Chapter 1: General introduction

1.1. Background

1.1.1. Models of ecosystem structure and function

Growing concern about the globally pervasive impacts of human modifications to riverine landscapes (Benke 1990, Allan & Flecker 1993, Vitousek *et al.* 1997, Jackson *et al.* 2001, Malmqvist & Rundle 2002), has led to increasing recognition of the need for quantitative procedures for assessing aquatic ecosystem 'health' and monitoring biotic responses to remedial management. Natural functioning aquatic ecosystems have important intrinsic values and also provide many goods, services and long-term benefits to human society (Costanza *et al*. 1997, Baron *et al*. 2002), hence their protection, remediation and restoration is of critical importance.

Rivers and streams are influenced by the landscapes through which they flow (Hynes 1975, Vannote *et al*. 1980), a fundamental link recognised in many of the conceptual models describing the structure and functioning of natural river systems developed over the past 25 years or more. Early concepts such the River Continuum Concept (RCC, Vannote *et al.* 1980) and its corollaries, for example nutrient spiralling (Elwood *et al.* 1983) and the role of stream hydraulics (Statzner & Higler 1986), and subsequent developments including the Flood-Pulse Concept (Junk *et al.* 1989) and the Riverine Productivity Model (Thorp & Delong 1994), tend to emphasise linear (i.e. longitudinal and lateral) relationships between physical and biological processes in rivers. However, others have suggested that lotic systems should be viewed as being organised within a nested hierarchy, functioning at a variety of spatial and temporal scales (e.g. Frissell *et al.* 1986, Amorous *et al.* 1987, Pringle *et al.* 1988, Johnson *et al*. 1995, Fausch *et al.* 2002). The integration of hierarchical models with concepts of patch dynamics (White & Pickett 1985, Pringle *et al.* 1988, Townsend 1989) and the role of disturbance (Resh *et al.* 1988) has led to further insights into the relative importance of physical and biological processes and the role of natural spatial and temporal variation in driving aquatic ecosystem structure and function (Fig. 1.1).

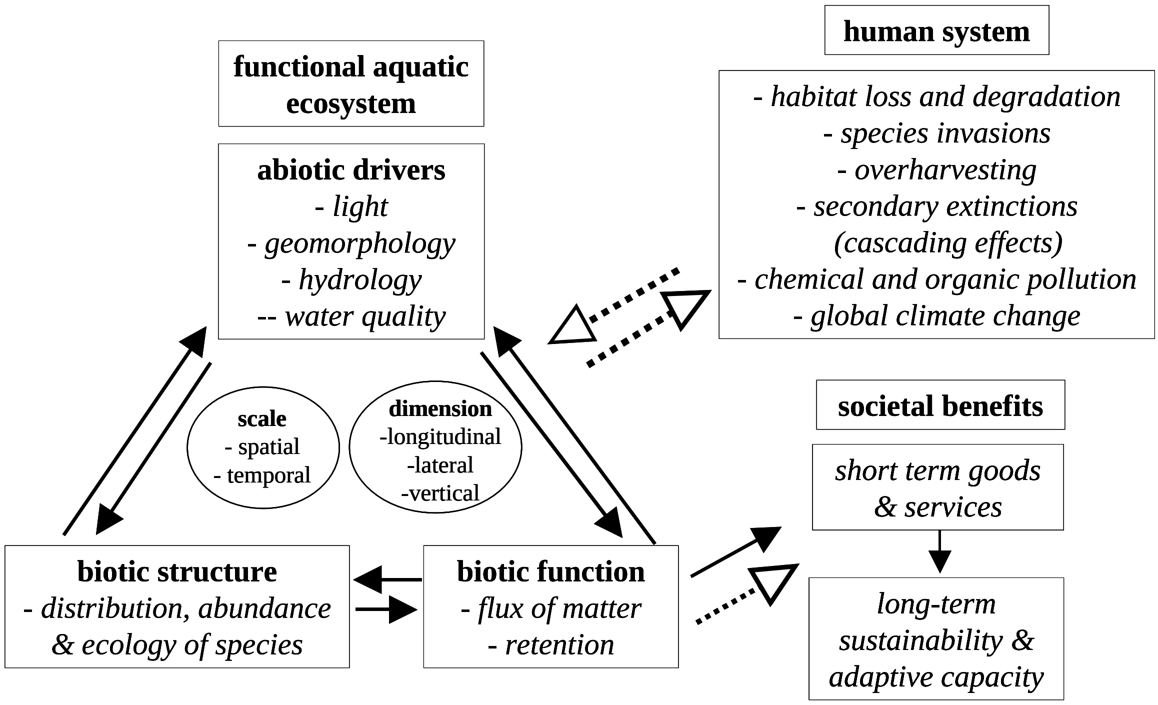

Figure 1.1. Simplified conceptual model of linkages between abiotic drivers and biotic responses in functional aquatic ecosystems that deliver goods, services and long-term benefits for human society (linkages shown with dark solid arrows). Human disturbances impact natural aquatic ecosystems and hence affect the societal benefits they provide (linkages shown with dashed open arrows). The nature and strength of these relationships is contingent on the spatial and temporal scale of examination. Figure modified from Lorenz *et al.* (1997) and Baron *et al.* (2002), using information sourced from Allan & Flecker (1993).

Of overriding importance are large scale physical factors such as geomorphology and climate which define higher levels of organisation of the physical (e.g. catchment boundaries, river morphologies, flow regimes and water quality) or biological (e.g. species pools and floral and faunal distributions) aspects of the river-floodplain ecosystem (i.e. the concept of landscape filters, Smith & Powell 1971, Poff 1997) (Fig. 1.1). These higher levels of organisation, in turn, contain lower levels of organisation (e.g. microhabitat attributes and biotic communities) and ecosystem function (e.g. flux and retention of organic matter) that may be determined by smaller scale processes (e.g. flow characteristics and species interactions) (Johnson *et al.* 1995). Undoubtedly, combinations of regional and local abiotic and biotic factors are responsible for structuring local ecosystem structure and function (e.g. Angermeier and Winston 1998) and impressions of their relative importance may be dependent upon the scale at which these mechanisms are examined (Jackson *et al.* 2001).

These concepts are potentially useful for understanding mechanistic pathways for the impacts of human disturbances on river ecosystems (Lorenz *et al*. 1997). Unfortunately however, many of the linkages in natural river ecosystems portrayed in Figure 1.1 are poorly understood, poorly validated, and remain largely descriptive hypotheses rather than predictive or quantitative models. Furthermore, except in broad terms, they do not provide predictions of the effects of human disturbances, or candidate physical, chemical and biological indicators that environmental managers can use to assess ecological health, or mechanisms by which management or rehabilitation of rivers may be effectively undertaken (Walker 1993, Johnson *et al.* 1995, Ward *et al*. 2001).

1.1.2. Human impacts on aquatic ecosystems

A range of human disturbances can function individually or interact to directly and/or indirectly affect functional aquatic ecosystems and hence the goods, services and long-term benefits they provide for human society (Allan (2004), Baron *et al.* (2004) and Figure 1.1). Soule (1991) described a “sinister sextet” of the major sources of global species loss in general, which Allan & Flecker (1993) adapted to identify six major threats to biodiversity in flowing river systems. These include:

1. Habitat loss and degradation caused by water infrastructure projects, land transformations and agriculture that cause modifications to hydrology, connectivity, riparian-aquatic linkages and in-stream habitat integrity;
2. Species invasions;
3. Overharvesting;
4. Secondary extinctions due to cascading effects;
5. Chemical and organic pollution; and
6. Global climate change.

Allan (2004) emphasised the importance of land use impacts associated with agriculture, extractive industries and urbanisation. He highlighted six primary sources of impact on aquatic structure and function: sedimentation; nutrient enrichment; contaminant pollution; hydrologic alteration; riparian clearing/canopy opening; and loss of large woody debris (Allan 2004). Evaluating the importance and mechanisms of impact on aquatic ecosystems is hampered by the current limitations in our understanding of relationships between natural environmental drivers and ecosystem responses. Progress requires studies in regions with a representative suite of undisturbed rivers in which such relationships can be quantitatively examined and can serve as benchmarks for assessing human impacts (Ward *et al.* 2001, Chessman & Royal 2004). Unfortunately, such areas are becoming increasingly scarce. Another problem with detecting and diagnosing sources of impacts of human disturbance on aquatic ecosystems or defining the magnitude of the problem, has been historical imprecision and/or disagreement in what exactly is the demonstrable evidence of human impacts and what are the targets for a healthy ecosystem against which impacts are to be assessed.

1.1.3. Assessment of aquatic ecosystem 'health'

The status of aquatic ecosystems and their response to human impacts are commonly described using terms such as condition, biotic or ecological integrity, or health (Karr 1999, Norris & Thoms 1999, Allan 2004). Implicit in these concepts is the reference to some pre-defined state. The state may include the natural or inferred-natural condition prior to human impact, or a condition determined by community expectations, values and uses of the aquatic ecosystem (Norris & Thoms 1999). Perhaps as a consequence of this, there has been considerable debate among scientists and managers as to the meaning of terms such as condition, integrity and health, and their value in conveying to the community important principles about the impacts of human disturbances on aquatic ecosystems (Calow 1992, Suter 1993, Wicklum & Davies 1995, Westra 1996, Boulton 1999, Karr 1999, Norris & Thoms 1999). Frey (1977, p.128, cited in Norris & Hawkins 2000) defined biotic integrity as "the capability of supporting and maintaining a balanced, integrated, adaptive community or organisms having a composition and diversity comparable to the natural habitats of the region". Definitions of ecosystem health are based on parallels with human health and emphasise principles of ecosystem organisation, resilience and vigour, as well as the absence of signs of ecosystem stress (Rapport 1995, Rapport *et al.* 1998). The incorporation of a human dimension in which humans value rivers and the goods and services they provide for a range of needs and uses, and where unhealthy rivers satisfy only a subset of these, is critical (Karr 1996, Meyer 1997). Although many have argued that an analogy between human health and ecosystem health oversimplifies a complex issue (see Boulton 1999), incorporating principles of ecological integrity (maintaining ecosystem structure and function) and human values (what society values in the ecosystem) into the definition of river health may provide impetus for advances in aquatic ecology, more effective and sustainable management of aquatic ecosystems, and broader acceptance of management goals and activities by the community (Karr 1996, 1999, Meyer 1997, Boulton 1999).

1.1.4. Indicators of ecosystem health

A range of methodologies based on the use of indicators of the physical, chemical, biological (structural) and functional (process) characteristics of ecosystems has been developed for assessment, diagnosis and prognosis of ecosystem health (Norris & Thoms 1999, Gergel *et al*. 2002, Niemi & McDonald 2004). The choice of physical, chemical or biological indicator depends largely on the reasons for undertaking the work or the type of anthropogenic impact to be assessed. A wide range of aquatic organisms has been used including algae, macrophytes, macroinvertebrates and fish (Norris and Norris 1995). More recently, indicators of ecosystem processes (e.g. benthic metabolism) have been used (e.g. Bunn 1995, Bunn *et al.* 1999, Bunn & Davies 2000). Indicators are useful tools because, ideally, they have an observable measurable quantity with significance beyond what is actually being measured (Lorenz *et al*. 1997). However, indicators are, by definition, suggestive of some unmeasurable condition and have been criticised on this basis (e.g. Suter 2001). Desirable qualities of river health indicators include accuracy, sensitivity, precision, rapidity, robustness, proven worth, cost effectiveness, simplicity and/or clarity of outputs. However, many of these features may be in mutual conflict (e.g. the robustness of an indicator *versus* its sensitivity) thus there must be some direct trade-off between these desirable characteristics (Fairweather 1999). Ultimately, indicators should be widely applicable, simple to interpret and easy to communicate (Fairweather 1999).

Cairns (1995) suggested that suitable indicators of aquatic ecosystem condition should: be based on ecological knowledge and conceptual models of ecosystems; incorporate elements of biological structure, composition and function; be useful in waters other than those in which they have been developed; be diagnostic, heuristic or both; and have sufficiently small sampling and annual variability to be responsive to marked differences or changes in habitat quality or disturbance levels.

Fish have been advocated as useful indicators of biotic integrity or river health (e.g. Fausch *et al*. 1990, Harris 1995, Paller *et al*. 1996, Simon 1999, Karr & Chu 1999) because:

1. they are almost ubiquitous components of aquatic ecosystems;
2. they are relatively long-lived and mobile and therefore reflect conditions over broad spatial and temporal scales;
3. local assemblages generally include a range of species representing a variety of trophic levels and therefore integrate effects from lower trophic levels;
4. fish are at the top of the aquatic food web and are consumed by humans, making them important for assessing contamination;
5. environmental and life history requirements are comparatively well understood; and
6. they are relatively easy to collect, identify and subsequently release unharmed.

However, freshwater fish present some potential problems as indicators (Berkman *et al.* 1986) because:

1. quantitative samples are difficult to obtain;
2. species distributions and abundances may vary between regions or drainages due to factors other than disturbance;
3. site by site differences may be difficult to interpret due to spatial and temporal variation in species composition and abundance;
4. fish are mobile and thus may avoid areas of stress; and
5. hypotheses concerning likely responses of indicators of fish assemblage structure and function to specific disturbance types are not well developed or explicitly stated.

The relative mobility of many fishes also highlights the potential for impacts occurring outside the scale of investigation to bias assessments at smaller spatial scales. The longevity of some species of fishes can also lead to circumstances where their absence may reflect impacts occurring several years to decades previously (Schlosser 1990). Also, because fish integrate effects from lower trophic levels, by the time effects are visible in fish communities, the ecological health of lower trophic levels may be irreparably damaged. In addition, because of their mobility, the presence of species indicates suitable conditions (*viz.* water quality and habitat), whereas the absence of a particular species does not necessarily reflect the converse. Nevertheless, freshwater fishes are used widely as indicator organisms (Harris 1995, Simon 1999) and are the focus of this thesis.

1.1.5. Approaches to ecosystem health assessment using biota

A critical underpinning of ecosystem health indicators is the ability to accurately define the expected condition for those attributes upon which the indicators are based. This requires that natural spatial and temporal variation in the attributes, driven by variation in environmental conditions, can be accounted for to the extent that impacts of human-induced disturbance can be accurately assessed (Resh & Rosenberg 1989, Grossman *et al.* 1990). Most approaches to assessing ecosystem health using ecological indicators specifically incorporate the concept of reference to the natural state as a mechanism to assess whether a location is impacted or not (Norris 1995, Reynoldson *et al.* 1997). The attributes of the reference condition are usually derived from surveys of "undisturbed" or "least-disturbed" systems. Such surveys need to be extensive so as to incorporate spatial and temporal variation in the physical and biological characteristics of aquatic systems. The actual description of the characteristics of natural systems may also be problematic in landscapes that have already been substantially altered by anthropogenic activities and for which little historical information exists.

There are two principal methods for river health assessment using the reference condition concept: multivariate predictive models of biotic community composition (e.g. Wright 1995, Clarke *et al.* 1996, Simpson & Norris 2000, Oberdorff *et al.* 2001) and of summary attributes of community structure and function (e.g. Index of Biotic Integrity – IBI, Karr 1981, Karr *et al.* 1986). Multivariate predictive models of biotic structure are widely used tools for assessment of aquatic ecosystem health and models have been successfully developed for the prediction and assessment of aquatic macroinvertebrates, diatoms, local stream habitat features and fish. Predictive models are developed that enable site-specific predictions of biotic community composition expected in the absence of major human disturbance. The expected fauna is derived using a small number of environmental characteristics as predictors of species composition. An evaluation of the biological integrity of the site is obtained by comparing the expected fauna at a new site, with that observed. This method, based on a predictive modelling procedure originally developed for assessing the biological quality of rivers in the United Kingdom using aquatic macroinvertebrates - the RIVPACS method (Wright *et al.* 1984), has been packaged as AUSRIVAS (the Australian River Assessment Scheme) and is now implemented widely throughout Australia under the National River Health Program (Simpson & Norris 2000). The development of multivariate predictive models of fish assemblage composition and their utility in stream bioassessment programs in Australia has received little attention. However, fish-based predictive modelling methods have been demonstrated to provide a sensitive tool for biomonitoring river health in Europe (Oberdorff *et al*. 2001) and New Zealand (Joy & Death 2000, 2002, 2003).

The other common approach to bioassessment based on the reference condition concept has been to relate changes in summary attributes or ‘metrics’ that describe aspects of biotic assemblage structure and function to environmental stress. Summary metrics have been advocated as an effective means of encapsulating the complexity of natural communities sufficiently to assess the types and strengths of human impacts and to communicate results of studies to others (e.g. environmental managers) (Karr *et al*. 1986, Fausch *et al*. 1990, Karr & Chu 1999, Barbour *et al*. 1995, Simon 1999). Individual summary metrics based on species richness and composition, trophic composition and individual abundance and condition are usually combined into a ‘multimetric’ index (Barbour *et al*. 1995) for the assessment of aquatic systems. The multimetric approach was first developed for fish (the Index of Biotic Integrity - IBI, Karr 1981) has subsequently been adapted for macroinvertebrates and applied in a range of aquatic ecosystems throughout the world. Like multivariate predictive models, multimetric methods are referential in their approach, however the methods used for defining the expected conditions in the absence of human disturbance differs markedly in that they do not generally employ multivariate statistical models for this purpose. Indeed, it is this conceptual simplicity in defining the reference condition that is commonly regarded as one of the method’s strengths (Karr & Chu 1999). Harris (1995) suggested that multimetric methods such as the IBI are potentially applicable to stream health assessment in Australia and, to this end, the IBI has been tested and applied in several rivers of southern Australia (Harris & Silveira 1999, Murray Darling Basin Commission 2004).

There has been extensive debate on the respective merits of the predictive modelling and multimetric approaches (e.g. Suter 1993, Norris 1995, Reynoldson *et al*. 1997, Karr 1999, Karr & Chu 1999, Norris & Thoms, 1999, Karr & Chu 2000, Norris & Hawkins 2000). Much of this difference of opinion appears to have arisen from fundamental differences in philosophical approaches to river health assessment and perhaps also due to misconceptions and misrepresentations of what the two approaches aim to achieve and how they differ (see Norris & Hawkins 2000).

The central goal of bioassessment is to decide whether a site exposed to anthropogenic stress is impaired while minimising Type I errors (incorrectly classifying a site as impaired) and Type II errors (incorrectly classifying a site as unimpaired) (Bailey *et al*. 1998, Linke *et al*. 1999). Irrespective of the approach used, both multivariate and multimetric methods have several key requirements that should be satisfied before they can be applied validly and quantitatively for river health assessment in a given river or region, while simultaneously minimising Type I and Type II errors. These requirements include (but are not limited to):

1. the ability to collect raw biological data in a standardised fashion and with sufficient accuracy and precision such that it truly represents the locality in question and is directly comparable with other locations;
2. assessment of the natural ranges in spatial and temporal variation of the biological attributes in question and the drivers of this variation;
3. the ability to accurately define the reference condition for biological attributes expected in the absence of anthropogenic stress based on relationships between natural environmental drivers and biotic patterns, such that human disturbance-induced changes can be quantified using biological indicators;
4. the sensitivity and demonstrated ability of the chosen indictors to reflect/respond to human disturbance (irrespective of the methods used to define their expected state in the absence of human stress); and
5. the relative importance of potentially confounding environmental and biological factors in interpreting spatial and temporal variation in biological attributes, such that the accuracy and sensitivity of the indicators to human disturbances can be assessed.

Satisfying these requirements can provide a quantitative basis for the use of fish as indicators of river health that is not only rigorous and scientifically defensible, but more importantly, is crucial to justify management interventions and acceptance by the community.

1.2. Aims and structure of thesis

This thesis aims to evaluate the assumptions and requirements listed above to assess whether referential approaches to bioassessment using fish can be incorporated into an ecosystem health monitoring program for wadeable rivers and streams in coastal catchments of south-eastern Queensland, Australia. There are substantial challenges to developing such a program in this region including: the relatively high environmental variability in river flow regimes; substantial and diverse existing human disturbances; low diversity and (arguably) an ecologically generalist fish fauna in comparison to elsewhere (Harris 1995); a lack of understanding of the expected responses of individual fish species to the common human impacts; and a lack of established or rigorously validated protocols for using fish as indicators of river health in south-eastern Queensland, or Australia in general. I will argue that these challenges are not insurmountable and that once the assumptions and requirements of bioassessment programs are satisfied, bioassessment methods based on fish are applicable to south-eastern Queensland and elsewhere.

The thesis is structured as follows: Chapter 2 describes the location of the study region in south-eastern Queensland and presents a brief summary of the climate, hydrology, land use and human impacts, and data sets used in the thesis. The methodology used for sampling of fish and habitat characteristics is described in Chapter 3. In this chapter, I use a set of least-disturbed sites to compare the accuracy, precision and efficiency of two fish sampling methods (single pass electrofishing or multiple pass electrofishing plus seine netting) to estimate stream fish assemblage attributes at two spatial scales (within discrete mesohabitat units and within stream reaches consisting of multiple mesohabitat units). I examine the extent to which the efficiency of each sampling method may be influenced by interspecific variation in fish behaviour and habitat use, and spatial variation in environmental conditions. My ultimate goal is to evaluate how changes in sampling effort (within and among mesohabitat units) influence the accuracy, precision and efficiency of fish assemblage estimates, depending on the fish sampling method employed.

Chapter 4 examines the influence of natural hydrologic disturbances (due to extreme high and low flow events) on the stability, persistence and resilience of stream fishes at least-disturbed sites in the study area. Conclusions about the magnitude and drivers of temporal variation in biotic assemblages are dependent on many factors including the manner in which assemblage variation is described, the role of rare species and sampling error, and the potentially confounding relationships of environmental variability gradients with other natural environmental and biological gradients. In this chapter, and also in Chapter 5, I specifically address the implications of natural environmental and biological variability for river health assessment, particularly with respect to the ability to accurately and precisely define the reference condition, such that human disturbance signals can be detected.

In Chapter 5, I construct a multivariate predictive model of native fish assemblage composition based on relationships with a small number of catchment scale and local scale environmental features. I address the question of whether accurate and precise multivariate predictive models can be constructed in a region with highly variable and unpredictable flow regimes. The model is constructed using a set of least-disturbed reference sites sampled on one occasion during one season. I evaluate the effect of low species richness on model performance and validate the predictive capacity of the model using two further sets of temporally sampled data from reference sites in two rivers. Here, I address an assumption common to many bioassessment programs of whether the reference communities from which predictions are derived are stable through time, and therefore whether valid comparisons can be made with test sites often sampled years afterwards (Barmuta *et al*. 2003). In this chapter, I also describe the application of the model to evaluate the sensitivity of fish assemblage composition and individual species as indicators of human disturbance at a set of independent test sites sampled along known gradients of human disturbance brought about by land use pressures. I use a set of independent measures of anthropogenic disturbance to describe this gradient of human impact and describe the methodology for doing so.

In Chapter 6, I critically evaluate some of the assumptions underlying the approach for defining the reference condition for summary biotic metrics commonly used in the

Index of Biotic Integrity (Karr & Chu 1999). I examine whether stratification by a single environmental descriptor (e.g. catchment area) and the use of a scoring system based on deviations from a maximum expected condition, is appropriate for defining the reference condition for a fish assemblage metric (native species richness). Native species richness is a commonly used measure of the general ecological condition of aquatic ecosystems and several species richness metrics are important components of the IBI, generally (but not always) being expected to decline with increasing environmental stress (Harris 1995, Oberdorff *et al*. 2001, 2002). I compare the predictive accuracy of the IBI approach with three regression-based methods that use a range of local and landscape variables as predictors of species richness. I develop and validate predictive models based on a set of least-disturbed reference sites and use the models to predict species richness at a set of test sites impacted to varying degrees by human disturbance. I also compare the frequency of classification errors from each method against set biocriteria and evaluate the ability of metrics derived from each method to accurately reflect the human disturbance gradient at the test sites.

Chapter 7 describes an evaluation of another commonly used indictor of river health, namely the presence and abundance of alien fish (i.e. those species introduced from other countries). The ability of many introduced fish species to thrive in degraded aquatic habitats, and their potential to impact on aquatic ecosystem structure and function, suggest that they may represent both a symptom and a cause of declines in river health and the integrity of native aquatic communities. The varying sensitivities of many commonly introduced fish species to degraded stream conditions, the mechanism and reason for their introduction and the differential susceptibility of local stream habitats to invasion due to the environmental and biological characteristics of the receiving water body, are all confounding factors that may obscure the interpretation of patterns of introduced fish species distribution and abundance and therefore their reliability as indicators of river health. In this chapter, I examine relationships of alien fish species distributions and indices of abundance and biomass with the natural environmental features, the biotic characteristics of the local native fish assemblages and indicators of anthropogenic disturbance at a large number of sites subject to varying sources and intensities of human impact.

Chapter 8 summarises the major findings of the thesis, highlights the implications of this research for the development of a river health monitoring program using fish in south-eastern Queensland, Australia, and identifies areas of future research that would further strengthen such a monitoring program.