUSRIVAS bioassessment applied within a robust study design and an adaptive management framework in a variable environment (drought, bushfires and floods). The system understanding gained from these studies, through time, made a significant contribution to increasing the certainty of generalizations and the confidence for future management decisions when circumstances changed. Ongoing bioassessment of environmental flow releases on the Cotter River has provided ecological data and improved understanding to help managers balance water supply demands and environmental water needs.

Of major importance in this research, and a main component of the adaptive management approach, was the implementation of biological assessment within a study design that could cope with changing questions and unforeseen events, such as extended drought and bushfire (also see Peat et al. 2005; Peat and Norris 2007). Within this framework, AUSRIVAS methods and the measurement of other biological components demonstrated the ecological success of the environmental-flows program (ACT-Government 2006). Assessment confirmed that desired changes to aquatic invertebrate assemblage structure and algae growth were achieved, while reducing the overall volume of water released as environmental flows under severe drought conditions (see Chapter 9 for a detailed case study, White et al. 2012). Chapter 8 documents the research into the effects of managed flows on water quality and ecological condition of the Cotter River, which has made a major contribution to the knowledge-base of scientific information to inform revisions to ACT's environmental-flow guidelines (Nichols et al. 2009; Nichols et al. 2010a).

More for less: a study of environmental flows during drought in two Australian rivers (Chapter 9)

This research (White et al. 2012), published in Freshwater Biology (impact factor 3.082), measured instream biotic responses to various environmental-flow releases, manipulated to achieve beneficial ecological outcomes for the Cotter River, ACT, during extended drought conditions. The types of disturbance may be described as pulse, press and ramp (Lake 2000). Floods are usually pulses, sedimentation after intense bushfires in the catchment may be a press, and a steadily increasing multi-year drought may be considered a ramp disturbance (Lake 2000). In freshwater systems, there appear to be two types of droughts: short and periodic, and the longer, highly unpredictable multi-year droughts (Lake 2003). Studies on the ecological effects of extended drought on streams are limited by their infrequency (Humphries and Baldwin 2003), the paucity of pre-drought data and the absence of reference sites unaffected by drought (Boulton 2003).

This paper provided vital information that contributed to advancing scientific understanding to underpin water management decisions regarding environmental flow releases in the Cotter River during an extended drought. This research applied bioassessment to evaluate the minimum environmental-flow requirements needed to maintain the invertebrate assemblages and algae in a healthy condition. Maintaining ecosystem resilience under changing conditions is a worthy management goal. Resilience is lost if a system moves into an alternate state that passes an ecological threshold (Folke et al. 2004; Garmestani et al. 2009). Modifying the environmental flow regime achieved short-term ecological objectives, such as reduced periphyton accumulation and increased habitat availability, while securing Canberra’s water supply under severe drought conditions. This application of AUSRIVAS showed that the management actions maintained the river’s resilience to recover when higher flows returned.

2.3 Synthesizing multiple studies (Chapters 10 and 11)

Analyzing cause and effect in environmental assessments: using weighted evidence from the literature (Chapter 10)

The major contribution of this paper (Norris et al. 2012), published in Freshwater Science (impact factor 2.974), was to document the 'Eco Evidence' method, a causal criteria approach for assessing the evidence for and against environmental cause–effect hypotheses. This paper was published in the 'Perspectives' section of the journal reserved for the expression of new ideas, points of view, and comments on topics of interest to aquatic scientists. The Eco Evidence weighting system for individual studies is a major advancement in environmental causal assessment.

The Eco Evidence analysis method is sometimes confused with a meta-analysis, which is another cross-study synthesis technique (Osenberg et al. 1999). Meta-analysis involves estimating a collective effect size across a number of studies (Gurevitch and Hedges 2001; Sutton and Higgins 2008). However, unlike Eco Evidence, there is no requirement in meta-analysis to assess the plausibility of the hypothesized cause—effect association (Weed 2000). Further, the synthesis of environmental science studies often fails to provide the summary statistics necessary for meta-analysis (Greet et al. 2011). A major advantage of the Eco Evidence approach compared with meta-analysis is that Eco Evidence allows the inclusion of a greater range of literature in an analysis. This is important because environmental evidence on specific cause–effect relationships is often scarce. Indeed, where a meta-analysis is possible it could provide valuable evidence for use within Eco Evidence.

This paper is highly relevant to other researchers and environmental practitioners needing to use a method for quantifying and combining the evidence from multiple studies for ecological causal assessment. This paper is the first high-impact journal publication to describe the Eco Evidence method and is an important step in facilitating broader use of systematic methods to assess cause–effect relationships in environmental sciences.

Ecological responses to flow alteration: assessing causal relationships with Eco Evidence (Chapter 11)

This paper (Webb et al. 2012), published in the Wetlands journal (impact factor 1.238), reiterates the Eco Evidence method and provides a critical evaluation of the method for systematic literature review of evidence for and against ecosystem responses (vegetation, invertebrates and geomorphology) to flow within the Murray-Darling Basin.

There have been few applications of causal criteria analysis in environmental science. This may partly result from the previous lack of standardized methods and analysis tools. The causal criteria method (Nichols et al. 2011; Norris et al. 2012) is now implemented as freely available 'Eco Evidence' software (eWaterCRC 2011; Webb et al. 2011; eWaterCRC 2012a). It comprises two components: an Eco Evidence Database, which is a web application for storing and sharing 'evidence items' (the information extracted from individual studies necessary for the causal criteria analysis) and the Eco Evidence Analyser (Wealands et al. 2009; Webb et al. 2011; Norris et al. 2012; Webb et al. 2012). The Eco Evidence Analyser is desktop analysis software based on the causal criteria analysis method (see Chapter 10) that can use evidence shared via a web application.

Working with Eco Evidence, users can search and access a reusable 'knowledge bank' to obtain a list of citations relevant to specific cause–effect associations. It also provides condensed information extracted from scientific papers on which to base a systematic literature review or causal assessment. Furthermore, users can access evidence entered by previous users, thereby reducing the burden of extracting evidence from the literature. This research effort is part of a worldwide trend towards facilitating greater use of evidence-based methods in environmental management, and the tools described here are contributing to change the way scientific evidence is used to solve environmental problems.

Chapter 3: Sample variability influences on the precision of predictive bioassessment

The nature and extent of my contribution to the work was as follows:

Nature of my contribution	Extent of contribution (%)
Research, data collection, data analysis, and led the writing of the manuscript.	75

Chapter 3

This chapter has been removed due to copyright restrictions.

This chapter is available as:

Nichols, Susan J. Robinson, Wayne A. & Norris, Richard H. (2006) Richard H. Sample variability influences on the precision of predictive bioassessment. *Hydrobiologia.* 572, 215-233.

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Online subscribed content (UC community)	http://ezproxy.canberra.edu.au/login?url=http://search.ebscohost.com/login.aspx?direct=true&db=a9h&AN=22530206
Online general public	http://link.springer.com/article/10.1007/s10750-005-9003-4
DOI	10.1007/s10750-005-9003-4

Abstract

The rapid bioassessment technique we investigate (AUSRIVAS) requires a nationally standardized sampling protocol that uses a single collection of macroinvertebrates (without replication) taken from 10 m of specific habitats (e.g. stream edge and/or riffle) and sub-samples of 200 animals. The macroinvertebrate data are run through predictive models that provide an assessment of biological condition based on a comparison of the animals found in the collection (the observed) and those expected to be there given the site-specific characteristics of the stream (the O/E taxa score). The important questions are related to the conclusions regarding river condition that can be drawn from the biological assessment. Rapid bioassessment studies are generally of two types: those for assessment of individual sites and those where many sites are selected to collectively assess the potential impacts of some human activity such as forestry or agriculture. We wanted to identify the effects of sample variability on the outputs of this predictive bioassessment technique. We found that a single collection of benthic macroinvertebrates was sufficient for bioassessment when taken from a site that had a large area of nearly uniform substrate and was in good condition. Also, collections taken from a larger and smaller area of substrate (1.75, 3.5 or 7 m2) gave the same bioassessment. In other sites, not in such good condition, the variability in bioassessment from different collections could result in different interpretations of biological condition. For all sites, regardless of condition, much of the variation in bioassessment was derived from sub-sampling the macroinvertebrates. We develop a statistical sub-sampling and solver algorithm that provides a measure of variability and a statistically valid probability of impairment for a single site, without the need to actually collect the hundreds of replicated collections needed for this study. We found that assessment at impaired sites, where only 1 collection and 1 sub-sample are taken (a

common situation in rapid assessment), the 95% confidence level for O/E taxa scores is estimated to be as much as ±0.22. At sites in reference condition, the 95% confidence interval may be much narrower (~±0.1 O/E units). Therefore, assessments of sites at, or near, reference condition will be more precise than for impaired sites. Power analysis revealed that where single sites are being assessed we recommend a sample collected from 3.5 m2 of habitat, but replicate collections should be taken at a site (rather than one only) and we recommend replicate sub-samples of each collection (total of six sub-samples from a site). However, this would remove a 'rapid' component of the bioassessment. We recommend the addition of sub-sampling and solver algorithms to the predictive models such as AUSRIVAS to provide a statistical measure of probability of impairment. An adaptive sub-sampling regime could then be used to optimize sampling effort. For example, a single sub-sample may be sufficient for screening or the agency could use the sub-sample and solver algorithms to sub-sample the parent sample for a more precise estimate of the biological condition. Replication should be maximized at the spatial scale required for reporting: site, or regional. But as a general rule, catchment or land-use scale studies should maximize replicate sites, and site-scale assessments should maximize replication within sites.

Chapter 4: River condition assessment may depend on the sub-sampling method: field live-sort versus laboratory sub-sampling of invertebrates for bioassessment

The nature and extent of my contribution to the work was as follows:

Nature of my contribution	Extent of contribution (%)
Research, data collection, predictive modelling, data analysis, and led the writing of the manuscript.	90

Chapter 4

This chapter has been removed due to copyright restrictions.

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Nichols, Susan J. & Norris, Richard H. (2006) River condition assessment may depend on the sub-sampling method: field live-sort versus laboratory sub-sampling of invertebrates for bioassessment. *Hydrobiologia.* 572, 195-213.

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Online subscribed content (UC community)	http://ezproxy.canberra.edu.au/login?url=http://search.ebscohost.com/login.aspx?direct=true&db=a9h&AN=22530201
Online general public	http://link.springer.com/article/10.1007/s10750-006-0253-6
DOI	10.1007/s10750-006-0253-6

Abstract

Aquatic macroinvertebrates are commonly used biological indicators for assessing the health of freshwater ecosystems. However, counting all the invertebrates in the large samples that are usually collected for rapid site assessment is time-consuming and costly. Therefore, sub-sampling is often done with fixed time or fixed count live-sorting in the field or with preserved material using sample splitters in the laboratory. We investigate the differences between site assessments provided when the two sub-sampling approaches (Live-sort and Lab-sort) were used in conjunction with predictive bioassessment models. The samples showed a method bias. The Live-sort sub-samples tended to have more large, conspicuous invertebrates and often fewer small and, or cryptic animals that were more likely to be found in Lab-sort samples where a microscope was used. The Live-sort method recovered 4–6 more taxa than Lab-sorting in spring, but not in autumn. The magnitude of the significant differences between Live-sort and Lab-sort predictive model outputs, observed to expected (O/E) taxa scores, for the same sites ranged from 0.12 to 0.53. These differences in the methods resulted in different assessments of some sites only and the number of sites that were assessed differently depended on the season, with spring samples showing most disparity. The samples may differ most in spring because many of the invertebrates are larger at that time (and thus are more conspicuous targets for live-sorters). The Live-sort data cannot be run through a predictive model created from Lab-sort data (and vice versa) because of the taxonomic differences in sub-sample composition and the sub-sampling methods must be standardized within and among studies if biological assessment is to provide valid comparisons of site condition. Assessments that rely on the Live-sorting method may indicate that sites are 'less impaired' in spring compared to autumn because more taxa are retrieved in spring when they are larger and more visible. Laboratory sub-sampling may return fewer taxa in spring, which may affect assessments relying on taxonomic richness.

Chapter 5: Ecological effects of serial impoundment on the Cotter River, Australia

The nature and extent of my contribution to the work was as follows:

Nature of my contribution	Extent of contribution (%)
Research, data collection, data analysis, and led the writing of the manuscript.	70

Chapter 5

This chapter has been removed due to copyright restrictions.

This chapter is available as:

Nichols, Susan, Norris, Richard, Maher, William & Thoms, Martin (2006). Ecological effects of serial impoundment on the Cotter River, Australia. *Hydrobiologia.* 572, 255-273.

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Online subscribed content (UC community)	http://ezproxy.canberra.edu.au/login?url=http://search.ebscohost.com/login.aspx?direct=true&db=a9h&AN=22530215
Online general public	http://link.springer.com/article/10.1007/s10750-005-0995-6
DOI	10.1007/s10750-005-0995-6

Abstract

This study examines the ecological effects of serial impoundments (three dams) on a rocky upland stream in southeastern Australia. Physical, chemical and biological changes were quantified and interpreted within a three-level hierarchy of effects model developed previously by Petts [1984, Impounded Rivers. John Wiley and Sons, New York] and the Australian Rivers Assessment System (AUSRIVAS) to predict pre-dam biota. First-order effects were decreased median monthly discharges and floods of lesser magnitude following construction of the dams. No effect on water characteristics (pH, electrical conductivity and major ions) was evident. The second-order effect on channel morphology was a decrease in bank-full cross-sectional area by up to 75% because of reduced flows. At all sites, the predominantly cobble streambed was armoured and generally highly stable. The discharge required to initiate movement of the streambed surface sediments (38.9 $m^3\ s^{-1}$) was 40% less frequent since construction of the dams, implying alteration to the natural disturbance regime for benthic biota. Benthic algal growth appeared more prolific at sites directly below dams. Fewer macroinvertebrate taxa than expected and modified assemblages within 1 km of all three dams were third-order effects. Compared to reference conditions, macroinvertebrate samples from the sites directly below the dams had relatively more Chironomidae larvae, Oligochaeta and Acarina, and fewer of the more sensitive taxa, Plecoptera, Ephemeroptera, Trichoptera and Coleoptera. Biological recovery to the macroinvertebrate assemblage was evident within 4 km downstream of the second dam.

Chapter 6: Water quality assessment of Portuguese streams: regional or national predictive models?

The nature and extent of my contribution to the work was as follows:

Nature of my contribution	Extent of contribution (%)
Intellectual contribution as advice and assistance with data analysis, site classification and predictive modelling, and contributed to writing and editing of the manuscript.	25

Chapter 6

This chapter has been removed due to copyright restrictions.

This chapter is available as:

Feioa, M.J., Norris, R.H., Graça, M.A.S., Nichols, S. (2009). Water quality assessment of Portuguese streams: Regional or national predictive models? *Ecological Indicators.* 9(4) 791-806.

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Online general public	http://www.sciencedirect.com/science/article/pii/S1470160X08001258
DOI	http://dx.doi.org/10.1016/j.ecolind.2008.09.012

Abstract

The European Water Framework Directive (WFD 2000) brought the need in European Union countries to establish consistent quantitative methods for the water quality assessment of streams, using aquatic communities. With this work we aimed to develop predictive models using macroinvertebrate communities that could be used in Portugal as an alternative to the more traditional indices and metrics. We used data from 197 reference sites and 174 sites suspected of being impaired, which were obtained in a national survey conducted in 2004–2005 by the Instituto da Água (INAG, Portugal). The spatial scale at which to develop predictive models was an issue to address because the Portuguese territory covers a wide variety of landscapes in a small area. We built three models using the AUSRIVAS methods, a national and two regional (North and South) models that produced acceptable assessments. However, the regional models, predicted more taxa than the National model, were more accurate and had lower misclassification errors when placing sites into pre-defined groups. The regional models were also more sensitive to some disturbances related to water chemistry (e.g., nutrients, BOD5, oxidability) and land use. The exception was for the northern costal area, which had few reference sites. In the northern costal area the National model provides more useful results than the regional model. The 5-class WFD quality assessment scheme, adapted from the AUSRIVAS bands, appears to be justified because of the good correspondence between the human disturbance level and the classes to which test sites were allocated. Elimination of the AUSRIVAS X band in the WFD scheme has produced a clearer relationship. The predictive models were able to detect a decline in river health, responded to several causes of degradation and provided site-specific assessments.

Chapter 7: Using the reference condition maintains the integrity of a bioassessment program in a changing climate

The nature and extent of my contribution to the work was as follows:

Nature of my contribution	Extent of contribution (%)
Research, data collection, data analysis, and led the writing of the manuscript.	75

Chapter 7

This chapter has been removed due to copyright restrictions.

This chapter is available as:

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Online general public	http://www.jnabs.org/doi/abs/10.1899/09-165.1
DOI	10.1899/09-165.1

Abstract

Climate change is gradual and long-term, consistently collected data are required to detect resulting biological responses and to separate such responses from local effects of human activities that monitoring programs usually are designed to assess. The reference-condition approach is commonly used in freshwater assessments that use predictive modeling, but a consistent reference condition is required to maintain the relevance and integrity of results over the long term. We investigated whether external influences, such as climate change, inhibited clear interpretation of bioassessment results in a study design using reference vs test sites. Macroinvertebrates were collected from 16 sites (11 sites affected by ski resorts and 5 reference sites) on 5 streams in 4 seasons each year from 1994 to 2008 within Kosciuszko National Park, Australia. We analyzed trends over 15 y to address questions regarding climate-change and macroinvertebrate bioindicators of stream condition (observed/expected [O/E] taxa; Stream Invertebrate Grade Number Average Level [SIGNAL] 2 scores; Simpson's Diversity; Ephemeroptera, Plecoptera, Trichoptera [EPT] richness ratio; and Oligochaeta abundance). Climate became slightly warmer and less humid ($p < 0.0001$), but no significant relationships between climate variables and bioindicators were evident. All bioindicators consistently distinguished between test and reference sites in all seasons. All bioindicators except for O/E taxa scores differed among streams (regardless of site type). O/E taxa are inherently adjusted for specific stream characteristics, and, thus, were robust to differences in stream type while remaining sensitive to reference and test site variation. Generally, reference and test sites did not respond differently to any gradual climate changes. Furthermore, the reference sites sampled through time remained in a condition equivalent to the previously defined reference condition and provided a valid comparison for current test sites of unknown condition. The bioindicators used here were insensitive to the small but significant changes in climate detected over the 15-y study. However, extreme climate-related events (such as severe drought and extensive bushfire) were detected by the chosen bioindicators at both reference and test sites. Ecological outcomes of climate change can be accounted for only by an appropriate study design that includes standardized sampling of fixed sites (both test and reference) over long periods.

Chapter 8: Environmental flows: achieving ecological outcomes in variable environments

The nature and extent of my contribution to the work was as follows:

Nature of my contribution	Extent of contribution (%)
Research, data collection, data analysis and significant contribution to writing and editing of the manuscript.	50

Chapter 8

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Online subscribed content (UC community)	http://canberra.eblib.com.au/patron/FullRecord.aspx?p=647376
Online general public	http://trove.nla.gov.au/version/170544094
Print general public	http://trove.nla.gov.au/version/50480139

Abstract

Environmental flows were implemented in the Cotter River in 1999 as a requirement of the Australian Capital Territory (ACT) Water Resources Act. A multi-disciplinary group composed of representatives from a water utility, ACT government, and research organisations was formed to manage the Cotter River environmental flows program, aiming to achieve specified ecological outcomes and increased water security through adaptive management. Based on scientific knowledge, changes were made to the delivery of environmental flows after drought in 2002 and bushfires in January 2003. Ongoing ecological assessment formed a major component of the adaptive management approach; it informed decisions regarding the achievement of desired ecological outcomes by using trial flow release strategies that involved smaller overall volumes of water. In this way, a feedback loop for the decision-making process was formed; it included a statement of the desired ecological outcomes, specified the flows needed to achieve them, how the effects would be assessed, and provided feedback to the decision makers. Another major component of the adaptive management approach was the formulation of a study design that was able to cope with changing questions and unforeseen events, such as drought and fire. The success of the environmental flows program has been demonstrated through attainment of desired changes to macroinvertebrate assemblage structure and periphyton , together with a significant reduction in the overall volume of water released as environmental flows. The value of adaptive management and collaboration between a utility, government, and researchers to achieve a balance between water supply demands and environmental water needs has also been shown.

Chapter 9: More for less: a study of environmental flows during drought in two Australian rivers

The nature and extent of my contribution to the work was as follows:

Nature of my contribution	Extent of contribution (%)
Research, data analysis, statistical analysis, intellectual input and major contribution to writing of the manuscript.	40

Chapter 9

This chapter has been removed due to copyright restrictions.

This chapter is available as:

White, Hayley L., Nichols, Susan J., Robinson, Wayne A. & Norris, Richard H. (2012). More for less: a study of environmental flows during drought in two Australian rivers. *Freshwater Biology.* 57(4): 858-873.

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Abstract

1. In rivers affected by drought, flow regulation can further reduce flow and intensify its effects. We measured ecological responses to environmental flows, during a prolonged drought in a regulated river (Cotter River), compared with a drought affected, unregulated river (Goodradigbee River) in south-eastern Australia.

2. Environmental flows in the regulated Cotter River were reduced from a monthly average base flow of 15 MLd−1 to only 5 MLd−1, which was implemented as two test flow regimes. Initially, flows were delivered in cycles of 14 days at 3 MLd−1 followed by 3 days at 14 MLd−1 and then another 14 days at 3 MLd−1 to make up the monthly average of 5 MLd−1. This flow regime continued for 6 months, after which a preliminary ecological assessment indicated deterioration in river condition. Consequently, the flow regime was altered to a cycle of 2 MLd−1 for 28 days followed by 20 MLd−1 for either 3 or 4 days. This new flow regime continued for another 5 months.

3. The ecological outcomes of the test flow regimes were assessed in terms of (i) the provision of available habitat (wetted channel) for aquatic biota; (ii) the accumulation of periphyton; and (iii) the structure and richness of macroinvertebrate assemblages.

4. Flow of 20 MLd−1 covered most of the streambed in the Cotter River, thus providing more wetted area and connectivity between habitats than flows of 2, 3 or 14 MLd−1. Depth and velocity were always less in the Cotter River than in the unregulated Goodradigbee River. Periphyton decreased in the Cotter River during the 2/20 MLd−1 flow regime, which combined the lowest and greatest test flow volumes, while periphyton did not change significantly in the unregulated river.

5. The reduced flow in the Cotter River resulted in fewer macroinvertebrates than expected (13) compared with unregulated Goodradigbee sites (19), although the magnitude of the differences did not depend on the test flow releases. Macroinvertebrates in the Cotter River became numerically dominated by Diptera and Oligochaeta, while Ephemeroptera, Plecoptera and Trichoptera decreased in abundance.

6. In the Cotter River, the monthly average flow of 5 MLd−1 (exceeded 97% of the time pre-regulation) was insufficient to maintain the macroinvertebrate assemblages in reference condition, regardless of release patterns. However, short-term ecological objectives were achieved, such as reduced periphyton accumulation and increased habitat availability, and the environmental flows maintained the river's ability to recover (resilience) when higher flows returned.

Chapter 10: Analyzing cause and effect in environmental assessments: using weighted evidence from the literature

The nature and extent of my contribution to the work was as follows:

Nature of my contribution	Extent of contribution (%)
Significant contribution to writing and editing of the manuscript, research, method development and trial (Norris et al. 2005), documentation of framework and analysis methods (methods manual Nichols et al. 2011), conceptual development of Eco Evidence Analyser software (eWaterCRC 2011), and trial of the Eco Evidence Analyser software.	30

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Chapter 10

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Abstract

Sound decision making in environmental research and management requires an understanding of causal relationships between stressors and ecological responses. However, demonstrating cause–effect relationships in natural systems is challenging because of difficulties with natural variability, performing experiments, lack of replication, and the presence of confounding influences. Thus, even the best-designed study may not establish causality. We describe a method that uses evidence available in the extensive published ecological literature to assess support for cause–effect hypotheses in environmental investigations. Our method, called Eco Evidence, is a form of causal criteria analysis—a technique developed by epidemiologists in the 1960s—who faced similar difficulties in attributing causation. The Eco Evidence method is an 8-step process in which the user conducts a systematic review of the evidence for one or more cause–effect hypotheses to assess the level of support for an overall question. In contrast to causal criteria analyses in epidemiology, users of Eco Evidence use a subset of criteria most relevant to environmental investigations and weight each piece of evidence according to its study design. Stronger studies contribute more to the assessment of causality, but weaker evidence is not discarded. This feature is important because environmental evidence is often scarce. The outputs of the analysis are a guide to the strength of evidence for or against the cause–effect hypotheses. They strengthen confidence in the conclusions drawn from that evidence, but cannot ever prove causality. They also indicate situations where knowledge gaps signify insufficient evidence to reach a conclusion. The method is supported by the freely available Eco Evidence software package, which produces a standard report, maximizing the transparency and repeatability of any assessment. Environmental science has lagged behind other disciplines in systematic assessment of evidence to improve research and management. Using the Eco Evidence method, environmental scientists can better use the extensive published literature to guide evidence-based decisions and undertake transparent assessments of ecological cause and effect.

Chapter 11: Ecological responses to flow alteration: assessing causal relationships with Eco Evidence

The nature and extent of my contribution to the work was as follows:

Nature of my contribution	Extent of contribution (%)
Method development, significant contribution to writing and editing of the manuscript, equal contribution for the first three authors.	25

Chapter 11

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Abstract

The environment is being increasingly recognized as a legitimate user of water. However, tension between environmental and consumptive uses remains and environmental water allocations may be subject to legal challenge. Current predictions of ecological response to altered flow regimes are not sufficiently transparent or robust to withstand such challenges. We review the use of causal criteria analysis to systematically review ecological responses to changes in flow regimes. Causal criteria analysis provides a method to assess the evidence for and against cause-effect hypotheses. Relationships supported by sufficient evidence can inform transparent and robust environmental flow recommendations. The use of causal criteria analysis in environmental science has been facilitated by the development of the Eco Evidence method and software—a standardized approach for synthesizing evidence from the scientific literature. Eco Evidence has thus far been used to assess the evidence concerning responses of vegetation, fish, macroinvertebrates, and floodplain geomorphology to changes in flow regime, and provides a robust and transparent assessment of this evidence. There is a growing movement internationally to shift from experience-based to evidence-based methods in environmental science and management. The research presented here is at the leading edge of a fundamental change in the way environmental scientists use evidence.

Chapter 12: Problems and future directions

In Chapter 2 I discussed the contribution of my research papers to the development and application of the AUSRIVAS method for bioassessment of river condition (Chapters 3–9) and Eco Evidence (Chapters 10 and 11). In this Chapter, I outline the various issues concerning the continued use of AUSRIVAS and discuss future directions for the development of AUSRIVAS and the reference condition approach for bioassessment in Australia. I also discuss the prospects for Eco Evidence.

12.1 Problems and future directions for bioassessment in Australia

Continued use of AUSRIVAS

The sustained use of the AUSRIVAS method continues to provide data for targeted impact assessment (e.g. Marchant and Hehir 2002; Sloane and Norris 2003; Nichols et al. 2006a; Growns et al. 2009; White et al. 2012), state/regional assessments of river condition (e.g. Turak et al. 1999; Rose et al. 2008) and community based river assessment programs e.g. Waterwatch (Davies 2007). AUSRIVAS data also provides for very broad-scale assessment at multijurisdictional and national levels, for example, the Snapshot of the Murray-Darling Basin river condition, State of the Environment reporting and the Murray Darling Basin Authority's Sustainable Rivers Audit (Turak et al. 1999; Norris et al. 2001a; Norris et al. 2001b; Davies et al. 2010; Harrison et al. 2011). AUSRIVAS outputs have been implemented in policy and used to evaluate management actions at local, state or national scales, for example the ACT Environmental Flow Guidelines (ACT-Government 2006), Victorian biological objectives for streams and rivers (EPA 2004), and the National Water Quality Guidelines (ANZECC and ARMCANZ 2000). The Minister for Environment Protection, Heritage and the Arts is required, under the Environment Protection and Biodiversity Conservation Act 1999, to table a report in Parliament every five years on the State of the Environment. For the 2011 State of the Environment report, the AUSRIVAS data were the only data with national coverage to report the instream biological condition of Australia's rivers (Harrison et al. 2011). AUSRIVAS has national significance for monitoring and assessing river condition in Australia.

A frequently asked question is whether monitoring and evaluation produce positive outcomes for our rivers (Lee and Ancev 2009). Despite broad-scale assessments indicating a decline in the health of Australia's rivers, the trend continues (Schofield 2010), notwithstanding reported increases in sustainable land-use practices (e.g. conservation farming, minimum tillage, zero tillage) (Bowmer 2011) and numerous river restoration and rehabilitation projects (Lake 2001; Bond and Lake 2005). Attempts to explain this conundrum blame confounding factors that could disguise local benefits (Bowmer 2011), such as;

- climate change, drought and climate variability;
- an increase in bushfires;
- lag times between land-use change and resulting benefits in river systems;
- variations in the extent of ecosystem resilience;
- effects of river regulation; and the influence of introduced species;
- the use of inappropriately scaled restorations; or
- inadequately designed evaluations of restoration projects that are not powerful enough to detect an ecological response.

An alternative or contributing explanation is that many river restoration projects do not constitute 'ecological' restoration and could actually degrade nearby waterways (Palmer et al. 2005). For example, a riverfront restoration designed to create recreational areas may constrain the natural functioning of the river and floodplain (Johansson and Nilsson 2002; Palmer et al. 2005). Thus, all river restoration projects will not necessarily result in an ecological success if designed to achieve some other improvement (which may be valued for other reasons), are not strongly related to cause and effect, or if evaluation studies are not adequately designed to detect ecological success.

One positive outcome of monitoring and assessment programs is their pivotal role in identifying and raising awareness of the problems associated with degraded river conditions (a necessary first step). However, resource managers now need to think to the next phase: namely, conducting rigorous, large-scale experiments within an adaptive management framework (Poff et al. 2003; King et al. 2010). Within such a framework, stakeholders will gain better understanding of environmental requirements while evaluating management actions designed to restore condition and measure ecological success. Such programs that span state borders would necessitate inter-jurisdictional cooperation and agreements, and require coordination of sampling, protocols and training, analytical methods, reporting tools, and cross-boundary standardization. An example from the Murray Darling Basin, is the

Sustainable Rivers Audit sampling protocol that provides sound data and a monitoring and assessment framework that is amenable to adaptive management to provide information at different management and reporting scales (Davies et al. 2010). The Sustainable Rivers Audit assessment program incorporates AUSRIVAS sampling methods and is expected to play a key role in future water and catchment management through integration of the assessment protocol into the Murray-Darling Basin Plan (Davies et al. 2010).

To date, evidence of positive ecological monitoring and assessment outcomes in Australia are more likely to be found where studies are designed to evaluate specific ecological objectives (see White et al. 2012). However, long-term monitoring of English rivers has shown considerable improvement in condition since 1990 (DEFRA 2012). Assessment for the large-scale State of the Environment (SoE) reporting in Australia is not currently underpinned by a well-designed national sampling program, thus changes in AUSRIVAS O/E score, sampling intensity and invertebrate taxonomic richness over time could not be reported in the 2011 SoE (Harrison et al. 2011). Therefore, quantitative comparisons with previously reported results and interpretation of temporal changes in stream biological condition were not possible (Schofield 2010). Randomized sampling sites and the use of standardized invertebrate sampling methods between SoE reporting periods would enable trend analysis. A coordinated effort is required to improve river condition, combined with large-scale assessment programs specifically designed to detect and evaluate their ecological effectiveness (Schofield 2010). Assessment tools such as AUSRIVAS when used within frameworks like the Sustainable Rivers Audit are pivotal to the success of such monitoring and assessment programs.

Use of a nationally standardized sampling protocol is strongly supported by major government agencies because it provides data sets that are comparable spatially and temporally (e.g. for SoE reporting, Harrison et al. 2011). One of the major strengths of AUSRIVAS is it allows assessment at broad scales, such as for large catchments (e.g. Murray Darling Basin), state or national level (Norris et al. 2001a). However, AUSRIVAS sampling methods do vary by state and territory (namely in the live-sorting or laboratory-sorting sub-sampling method). As demonstrated in Chapter 4 (Nichols & Norris 2006), the taxonomic differences in the composition of invertebrate sub-samples obtained using different sorting strategies mean that standardization is important within and among studies if biological assessment is to provide valid comparisons of river condition across jurisdictions. The assessments provided by the method-specific AUSRIVAS model (that is, the O/E score) can accommodate such method-related differences when different jurisdictions use different sub-sampling methods (Nichols and Norris 2006). All Australian states and territories use the AUSRIVAS sampling methods but a number of jurisdictions no longer use the AUSRIVAS O/E scores and other model outputs for river assessment (Davies 2007), which presents a potential problem for future multijurisdictional assessments of river condition that rely on AUSRIVAS.

Adoption of AUSRIVAS bioassessment by water and environment agencies was initially rapid and the support strong and widespread. Davies (2000) anticipated that the future would see continued adoption of AUSRIVAS bioassessment into state water and environmental policy and regulatory frameworks and broader adoption in a wide variety of environmental management settings, by government, community and industry. Although AUSRIVAS has national significance as a bioassessment tool, it is not currently nationally coordinated or funded to the level required to support the expansion in the breadth or continued development that was anticipated by Davies (2000) (also see Schofield 2010). Further development was intended to include biological assessment indicators other than invertebrates and the expansion in other modelling techniques (Davies 2000), and to address concerns such as the difficulties of using the AUSRIVAS approach where reference sites are problematic (Chessman et al. 2010). The lack of national coordination and funding to address AUSRIVAS research and development needs is undermining user confidence in AUSRIVAS bioassessment results (see Davies 2007 for a comprehensive report to government outlining governance and funding issues).

Adequacy of current AUSRIVAS models

User confidence in the AUSRIVAS models also relates to the adequacy of current models to detect disturbance. The research outputs within this thesis (Chapters 3–9) and studies by other researchers have reviewed and evaluated various aspects of the AUSRIVAS method and some proposed new methods (Turak et al. 1999; Marchant 2002; Metzeling et al. 2003; Hose et al. 2004; Halse et al. 2007; Gillies et al. 2009; Chessman et al. 2010). However, since the initial development of the AUSRIVAS models, few studies in Australia have critically reviewed the modelling approach and compared the advantages or limitations of various methods (Davies 2007). The current AUSRIVAS models were largely developed in the mid-1990s (Simpson and Norris 2000). New spatial tools, particularly GIS and the related access to an array of map layers describing attributes such as, geology, land use, vegetation type and climate are now available at the catchment scale (Frazier et al. 2012). Use of this landscape-scale data is a source of potential new predictor variables to combine with local site data to develop new (and potentially improved) models. Updating of models should involve careful evaluation and comparison of their performance against current models, and may need to consider alternative modelling options. Such evaluation should also be used to compare AUSRIVAS O/E to alternative assessment techniques such as other indices and metrics, environmental filters (Chessman and Royal 2004), traits-based approaches (Peru and Doledec 2010) or other assessment tools that use reference condition approach (e.g. BEAST ordination, Rosenberg et al. 2000).

Developing the predictive models used for the research described within this thesis (Chapters 3–9) employed discriminant function analysis and required identification of reference-site groups, which is a common method used to predict the probability of test-site membership to a specific reference state (Van Sickle et al. 2006). However, the method does have limitations. First, it requires the definition of groups of reference sites but in most reference databases with many sites, discrete community assemblages do not characterize the invertebrate data (Hawkins and Vinson 2000). Rather, the data structure displays sites along a continuum of one or more taxonomic gradients (Fig. 8). Each taxon's own array of environmental requirements and habitat preferences determine the gradients evident in the invertebrate datasets (Resh et al. 1994; Menezes et al. 2010). The spatial scale of sampling will also influence the underlying gradient revealed by the classification and ordination (Marchant et al. 1999) and gradients may become more obvious as the size of the reference dataset (or the density of reference site coverage) increases (Turak et al. 1999). Classifying discrete groups of sites along these gradients is a requirement of discriminant function analysis rather than a representation of the reality of the invertebrate assemblages. Other approaches may acknowledge the continuum in species distributions and avoid the use of classification groups, and instead use the ordination space of reference sites as the basis of prediction (Linke et al. 2005). However, when assessing test-site condition, AUSRIVAS does not use the probability of taxon occurrence based on just one classification group that is most similar to the test site (Simpson and Norris 2000). Rather, AUSRIVAS uses the weighted probabilities of the test-site membership to all of the groups (in a sense accounting for the assemblage continuum). In many cases the AUSRIVAS method has produced models that work well (based on how well the discriminant models predicted group membership, the degree to which O/E values differed among reference sites, and how well models predicted the taxa found at new reference sites)

(Coysh et al. 2000; Hawkins et al. 2000). A 2000 review of alternatives to the RIVPACS type of predictive model (Johnson 2000) concluded that it was a robust approach for predicting assemblage structure and found no compelling reason to justify changing to alternative techniques.

Since Johnson's review in 2000, other modelling methods have been used more extensively (Linke et al. 2005; Van Sickle et al. 2006; Chessman 2009; Webb and King 2009; Aroviita et al. 2010; Feio and Poquet 2011). Different approaches may offer alternatives where the current modelling methods have not produced good models. For example, neural networks and Bayesian techniques are able to link levels of expert-derived information and can provide diagnostic capability (Olden et al. 2006). Predictive approaches that incorporate multi-metrics based on assemblage structure and function or functional traits (Merritt et al. 2002; Menezes et al. 2010) may also provide diagnostic information (Poff et al. 2006; Pont et al. 2006). Given the major initial investment in the AUSRIVAS bioassessment approach and the utility of the method, and almost 20 years of experience since its inception, an appraisal seems timely.

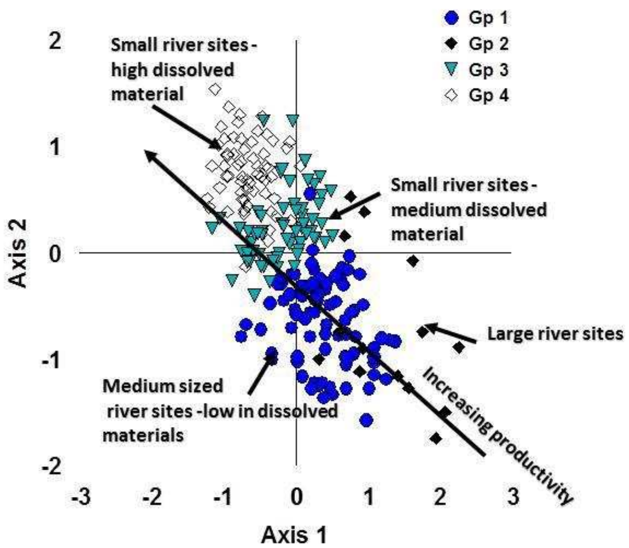

Figure 8. 233 reference sites from the Fraser River, British Columbia, showing distribution along environmental gradients and groups identified by cluster analysis (from Sylvestre and Reynoldson 2006).

The current AUSRIVAS model building approach is essentially standard among the large national bioassessment programs (e.g. AUSRIVAS; RIVPACS; CABIN: the Canadian Aquatic Biomonitoring Network, Environment-Canada 2012) but there are differences among them. The most widely used model output for assessment is the O/E score, which expresses the ratio between the expected number and the observed number of taxa, in essence based on taxon richness. The UK (RIVPACS), Australia (AUSRIVAS) and the USA (where RIVPACS type models are used) all use the O/E indicator of river condition. In Canada a different approach is taken where the ordination distance of a test site from the (most similar) reference group centroid is used as the measure of biological condition (Reynoldson et al. 2000). Two factors compelled an alternate assessment method. First, the relatively small number of invertebrate families found in Canadian streams restricted the use of an O/E score, and second the view that changes in abundance within taxa also provided valuable information on community response (Reynoldson 2012, pers. comm. 12 April). O/E indicates the number of taxa that actually occurs at a test site as a percentage of those (from a given list) predicted to occur (Marchant 2002). The problem with few expected taxa is that if a taxon is missing (not observed in the sample) from a list of 10 expected, that absence will have greater effect on the O/E value than one missing from a list of 20. Marchant (2002) found that models with an expected taxa list shorter than 20 produced increasingly variable O/E results as the list shortened, and the O/E from a model with < 10 expected taxa was unreliable. Another alternative to the O/E score is an adaptation of Bray–Curtis distance (the BC index) (Van Sickle 2008), which was found to be more sensitive in a US study than the O/E and could include low-probability taxa without reducing its power to detect non-reference conditions. Thus, other measures of biological condition may provide an alternative to the O/E score where current AUSRIVAS modelling is problematic for similar reasons.

Although users of the AUSRIVAS predictive models rely largely on the O/E score and the bands of biological condition to make an assessment, improving the diagnostic capacity provided by other AUSRIVAS software outputs could increase their usability. For example, AUSRIVAS already provides a taxonomic list and each taxon’s probability of occurrence; further development could enhance the interpretive and diagnostic capacity by developing ‘expert system’ software to employ the known sensitivities of taxa and utilize the diagnostic capacity of taxa that are be missing though expected to be present, as well as those present but unpredicted. Such a computer system could emulate the interpretive ability of a human expert. Linking a database of sensitivities and habitat preferences would be analogous to trait-based characteristics but incorporating predictive modelling (Feio and Doledec 2012). The inclusion of other invertebrate assessment indicators as AUSRIVAS software outputs — for example, percentage of Ephemeroptera-Plecoptera-Trichoptera (EPT) taxa — and other metrics (see Barbour et al. 1999) and the development of diagnostic elements (through the use of biological traits and preferences) is an area for further research and development to improve the utility of AUSRIVAS software. Such improvements could advance the scientific underpinning of biological assessment of river condition by facilitating user interpretations beyond the rudimentary O/E values.

The intent was to expand AUSRIVAS beyond invertebrates to possibly include fish, diatoms, macrophytes, riparian vegetation and functional measures (Davies 2000). Draft national protocols have been developed for diatoms (John 2004), habitat and riparian vegetation (Ladson et al. 1999; Parsons et al. 2004), fish (Davies et al. 2010) and benthic metabolism (Fellows et al. 2006). Leaf-litter breakdown and cellulose decomposition also hold potential for assessing the functional integrity of riverine environments (Boulton and Quinn 2000; Gessner and Chauvet 2002). Predictive modelling for fish and invertebrate habitat has also been trialled (Davies et al. 2000; Mugodo et al. 2006). The development of predictive models for other biota to the same sophistication and scale of the AUSRIVAS invertebrate models would require major financial investment.

A current initiative may provide a suitable platform for integrating AUSRIVAS and other indicators of river health, the Framework for the Assessment of River and Wetland Health (FARWH) (NWC 2012). The FARWH is based on the premise that ecological integrity of a river system is represented by all the major environmental components (not unlike the original premise underpinning AUSRIVAS) (Alluvium_Consulting 2011). Other approaches for the integration of multiple assessment indicators include the Index of Stream Condition (Ladson et al. 1999), river report cards (Bunn et al. 2010) and Sustainable River Audit methods (Davies et al. 2010). However, all efforts for broad-scale integration of bioassessment approaches and efforts to address the AUSRIVAS model concerns require a nationally coordinated and funded management effort (Davies 2007; Schofield 2010).

Reference site concerns

Chapter 7 (Nichols et al. 2010b) addressed concerns about reference-site stability but users of AUSRIVAS have further concerns relating to the appropriateness of the reference site data used to create the predictive models, and this is a reason why some agencies are not using AUSRIVAS models (Davies 2007). AUSRIVAS reference sites may represent the ‘best available conditions’ or ‘minimally disturbed conditions’ and thus, are not required to be in ‘pristine condition’. However, these expectations need to be explicit from the outset of a bioassessment program (Stoddard et al. 2006). For AUSRIVAS, the appropriateness of reference site condition was originally determined by establishing independent criteria (Davies 1994). Alternative techniques for defining reference sites are now available through GIS layers and remotely sensed data (Hawkins et al. 2010; Yates and Bailey 2010). Quantitative methods use human-stress gradients derived from GIS land-use layers to provide a more objective way of selecting reference sites (Yates and Bailey 2010). GIS data may identify areas of natural environment and human activity to score sites by exposure to human stress thus allowing the selection of potential reference sites (Yates and Bailey 2010). However, the reference condition can also represent stressed conditions if the stressors are natural (e.g. drought effects), which presents another issue regarding temporal variability and how or when new reference site data should be incorporated into a predictive model (Rose et al. 2008).

Ongoing sampling should consider both additional reference sites in under-represented areas and periodic resampling of sub-sets of reference sites to consider long-term changes in reference sites under such scenarios as climate-change. Research has identified cases (see Chapter 7) where longer-term trends in reference site condition suggest that sites do remain within a stable reference condition (Metzeling et al. 2002; Reynoldson 2006; Sylvestre and Reynoldson 2006; Nichols et al. 2010b) and the concordance of a reference site to a reference group in predictive models appears robust for those environments and at the spatial scales (and taxonomic level) studied. However, this needs further review and validation before acceptance as a general conclusion (Reynoldson and Wright 2000), particularly in Australia where the long-term temporal variability and spatial variability is high.

Whether or not sufficient reference sites exist within a region to employ the reference condition approach is another concern, particularly for Australia’s large and/or dry-land rivers (Chessman et al. 2010). Testing a model’s power and sensitivity may determine the adequacy of a model's performance (Bailey et al. 1998). One approach is to use simulated impacts, thus removing any circularity in making a determination regarding the sensitivity of a method (Cao and Hawkins 2005; Bailey et al. 2012). However, addressing concerns regarding the lack of sufficient reference sites may need to consider alternative modelling approaches (Chessman et al. 2010).

12.2 The role of Eco Evidence in evidence-based practice

In Chapter 10, I presented a new method to advance ecological assessment by combining scientific evidence from many studies. Eight potential uses for Eco Evidence are stated but to date the published Eco Evidence case studies have been used for only one of those purposes, "*to focus a literature review to the point where the output can be published as a succinct review paper*" (Norris et al. 2012, p 16), and emphasize the use of peer-reviewed literature. This work is at the frontier of environmental causal assessment and future work should expand the adoption of Eco Evidence for the other purposes. Further research is required to validate the assumption that the type of scientific evidence defined by Eco Evidence is the right type to satisfy the practical needs for all the stated purposes.

'Science-intensive' management or policy decisions are never based purely on science, they usually involve political judgment and practical considerations (Briggs and Knight 2011). For example, achieving a balance between environmental and economic needs requires trade-offs and an understanding of the synergies involved (Collier et al. 2011). Realizing beneficial ecological outcomes for the 'science-intensive' issues requires relevant scientific knowledge to develop appropriate policy options and to assess their effectiveness (Cullen 2006). The complex relationship between scientific knowledge and policy (Juntti et al. 2009) has made implementation of evidence-based approaches difficult.

The argument against evidence-based practice is the need for quick decisions, often made with incomplete knowledge of the situation or the consequences of the actions. Nevertheless, practitioners seek to base decisions on best available evidence (Pullin and Knight 2001). Where decisions need to be based on science, the effectiveness of those options should be demonstrated by scientific experiment or systematic review of the scientific evidence in a transparent and defensibly logical process (Pullin et al. 2004). Without such process the decisions will be made regardless, without access to the best quality scientific evidence, thus increasing the probability that inappropriate options will be adopted (Pullin and Knight 2003).

In this regard, the role of Eco Evidence is to aid understanding of the consequences of different choices (Skinner et al. in press).

Early adopters of Eco Evidence undertook the case studies outlined in Chapter 11 for the purpose of systematic literature review. It is too soon to know if environmental practitioners will adopt the method to facilitate environmental evidence-based practice and improve environmental outcomes. Traditional approaches to improve uptake of research findings in the health discipline have focused on increasing the availability of information and improving the presentation of evidence (Straus et al. 2005a). This is achieved by synthesizing and disseminating evidence in accessible formats, such as reviews in journals, clinical guidelines, better access to electronic sources of information, training, and conferences (Grol and Grimshaw 2003). However, the consistent research finding that practice still lags behind scientific research by years, in both health and environmental sciences, indicates more is required to implement most innovations (Pullin and Knight 2001; Bates et al. 2003; Grol and Grimshaw 2003; Shanley and Lopez 2009; Likens 2010).

Among the reasons commonly cited as barriers to the uptake of evidence-based practice are lack of time, lack of facilities, lack of motivation, and information overload (Newman et al. 1998; Brownson et al. 2006). Also, the differences between medical and environmental practice are cited as reasons against the uptake of similar epidemiological approaches in conservation biology (that is, the complexity of the natural world compared to the human body, comparative lack of relevant studies, variable quality of studies in conservation compared to medicine, and the inaccessibility of the literature) despite its successes in public health (Stewart et al. 2005). However, most of these arguments seem directed to the randomized trials of medicine, whereas environmental studies may have more in common with public health studies. Both often lack a comparison, require more caveats on the interpretation of results, and have few relevant studies on which to base comparisons (Brownson et al. 2011). For environmental managers to use scientific evidence in practice requires them to make management interpretations of the scientific results, something currently done by scientific, rather than management, ‘experts’.

Pullin et al. (2004) see two challenges to developing environmental evidence-based practice; 1) to ensure that the results of research influence practice; 2) to increase good quality research regarding the effectiveness of interventions. Fundamental to meeting these challenges is systematic review, where the evidence undergoes critical appraisal using a standard protocol. Eco Evidence offers a standardized protocol for the evaluation of evidence and provides access to a reusable bank of evidence to address new questions, or to repeat a previous review when new evidence becomes available. The availability of intervention reviews would be valuable but of further value would be Eco Evidence training to empower managers to undertake their own reviews, which are specific to the management questions of interest.

The Eco Evidence database contains evidence 'items' that are in a usable form ready for systematic review and causal analysis. To date, researchers undertaking systematic reviews on specific questions (see Webb et al. 2012) have manually extracted the evidence from the literature and entered it into the Eco Evidence database. However, to overcome barriers to accessibility of primary research and time constraints associated with keeping up-to-date with the new research, the Eco Evidence database needs to be populated with far more 'reusable' evidence items than it currently contains (Webb et al. 2011). Potential pathways for larger-scale population of the Eco Evidence database include:

- the incentive for researchers to enter evidence from their own peer reviewed publications. The premise being that studies in the database will have a better chance of being cited in review articles than other studies;
- potential arrangements with targeted journal publishers where submitting authors can upload their evidence to the database. The potential for increased citation rates and impact factors may provide incentive for publishers to become involved;
- artificial intelligence techniques, such as natural language processing (Demner-Fushman et al. 2009) to at least partly automate the extraction of evidence from the extensive pool of existing literature;
- Eco Evidence training for management agencies to undertake systematic intervention reviews that would not only draw on information contained within the database but add to it as well; and
- compatibility of Eco Evidence with an international standard for ecological evidence storage and/or synthesis to enable sharing of ecological evidence.

On this last point, work with the US EPA, who regularly extract and analyse evidence from the scientific literature using their CADDIS framework (Norton et al. 2008) has established a draft standard definition of an 'evidence item' and web services to allow retrieval of evidence from both the US EPA's CADDIS database (USEPA 2012b) and the eWaterCRC's Eco Evidence Database (Webb et al. 2011). The Waterbodies in Europe project (WISER 2012), has also made early steps in developing evidence bases. This collaborative work aims to increase the scope of information available to environmental practitioners and to boost capacity in understanding and using ecological evidence. Continued research and development has the scope to build an ecological ontology to further enhance the searching, sharing and understanding of evidence and provide new analytical power to investigate large-scale patterns and ecological responses to environmental stressors (Chandrasekaran et al. 1999). Such research, evidence databases, and causal analysis methods, have the potential to revolutionize evidence-based practice in environmental management and policy.

Conclusions

The body of this thesis includes nine of my published research articles. These nine papers have contributed to a body of research on ecological assessment of river condition, in Australia, and overseas. The thesis traces the development of ecological assessment and shows where my work has made a significant contribution to knowledge about assessment of river condition. From field-based studies of environmental change to desktop studies of multiple lines and levels of evidence of cause–effect, I have provided the background and critical review to provide the research context and have identified areas for further research and development. I demonstrate the value of bioassessment by describing applications and evaluation of the Australian River Assessment System, which has been the national standard method of assessing river health for over a decade. AUSRIVAS: includes a standardized invertebrate sampling method, the 'reference condition approach', predictive models, and software for assessing river health. However, new methods to aid the synthesis of ecological studies are imperative if the ever-increasing scientific research is to transfer to practice to improve management and outcomes for freshwater systems. My most recent work has contributed to establishing a new causal criteria analysis method, 'Eco Evidence', for assessing evidence for and against environmental cause–effect hypotheses.

A national biological assessment program needs to produce results that are both rapid and soundly based on scientific principles. Evaluation of the advantages and limitations of the AUSRIVAS method was a necessary component in developing the national assessment system. Testing methods and establishing general rules for sample replication at various spatial scales validated the method for both site-scale and broad-scale assessments (Chapter 3).

The variability of AUSRIVAS assessments attributed to the size of the area sampled was negligible, indicating that area sampled was adequate for AUSRIVAS bioassessment in the upper Murrumbidgee region (Chapter 3). For more accurate site-scale AUSRIVAS assessments, investigators should consider replicated collections and sub-samples because a greater proportion of variability was attributed to sub-sampling of the collections (Table 3). Generally, sample replication should be maximized at the spatial scale required for reporting (e.g. at the river-reach, catchment or larger regional scale).

Table 3. Suggested sample replication for AUSRIVAS as applied to different scaled studies.

	Scale of study	
Replication	Site (river reach-scale 100–250 m)	Regional (catchment or multi-catchment scale)
Replicate sub-samples per sample	3	0
Replicate samples per site	2	1
Replicate sites per study	Replication should be maximized at the spatial scale required for reporting	Many single-site samples distributed to encompass as much spatial variability as possible

Given the differences in invertebrate composition of AUSRIVAS live-picked and lab-sorted samples (Chapter 4), the finding that data derived through different sub-sampling methods resulted in few method-related differences in site assessments (when run through the 'appropriate' predictive model) established the importance of using the O/E values output by models for across-border assessments. Although this study was not replicated in multiple jurisdictions, it would seem wise to exercise caution in broad-scale assessments that cross jurisdictions where different sub-sampling methods are used (such as for SoE reporting).

AUSRIVAS methods were tested within a variety of research settings from targeted assessment to broad-scale national assessment. AUSRIVAS predicted pre-dam biota, which allowed interpretation of the results within a multidisciplinary framework (Chapter 5) and demonstrated the method's utility to provide an assessment of river condition as a specific response to flow regulation. In Australia and elsewhere, spatial and temporal scales are important when considering environmental gradients and taxonomic distributions, for both using and building predictive models for bioassessment. AUSRIVAS biological assessment methods were adapted to develop predictive models to assess the condition of Portuguese streams (Chapter 6). This work was important in determining that regional, rather than the broader national-scale, was generally the most appropriate spatial scale at which to develop invertebrate predictive models for assessment of water quality in the Portuguese territory.

An AUSRIVAS-type model and a study design that included standardized sampling of fixed sites (both test and reference) over long periods was an important factor in distinguishing the ecological effects of human activities from those attributed to climate related influences in Kosciuszko National Park (Chapter 7). The National Park case-study demonstrated that the reference-condition approach for ecological assessment maintained the integrity of the bioassessment program through time by providing a stable benchmark to compare current test sites. Regardless of whether a change is the result of human activities or natural phenomena, without long-term data it is difficult to assess where the system is positioned along a recovery trajectory. This will be especially important in the result of extreme events like extended drought, extensive bushfires and major floods. With climate-change predictions of increasing frequency of extreme events, if river health is valued it should be part of a long-term assessment program.

Furthermore, effective long-term bioassessment and adaptive management demonstrated that experimental flow management of the Cotter River during severe drought maintained the Cotter River's resilience to recover when higher flows returned (Chapters 8 and 9). Ongoing biological assessment, adaptive management and a flexible policy instrument achieved beneficial ecological outcomes through ACT's environmental flows program. A major element in the success of ACT's adaptive flow management approach was the use of AUSRIVAS integrated with other indicators in study design that could cope with changing questions and unforeseen events, such as extended drought and extensive bushfire (Chapter 8).

The sustained use of the AUSRIVAS method continues to provide data for targeted impact assessment, state/regional, and very broad-scale assessments of river condition, as well as community based river assessment programs. Major government agencies strongly support the use of a nationally standardized sampling protocol because it provides data sets that are comparable spatially and temporally. One positive outcome of the large assessment programs is their pivotal role in identifying and raising awareness of the problem of degraded river conditions (a necessary first step). However, resource managers now need to think to the next phase – conducting rigorous, large-scale experiments within an adaptive management framework. However, AUSRIVAS is not currently coordinated nationally or funded to the level required to support expansion in breadth or to continue research and development, which is undermining its utility.

Bioassessment needs in Australia have evolved and environmental managers are asking new questions and have new reporting needs. For example, to increase diagnostic capabilities; to inform questions about aquatic ecosystem sustainability in relation to the flow regime types (perennial, intermittent and ephemeral); and to evaluate effectiveness of environmental water use and water allocation. AUSRIVAS needs to evolve with new methods to stay relevant to management needs. A sound strategy would be to protect the national investment in AUSRIVAS and address the concerns and deficiencies rather than adopt entirely new and largely untested approaches to bioassessment of river health.

The research outputs presented in this thesis have contributed to an ever-increasing number of freshwater publications, which will become a burden for environmental managers, rather than an asset, without knowledge-management systems designed to take advantage of new knowledge. The new causal analysis method presented in Chapter 10 and 11 draws from multiple studies to assess the evidence for and against environmental cause–effect hypotheses. Evidence-based practice relies on implementing interventions that have worked previously in studies of similar situations. Assembly of scientific evidence (from published literature, unpublished studies and grey literature) will only inform practice and decision-making if assembled to form compelling arguments to reduce uncertainty and provide the confidence to make decisions. With scare resources for environmental management, it makes sense to draw on existing evidence from published scientific literature to inform decisions. Eco Evidence, for quantifying and combining evidence from multiple scientific studies, along with the major advance of a weighting system for individual studies, is an important first step in facilitating broader use of systematic assessment of cause–effect relationships in environmental sciences.

The Eco Evidence method, database and software tools will address some of the perceived barriers to evidence-based practice and make better use of the extensive published scientific research currently underutilized for ecological causal assessment. Such assessment can be necessary for informing management actions aiming to improve environmental condition. The Eco Evidence research effort is part of a worldwide trend towards the greater use of evidence-based methods in environmental management, and the tools described here are contributing to change the way scientific evidence is used to solve environmental problems.

My research has contributed to improving the understanding of ecological assessment that uses invertebrate predictive models, the reference condition approach and causal criteria analysis. Rigorous bioassessment studies and the reference condition approach when applied within the contexts of adaptive management, long-term assessment, and a framework for causal assessment can provide the ecological evidence to inform current and future river management.

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Appendix 1: Statements verifying contribution to coauthored papers

In accordance with The Gold book Section 28, I provide the following statements from coauthors of joint-authored published research outputs verifying my contribution.

Please note that coauthors signed two of these statements before the final reviewers' comments. In response to reviewers' comments, the titles of the following papers have since changed slightly:

"Environmental flows and drought: what happens when the tap is turned off?" was changed to "More for less: a study of environmental flows during drought in two Australian rivers".

"The case for causal criteria analysis in environmental assessment: making best use of the scientific literature" was changed to "Analyzing cause and effect in environmental assessments: using weighted evidence from the literature".

Appendix 1

This appendix has been removed due to privacy restrictions.

To see this appendix, please contact the author.