Biological Indicators of Ecosystem Health in Wet Tropics Streams

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Final Report Task 3
Catchment to Reef *Research Program*
Cooperative Research Centre for Rainforest Ecology & Management and
Cooperative Research Centre for the Great Barrier Reef World Heritage Area

A contribution to the MTSRF Water Quality Project 3.7.3

CRC Reef and the Rainforest CRC were awarded a joint Supplementary Grant from the CRC Program for three years to develop new protocols and tools to identify, mitigate water quality problems and to assess the health of aquatic ecosystems in the wet tropics and Great Barrier Reef World Heritage Areas. The Catchment to Reef Steering Committee was established to manage the Catchment to Reef project and comprised the CEOs of both CRCs and management level representatives from stakeholder organisations including Griffith University, JCU, AIMS, WTMA, GBRMPA and the NRM Board.

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Dedication

We dedicate this Final Report from the Stream Ecosystem Health component of the *Catchment to Reef* Research Program to our colleague and friend

Garry Werren

a fine ecologist with vast experience in field studies of terrestrial, riparian and aquatic plant systems, vertebrate biology and the environmental flow requirements of rivers, streams and wetlands.

Garry played an important role in the early phases of the program, he entertained and informed us on numerous occasions in field and office, and he never ceased to be energetic and dedicated to the protection of natural ecosystems and their flora and fauna.

We salute you Garry and miss your high spirits, humorous banter and non-stop sharing of your encyclopaedic knowledge of natural history.

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Preface

This report is one of three major technical outputs of *Catchment to Reef*, a joint research program of the Reef and Rainforest Cooperative Research Centres, subsequently supported by the Marine and Tropical Science Research Facility. The research was designed to address issues raised in the Great Barrier Reef Water Quality Protection Plan, which aimed to 'halt and reverse the decline' of water quality entering Reef waters. The Plan explicitly includes catchments as well as Reef waters. The Catchment to Reef program aimed to develop new tools and protocols for monitoring of water quality and catchment health.

This report presents results of projects that have developed our scientific understanding of catchment integrity and water quality. It tests a number of monitoring methods and compares their performance. Guidelines and protocols will be further developed in the MTSRF program.

The research reported here builds on previous work done within the Rainforest CRC in which the biodiversity of streams in the wet tropics bioregion was assessed. However, the work reported here is based on a three-year program which commenced nominally in July 2003, but in fact could not commence until funding was provided in early 2004.

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Executive Summary

Introduction

This component of the *Catchment to Reef* program was designed to elucidate appropriate indicators of the ecological health of streams and rivers in the Queensland wet tropics. It built on previous research in the region and benefited from the cooperation of landholders and community members.

The research used a framework based on previous models of stream health monitoring and aimed to test the efficacy of those approaches in the wet tropics. The research focused on physical characteristics of the stream, water quality, riparian integrity, aquatic plants, macroinvertebrates and fish. Possible components not included (for pragmatic reasons) were microbial and micro-algal community composition and stream metabolism.

The research was based on a case study which allowed comparisons of approaches and methods across several response variables, and allowed determination of probable cause-effect relationships that pointed to appropriate indicators. We undertook detailed surveys of four streams in the Mulgrave and Russell catchments that differed in their land management: best management practice in the Mulgrave, but poorer practice in the Russell, in which cleared riparian vegetation and the invasion of weeds were particularly obvious problems.

Physical and chemical characteristics of the streams

The case study focussed on lowland sections of streams. All had forested uplands, good perennial flows and extensive agriculture, particularly sugar-cane growing, on the floodplain. Stream sediments had strong gradients in particle size, being rocky upstream and sandy downstream. The physical condition of the streams (bank structure, riparian integrity, etc.) decreased with distance downstream but with Woopen and Babinda Creeks in worse condition than the Little Mulgrave River and Behana Creek. Bank conditions were poor in Babinda Creek, as a consequence of poor riparian structure. Invasion by weeds (para grass and Singapore daisy) caused channelisation of flow, increased velocities and stream incision, contrasting with a shallow, meandering shady stream at Behana Creek.

Water quality in all study streams was generally good during base flows, although nutrient concentrations, particularly nitrate from fertilisers, increased substantially with distance downstream, and Babinda Creek had consistently higher nitrate concentrations than Behana Creek. Concentrations were higher at sites with greater areas of agriculture in the catchment. Lower concentrations in Behana than Babinda Creek were associated with the greater riparian vegetation cover along the stream length and best practice land management in Behana Creek. However, it was clear that a substantial change in nutrient management would be needed even in the Behana catchment if the instream water quality measures were to come close to the Queensland Water Quality Guidelines.

Assessing differences in the proportion and distribution of different land uses was a powerful means of examining these effects on a large scale and will benefit future health assessment of streams.

Macrophytes

This study investigated the composition of macrophyte (large plant) assemblages to determine whether they could be used as reliable indicators of catchment land use, riparian condition and water quality. Macrophyte assemblages of most lowland sites were dominated by emergent species, but only ferns appear to have any utility as bioindicators. They typically were absent from disturbed sites with greater proportion of anthropogenic land uses in the upstream catchment area and lower riparian condition and cover. Introduced para grass and Singapore daisy were particularly abundant where there was poor riparian canopy cover. Macrophyte cover is potentially the most useful indicator of condition in edge habitats, but the utility of macrophytes as indicators of catchment land use and water quality is limited.

Macroinvertebrates

There was a strong upstream-downstream gradient in composition and richness of invertebrate assemblages, associated with substratum particle size. The strength of the gradient provided predictability that facilitated comparisons between streams. Thus, the relationship between taxon richness and mean sediment size was similar in Behana and Babinda Creeks, but the number of taxa was significantly lower in Babinda Creek sites by about 20%. These results highlight how an understanding of the natural gradients is vital to prevent inappropriate comparisons and conclusions.

Differences in the macroinvertebrate assemblages between streams were attributable to the differences in riparian vegetation cover, which affected the amount of coarse particulate organic matter (CPOM) available in the streams. None of the water quality parameters correlated with macroinvertebrate distributions, despite there being high concentrations of agricultural nutrients in all streams surveyed. Significant differences in the macroinvertebrate assemblages between streams were detected using the number of species, the number of families, or the number of Plecoptera, Ephemeroptera and Trichoptera species collected at a site. However, the popular index, SIGNAL, was unable to detect differences between the macroinvertebrate assemblages in Behana and Babinda Creeks.

Macroinvertebrate assemblages thus proved to be powerful indicators of stream ecosystem health. Our results show that there is a trade-off between the number of sites sampled and the level of detail necessary for each site. Thus, indicative indices can be used to detect differences if several sites are sampled in each stream. However, if only a few sites are sampled, or if a reference model is used, then species richness is the measure that will have the best likelihood of detecting differences between the streams.

Fish

This study found strong changes in freshwater fish assemblages associated with variation in habitat structure and position in the catchment. Between-stream comparisons of fish assemblages must therefore recognise differences in catchment size and position within the landscape, as noted for macroinvertebrates.

The composition of the fauna in Babinda Creek was different from that in Behana Creek, which it closely resembled in size and position within the landscape. The number of fish collected per site was significantly lower in Babinda Creek than in the other streams, like the invertebrate samples. The downstream change in observed/expected scores in Babinda Creek showed that assemblage composition changed with increasing loss of riparian integrity and increasing agricultural land use. Despite the presence of intense agricultural development, streams remained ‘healthy’ as long as riparian gallery forests remained in good condition and an adequate buffer was maintained between sugar-cane lands and the stream channel.

This study demonstrates the value of using fish as indicators of stream degradation resulting from catchment land use and riparian degradation. Fish assemblages were particularly responsive to the effects of degraded riparian systems on stream habitat structure. The abundance of alien fish species was also correlated with altered habitat conditions.

Status of the test streams

The components of this case study show that the observed impacts had a general effect across the spectrum of biophysical variables. Even in the presence of intense agricultural development, some streams (e.g., Behana Creek) could be 'healthy' as demonstrated by all biological indicators, providing that riparian vegetation remained in good condition and an adequate buffer between adjacent agricultural land and the stream channel was maintained. The major impact of a reduction in riparian integrity was loss of shade and detrital input and weed invasion.

It is apparent that while riparian vegetation had a strong influence on biodiversity, it had a limited effect on contamination from fertilisers when a large proportion of the catchment was used for agriculture. This is significant because riparian rehabilitation is touted as a key management tool in reducing contaminant loads entering streams. Thus, while riparian restoration is vital for aquatic and terrestrial habitat values, its role in substantial contaminant stripping in wet tropics streams remains to be demonstrated.

It is probable that streams of different character will respond differently to the potential stressors. Thus, elevated nutrient concentrations and hypoxia are much more prevalent in slow-flowing streams and riverine waterholes in the wet tropics (Pearson et al. 2003) and the dry tropics (unpublished data). In such situations, local factors may have greater influence than catchment-scale effects, and riparian condition may have greater influence on water quality than in perennial streams in the wet tropics.

Comparison of methods

Good correspondence between the study components suggests that monitoring objectives might be achieved by measuring one or two ecosystem response variables. However, each of the elements of this study was important in the assessment of ecosystem health: the physical description of the stream was necessary to classify streams to allow comparisons of like with like; water quality data were vital for characterising stream conditions and identifying possible sources of problems; riparian and instream macrophytes had an important bearing on conditions experienced by the fauna; invertebrates gave clear signals in this study, and confirmed the value of invertebrates in monitoring; similarly, fish were good indicators of stream conditions, albeit at a larger scale because of their mobility.

Therefore, a composite approach to health monitoring is preferred. However, more components mean greater complexity and expense in a monitoring program, so application of those components needs to be judicious and cost-effective.

Monitoring protocols

A monitoring manual based on these results will be produced in the MTSRF program. It will deal with the questions *why, what*, *how*, *when* and *how often* to sample and *who* to plan, administer and undertake the exercise. Our suite of recommended variables to measure ecosystem health in wet tropics streams is:

- flow regime and physical condition of the stream;
- water quality characteristics;
- riparian condition;
- aquatic macrophyte cover, species richness of aquatic macrophytes and proportion of aquatic macrophyte species that are alien;
- species or family richness of invertebrates;
- fish species richness and assemblage composition, number of alien fish species and proportion of fish abundance due to alien species.

These variables will be included in monitoring programs of waterways outside the wet tropics, with modifications to protocols to accommodate the different character of different systems. Methods for different types of system (e.g., wet *vs.* dry tropics) will be tested in the MTSRF program.

Chapter 1

Introduction: the *Catchment to Reef* Program and Stream Ecosystem Health Monitoring

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1.1 Background

The *Catchment to Reef* program was developed as a joint initiative of the Reef and Rainforest Cooperative Research Centres to address concerns that runoff from the catchments of the Great Barrier Reef (GBR) was severely degrading reef ecosystems, and affecting the industries and tourism enterprises that depend on reef health and biodiversity (Moss *et al*. 1993; Pearson and Stork 2007). Controversy about this issue led to the commissioning of an expert report to the Queensland Government (Baker 2003) that highlighted the problems. Subsequently, the Commonwealth and Queensland Governments, through the Great Barrier Reef Marine Park Authority, produced the *Reef Water Quality Protection Plan* (RWQPP), which recognised the need for process studies, long-term targets and monitoring programs for improving water quality in the GBR lagoon and its catchments.

The RWQPP represents a framework for water quality and land management programs to 'halt and reverse' the decline in the health of Queensland's catchments, freshwater ecosystems and inshore waters. The intent includes protection of the environmental quality of GBR catchments, where streams, rivers and wetlands have many important social, economic and ecological values and also provide ecological services (e.g. biological linkages, drainage, flood retention). The RWQPP is explicit in asserting that to protect the GBR it is also necessary to protect and manage the adjacent catchments (land systems, streams, wetlands and estuaries): firstly because they influence the Reef through delivery of suspended and dissolved materials, and provision of habitat for many species that move between waterways and the Reef; and secondly for their intrinsic values in sustaining freshwater species, biodiversity and ecological services.

The *Catchment to Reef* program aimed to develop appropriate monitoring methods for water quality and ecosystem health in aquatic ecosystems in the Wet Tropics and GBR World Heritage Areas. The three-year program was seen as an essential step towards minimising the downstream effects of agriculture and improving the ecosystem health of the GBR lagoon and its feeder catchments. Its goal was to provide a sound scientific basis for the development of monitoring tools, protocols and guidelines appropriate to the wet tropics. The research plan addressed this goal via seven tasks that encompassed the concept of a continuum of processes from catchment source to sea (see Pearson and Stork 2007). The individual tasks represented themes that were closely interlinked and overlapped operationally, derived from and extending Rainforest CRC and Reef CRC research. This report presents the results of the research undertaken in *Catchment to Reef* Task 3 – *River Health Assessment Tools.*

This research was designed to explore the concept of river health and to represent it as an integrated suite of protocols and techniques for biological river health assessment in wet tropics streams. The ultimate goal was the adoption of the methodology by relevant agencies and persons responsible for or interested in ecosystem health monitoring. This report represents the technical output of this research. A manual detailing protocols and techniques for river health assessment in wet tropics waterways, based on this research, will be produced through the Marine and Tropical Science Research Facility, which commenced in 2006. Other outputs to date, including scientific journal articles and popular material, are detailed in the Rainforest CRC's final Annual Report (www.jcu.edu.au/rainforest/) and will be updated on the *Catchment to Reef* website (www.catchmenttoreef.com.au/).

1.2 Concepts of Stream Ecosystem Health

1.2.1 Background

Concerns for the condition of rivers have been at the forefront of recent conservation and environmental movements across much of the planet (Bunn and Arthington 2002; Naiman *et al*. 2002; Postel and Richter 2003; Dudgeon *et al*. 2006). As rivers drain landscapes, river 'health' is regarded as a barometer of environmental health at a broad scale. Although the term 'health' is a value-laden concept with widely debated relevance to scientific perceptions of ecological systems (Fairweather 1999; Karr and Chu 1999; Norris and Thoms 1999), it provides a metaphor for human health that can be very effective as a means of communicating concepts, processes and values among scientists, managers and the community.

Costanza *et al*. (1997) defined health as a measure of the overall performance of a system that is built upon the behaviour of its component parts. The term 'river health' encompasses notions of river structure and function, availability and viability of habitat, river character and behaviour. The contemporary view of river health arose from the work of Rapport *et al*. (1998), who defined ecological integrity as comprising organisation (biodiversity, species richness, assemblage composition), vigour (rates of production, nutrient cycling) and resilience (the capacity to recover from disturbance). To maintain ecosystem integrity requires the maintenance of biophysical integrity, ensuring an appropriate level of integration between hydrological, geomorphic and biotic processes (Karr 1991, 1996; Petts 2000; Everard and Powell 2002).

1.2.2 Indicators of stream health

Various frameworks, tools and approaches for examining river health have been developed in several environmental disciplines (e.g., hydrology, geomorphology, water quality, ecology). Central to these has been the definition and monitoring of indicators of health or condition. A wide 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 and Thoms 1999; Gergel *et al*. 2002; Niemi and McDonald 2004).

Physical integrity is defined in geomorphic terms and encompasses the abiotic components of the system (Petts 2000). It reflects the physical characteristics and behaviour that maintain the habitats in which organisms live and complete their life processes. Physical character can be defined by the types, heterogeneity and spatial arrangements of landforms along rivers (termed 'geomorphic units') and the flow conditions (water quantity, the timing and frequency of events such as low flows and floods, and flow variability) that maintain and modify sedimentary processes and geomorphic structure (Newson and Newson 2000; Baron *et al*. 2002; Clarke *et al*. 2003; James and Marcus 2006). The complex of landforms and river habitats is regarded as a physical template that partly determines the biotic community characteristics of rivers (Townsend and Hildrew 1994; Brierley and Fryirs 2005) (biogeographic history and connectivity are examples of other determinants). Changes to the geomorphic structure of a river can modify and fragment the physical template, severely diminishing its capacity to support normal ecological systems and, therefore, their ecological integrity and value (Baron *et al*. 2002; Naiman *et al*. 2002; Nilsson *et al*. 2005. In many cases, alterations to the availability, spatial relationships and connectivity of aquatic habitats, and/or the biotic components of habitat such as fallen trees and woody debris, represent the primary constraints on improving biophysical condition (e.g., Pusey and Arthington 2003; Brooks *et al*. 2006). Various geomorphic indicators are available to assess the physical health of streams (e.g., Rosgen 1996) and include classification metrics such as sensitivity score, erosion score, geomorphic condition, and their interactions with habitat permanence and riparian condition (e.g., Werren and Arthington 2002).

Water quality assessment (i.e., physical and chemical measures of the water and the contaminants suspended or dissolved in it) has been the most generally used approach to describing the state of aquatic systems (see the accompanying Water Quality report from the *Catchment to Reef* Program). It entails measurement of selected variables that are likely to affect ecosystems and has the advantage of providing information that directly describes the behaviour of contaminants. It can have the added benefit of often being able to identify the source of the contaminant and track its progress from its source, through the catchment and to the sea. However, water quality measures are frequently unable to indicate the health of the ecosystem: to meet this goal, biological measures are required.

Typically, the main contaminants monitored are sediments and nutrients, as agricultural and urban supplementation of them is most strongly implicated in impacts on coastal systems. Sediments and nutrients also have impacts within streams, rivers and wetlands: normal quantities of sediment provide the substratum of the stream bed and nutrients provide vital elements for normal plant and microbial growth; however, excess sediments can change the character of stream habitats by smothering the existing substratum, and by making the water turbid, thereby reducing light and plant production. Excess nutrients encourage abnormal plant growth, altering habitats, blocking streams and regularly causing hypoxia. Water quality is not only related to increases in contaminant concentrations, it includes physical measures such as light environment and water temperature, both of which have major influence on the nature and dynamics of aquatic communities (and both of which, incidentally, can be strongly influenced by the nature of the riparian vegetation).

Water quality is, therefore, a complex issue. Variables do not act alone and the actual impact of a contaminant is sometimes hard to predict. For example, hypoxia that can result from eutrophication might not occur if the water is also very turbid. Contamination of water can be very short-lived as a result of short-term flood events that might carry substantial loads of contaminants to the sea but have little long-term impact on streams. On the other hand, smaller but chronic inputs of contaminants have the greatest effect on ambient conditions – the conditions under which freshwater plants and animals spend most of their lives. Moreover, the biological environment is not only affected by water quality – it also may be the major determinant of water quality, especially in the warm waters of the tropics. Thus, hypoxia, which is a predominant water quality factor in some tropical waterways (Pearson *et al*. 2003), results from respiration by blooms of algae, macrophytes and microbes, which in turn are enhanced by high levels of nutrients, organic inputs, temperature and light. Thus, assessments of water quality need to take into account not only the physical and chemical nature of a water body, but also its biological state and dynamics.

Whether the catchment or the reef is the main focus of interest will determine the style of monitoring (ambient or event), as discussed in the accompanying Water Quality report. Moreover, water quality variables themselves do not necessarily relate directly to the system's health (e.g., normal biodiversity and ecological processes). For example, enhanced nutrient levels do not directly affect invertebrates or fish – it is only through interlinked processes that effects are felt (Pearson and Connolly 2000; Pearson *et al*. 2003; Pusey *et al*. 2005; Kennard *et al*. 2006a,b). Therefore, contemporary assessments of river health incorporate both physico-chemical measures and measures of ecological integrity.

Cairns (1995) suggested that suitable biological indicators of aquatic ecosystem condition should: (i) be based on ecological knowledge and conceptual models of ecosystems; (ii) incorporate elements of biological structure, composition and function; (iii) be useful in waters other than those in which they have been developed; (iv) be diagnostic, heuristic or both; and (v) have sufficiently small sampling and annual variability to be responsive to marked differences or changes in habitat quality or disturbance levels. A wide range of aquatic organisms have been used as indicators of stream health, including algae, macrophytes, macroinvertebrates, fish and frogs (e.g., Pearson and Penridge 1987; Bunn 1995; Mackay *et al*. 2003). In addition, indicators of ecosystem processes (e.g., benthic metabolism) have been developed and applied (e.g., Bunn and Davies 2000; Fellows *et al*. 2006). Indicators are useful tools because, ideally, they have an observable, measurable quantity with significance beyond what is actually being measured. 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 *vs.* its sensitivity), so 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).

The concept of river health considers not only the structural integrity of stream ecosystems, but also functional aspects such as the resilience of the system – that is, its capacity to resist or overcome disturbances (Rapport *et al*. 1998). However, our understanding of the functional aspects of streams and how they are altered by land-use disturbance is largely conceptual, with little quantification. Gross changes, such as when riparian vegetation is cleared and allochthonous production is replaced by autochthonous production, are relatively straightforward (Bunn *et al*. 1998; Pusey and Arthington 2003), but measuring more subtle changes in ecosystem function is much more difficult. As a result, most studies of impact rely on detecting structural changes in the biotic assemblages present and infer functional changes through shifts in functional guilds and food-web structure (Pearson and Penridge 1987; Bunn 1995; Bunn *et al*. 1997). Recent work in the wet tropics aims to demonstrate explicitly the biotic responses to particular contaminants (Pearson and Connolly 2000; Pearson *et al*. 2003; Connolly *et al*. 2004; Connolly and Pearson 2007) but this approach is at an early stage of development.

1.2.3 Approaches to health assessment

Several multivariate techniques comparing assemblages across sites have been used, as have many univariate biotic measures or indicators, such as: the number of taxa; the ratio of observed taxa relative to expected (RIVPACS in the UK – Wright 1995; AusRivAS in Australia – Norris and Hawkins 2000); scores based on weighting taxa by their tolerance (Index of Biotic Integrity (IBI) – Karr 1991; SIGNAL – Chessman 1995); and the presence and relative abundance of alien taxa (Kennard *et al*. 2005). Less common have been the use of biological or ecological characteristics such as body size, life history and behavioural traits (Townsend and Hildrew 1994; Richards *et al*. 1997; Pan *et al*. 1999; Usseglio-Polatera *et al*. 2000). Habitat and water quality are also evaluated using a series of measures and indices (Barbour *et al*. 1999; ANZECC & ARMCANZ 2000) and ecosystem processes such as photosynthesis and respiration are gaining popularity (Bunn *et al*. 1998). The univariate indicators, usually summary metrics or indices, have gained favour because of their apparent simplicity enabling them to be specified in community monitoring protocols. However, the choice of indicators requires clear objectives and an assessment of what each measure can reveal about the influence of land use or other factors in the situation being studied. Only with a foundation of diagnostic research can an ecological indicator be used to identify particular stressors and be used for prescriptive management. Even then, because of their aggregated nature, the responses of summary metrics and indices may be less easily interpreted than those of individual variables (Watzin and McIntosh 1999). Therefore, users of monitoring systems need to be clear about whether they are satisfied with an assessment of relative condition or want to identify relationships between specific causes and effects. The latter will usually require a greater range and refinement of response variables. In reality, many programs, although referred to as monitoring programs, aim to identify the specific cause as well as the effect of land-use disturbance. If management and restoration actions are to be guided effectively, cause as well as harm must be diagnosed, which requires an improved understanding of the ecological mechanisms through which land use affects stream ecosystems. The objectives of any monitoring program must be defined carefully and methods and measures selected accordingly, otherwise the data collected may not address nor achieve these objectives.

Previous programs have had variable success in describing condition and determining the influences of land-use and other disturbances on streams. Resource and design limitations have led to data collection that is insufficient to diagnose cause and effect or even to detect effects. At times there has been an expectation that a few small-scale samples can describe complex large-scale patterns. Poor results do not always lead to improvement of subsequent sampling designs and choice of indicators, so these errors are frequently repeated, resulting in poor outcomes. The inconsistent performance of stream monitoring programs in Queensland has prompted programs specifically aimed at improving methodologies and developing and testing indicators to measure ecosystem health. One example is the DIBM3 initiative for south-eastern Queensland (*Design and Implementation of Baseline Monitoring – Developing an Ecosystem Health Monitoring Program for Rivers and Streams in South-east Queensland*) (Smith and Storey 2001), which provided a starting model for the *Catchment to Reef* study reported here.

1.3 Study Design

Different indicators can be expected to reveal different aspects of stream health. Therefore, our field study incorporated a suite of important physical, chemical and biological measures into an integrated framework that could be used to assess the health of stream systems relative to reference conditions, measured pressures and known disturbances. Accordingly, multiple landscape variables were measured using GIS, and major ecosystem components were examined at multiple sites: they included water quality, geomorphology and habitat structure, aquatic and riparian vegetation, macroinvertebrates and fish.

The study was based on 40 sites across four streams in the Russell-Mulgrave basin. This system was chosen because, firstly, it is central to the wet tropics, in some ways epitomising riverine habitats in the bioregion (e.g., strong perennial flow and high fish diversity – Pusey *et al*. 2004, 2007); secondly, the study team had substantial prior knowledge of the system; and, thirdly, contrasts in land management across the basin provided the possibility of finding good reference sites in contrasting paired catchments.

The concept and design of the field study were based on two complementary approaches. Firstly, the DIBM3 initiative, which had involved several participants in the *Catchment to Reef* program, provided an initial model for the study (see Smith and Storey 2001; Kennard *et al*. 2005, 2006a,b; Fellows *et al*. 2006; www.healthywaterways.org). The full suite of indicators could not be adopted directly because of the likely differences between south-eastern Queensland and wet tropics systems, and because of the much lower investment in a future monitoring program that would be possible in the whole GBR catchment. However, the DIBM3 model provided an appropriate starting point for our study design. DIBM3 demonstrated the poor ecological condition of waterways in south-eastern Queensland resulting from catchment land use, loss of or change in riparian vegetation, diffuse chemical and sediment inputs, water quality deterioration and modification of habitat structure and stability. With regard to monitoring methods, DIBM3 found that macroinvertebrates were sensitive to habitat and water quality deterioration; fish assemblages lost species and changed in composition with increasing land-use stress, often with invasion of alien species (Kennard *et al*. 2005, 2006a,b); ecosystem process indicators (e.g., primary production) were sensitive to disturbance gradients; and a large suite of potential indices could be distilled down to five themes (nutrients, other physico-chemical variables, macroinvertebrates, fish, and ecological processes) that gave a robust assessment of stream condition with strong linkages to stressors and fluxes such as diffuse nutrients coming off rural lands.

In the DIBM3 study, attributes of biotic assemblages at test sites were compared with attributes from a range of reference sites (see Stoddard *et al*. 2006 for discussion). Accurately defining the expected condition requires that natural spatial and temporal variation in selected biological attributes, driven by variation in environmental conditions, can be accounted for, such that impacts of human-induced disturbance can be accurately assessed (Resh and 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 affected 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. Surveys to establish the reference condition need to be extensive so as to incorporate spatial and temporal variation in the physical and biological characteristics of aquatic systems. A reference condition may also be determined by means of a model predicting the species richness or biotic structure that would naturally occur at a site given a number of environmental variables (Kennard *et al*. 2006a,b); however, this approach depends on well-developed models based on extensive datasets that are essentially multiple reference sites. A referential approach involving the use of predictive models of expected diversity and assemblage structure was undertaken for the fish component of this study (Chapter 5), for which there was a substantial reference database available.

The second approach in our field study took advantage of the juxtaposition of disturbed and less disturbed subcatchments, which provided the opportunity to undertake a paired-catchment comparison. This addressed four specific challenges that need to be considered when investigating the influence of land use on stream ecosystems (Allan 2004): co-variation of anthropogenic and natural landscape features; spatial scale; non-linearities of responses; and legacy effects from previous impacts. These offer a useful set of considerations when designing individual studies or an overall framework to investigate the effects of land use that might otherwise mask patterns and confound data analysis making it impossible to detect effects. We aimed to sample at a scale relevant to the detection of large-scale patterns of biotic distributions and land-use impacts. It was important where possible to account for strong natural longitudinal gradients in the stream reaches crossing the floodplain of wet tropics catchments, where agricultural development was extensive and impacts were most intense. In these coastal systems the natural longitudinal gradient co-varies with the amount of agriculture in the catchment, affecting the level of impact. This means that comparisons of sites within a stream can be confounded by both natural and anthropogenic factors. Therefore, comparisons were made between impacted and non-impacted streams, at the same time as accounting for the natural longitudinal gradient. The most effective way to account for this natural gradient is to sample several sites along it and to use a covariance analysis to separate disturbance effects from natural longitudinal variation. This approach was particularly useful for the macroinvertebrates (Chapter 4) where high abundance and taxonomic diversity occurs at most sites.

Emphasis was placed on the assessment of physical and biological indicators of response to agricultural land use as the primary stressor, in the lowland agricultural catchments of the Russel-Mulgrave basin. Land-use management in the Mulgrave catchment approaches current 'best management practice' (BMP) with regard to protecting river health, whereas BMP in the Russell catchment is less well developed, with obvious visual impacts on the streams in that catchment. For example, streams in the Mulgrave catchment had mostly intact and continuous riparian vegetation, whereas the riparian vegetation of streams in the Russell catchment was highly disturbed, dominated by invasive grasses and other weeds, with large trees occurring only in occasional patches (Chapter 3). During heavy rainfall there are obvious differences in the colour of water in the streams of these two catchments, with streams in the Russell catchment carrying high loads of suspended sediments for extended periods. In contrast, streams in the Mulgrave catchment are relatively clear during extended rainfall periods.

It was predicted that the ecological health of streams in the Russell catchment would be poorer than that in streams of the Mulgrave catchment, with measurable impacts on biodiversity in these streams. The Russell and Mulgrave catchments provided the opportunity to build models of biotic distribution and determine cause-and-effect relationships with land-use change. These adjacent catchments are very similar in key attributes (size, topography, rainfall etc. (Chapter 2) and so enable a valid comparison between streams. As the Mulgrave catchment has been managed well, it is particularly important as it provides near-natural systems that can be used as a reference to compare with more impacted systems. Streams in the Mulgrave catchment deserve special attention and protection for this reason alone. Without a measure of conditions before land-use change and other developments, we have no other means to assess the extent of the changes in disturbed stream systems (Stoddard *et al*. 2006).

The field trials were conducted in 2005 during late June and early July, the winter (low flow) period in the wet tropics. Results from DIBM3 showed that under stable flow conditions (also the winter), monitoring provided a reliable set of ecosystem health signals that could be interpreted in terms of disturbance pressures rather than natural environmental variability. Previous work indicates that similar conditions apply in the wet tropics (Pearson *et al*. 1986; Benson and Pearson 1987; Pearson and Penridge 1987, 1992; Kapizke *et al*. 1998).

1.4 Aims

This study investigated the chain of influence from land-use to stream ecosystem response, via the responses of individual ecosystem components, to understand how these influences operate and to underpin the development of monitoring tools and guidelines appropriate to wet tropics streams (and eventually to other GBR and tropical catchments). It compiles accumulated knowledge of the ecology of wet tropics streams. The study fills a gap in our understanding of these streams and provides empirical relationships that will enable us to use existing broad-scale datasets to provide a wider geographic context for the findings of the study and allow greater generalisation of the results. It is also the first coordinated investigation of wet tropics streams where a variety of ecosystem components (geomorphology, water quality, aquatic and riparian vegetation, invertebrates and fish) have been surveyed simultaneously at multiple sites to describe natural gradients and to determine the effects of land-use and related stressors. The results provide a much needed benchmark description of the aquatic biota and environmental characteristics of these systems. The study also provides strong evidence of cause-and-effect relationships between natural biophysical variables and stressor variables representing land-use patterns and effects in tropical catchments.

This study addressed the concerns raised by Allan (2004) regarding separation of natural and stress gradients and responses, by undertaking an intensive sampling program, in which sample sites were nested within streams within catchments, and dispersed along natural gradients. This approach allowed us to address specific questions regarding gradients, covariance and impacts that have been difficult to address in many studies.

The specific objectives of the case study were:

- to measure natural distributions of biotic assemblages in wet tropics streams particularly with regard to natural physical gradients;

- to associate changes in biotic assemblage structure influenced by land-use activities with specific causal factors and investigate the mechanisms by which these effects operate;
- to investigate the nature of specific responses – that is, the relationship between cause and effect, whether linear or nonlinear;
- to test the utility of ecosystem components as indicators to detect land-use impacts;
- to develop a preliminary integrated model and suite of protocols and techniques for river health assessment in wet tropics streams;
- to contribute a sound basis for prescriptive management of streams in the wet tropics.

This report presents the results of the different components of the study: the geographic setting and the results of hydrological, geomorphologic and water quality investigations (Chapter 2); the aquatic macrophytes (Chapter 3); the macroinvertebrates (Chapter 4); and the fish (Chapter 5). The report ends with a summary of findings, compares outcomes from each component, examines future applications of the results and outlines future research requirements (Chapter 6).

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Chapter 2

Hydrology, Geomorphology and Water Quality of Four Wet Tropics Streams with Contrasting Land-use Management.

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2.1 Introduction

Physical and chemical characteristics structure stream habitats and have a major influence on aquatic communities and ecological processes (e.g., Hynes 1970, Cummins 1974). The nature of stream habitats is largely determined by water flow and sediment transport patterns, which are controlled by the climate, topography, geology, soils and vegetation characteristics of the stream valley. Different physical settings and hydrological regimes result in different fluvial processes, terrestrial and aquatic vegetation and, therefore, habitat types. Consequently, variation in the distributions of organisms can be explained substantially by the physical character of the habitat, including habitat alterations by vegetation, which is itself determined by physical and chemical conditions and processes in the catchment.

This biophysical habitat is frequently modified by human activities, with subsequent effects on the biotic assemblages. Land-use changes in a catchment alter the hydrological and geomorphologic processes, such as the flow regime, the particle sizes and distribution of substrata, and the availability of nutrients and energy that support food webs. When forest is cleared, hydrological processes are affected by changes in soil infiltration and evapotranspiration by trees. The pathway the water takes to the stream, and its interactions with physical, chemical and biological processes along the way, all affect its quality by changing the concentrations and types of dissolved and suspended materials it carries. Geomorphic processes, such as surface or bank erosion, may be altered, affecting the amount of suspended material transported to the stream channel. Channel form is altered by destabilising of banks through removal of riparian vegetation, affecting stream width-to-depth ratios and bank habitats. When weeds such as para grass (*Urochloa mutica*), colonise stream banks and bars they transform edge habitats and deflect flow, further modifying the channel, habitats and ecological processes.

Physico-chemical and vegetation characteristics are, therefore, important in the concept of 'river health' and it is crucial to understand the underlying biophysical character of the environment before other influences can be discerned. In this chapter we describe the physico-chemical environment of the study area for the Russell-Mulgrave case study and present the results of geomorphologic and water quality surveys. The aim of the study was to describe the physical and chemical characteristics of the study area, their variation and gradients, and any changes attributable to land-use management practices that might identify cause-effect relationships between land use and the condition of biotic components of the stream ecosystems. This study also identifies important physico-chemical variables and the 124unique character of these wet tropics streams for development into future river health programs. It is significant in a regional context because of the dearth of information of this type, especially in combination with biological data.

2.2 The Russell-Mulgrave study area

This case study was undertaken in four streams in the Russell-Mulgrave catchment, which lies approximately in the middle of the wet tropics latitudinal range, just south of Cairns (Figures 2.1 and 2.2). It is one of the three major catchments in the wet tropics in terms of area, rainfall and discharge to the Great Barrier Reef (GBR) lagoon (Table 2.1) (the other two are the Johnstone and the Tully), with over 60% of rainfall converted to runoff. The Herbert River also has high discharge but is less typical of the wet tropics because much of its upper catchment is outside the bioregion and is seasonally dry.

The Russell and Mulgrave rivers are approximately 65 km and 79 km long, with catchment areas of 560 km^2 and 807 km^2, respectively. They drain the eastern escarpment of the Great Dividing Range, flowing on either side of the mountain massif of Mt Bartle Frere and Mt Bellenden Ker, the two highest peaks in Queensland. During the late Pleistocene (500 000 to 10 000 years ago), the Mulgrave River flowed into Trinity Inlet but was deflected as a result of its own deposition (Willmott and Stephenson 1989) to join the Russell River near Deeral, shortly before discharging through Mutchuro Inlet into the GBR lagoon.

The Russell-Mulgrave catchment is contained almost wholly within the Cairns City Local Government Area, with small parts extending into Eacham and Johnstone Shires. The total human population of this area is approximately 7,200 but appears to be growing rapidly, with increasing areas being converted for rural-residential development. The main towns are Gordonvale and Babinda, with several other smaller settlements.

The region is warm and humid, with annual rainfall ranging from less than 1900 mm (at the Meringa Northern Sugar Experiment Station – Tracey 1982) to more than 8000 mm at the summit of Mount Bellenden Ker, with a mean annual average of 3233 mm (Hausler 1990) (Figure 2.3). Babinda, in the Russell River catchment, competes with Tully for the highest annual average rainfall for any Australian town (at 3016 mm – Furnas 2003; but also estimated at 4174 mm – Tracey 1982). Rainfall is seasonal with 60% falling in the summer wet season, December to March, but high rainfall can occur during other months (e.g., in association with Cyclone Monica on 19th April 2006).

The Russell-Mulgrave catchment includes a range of landforms, commencing in the mountain range through which the streams have cut deep valleys and gorges. Upper sections also drain parts of the rolling uplands of the Atherton Tablelands. The streams descend quickly down the mountain range or escarpment then abruptly change slope to flow across a flat and narrow coastal floodplain. The lower sections are characterised by colluvial-alluvial plains, freshwater wetlands, tidal estuaries and beach ridge systems (McDonald 1994). The catchment is divided into seven sections according to its physiography and land use (Table 2.2). This study was undertaken within the Lower Northern and Lower Southern sections.

The Mt Bartle Frere – Mt Bellenden Ker mountain massif consists mainly of acid plutonic granites (Figure 2.4). However, both the Russell and the Mulgrave Rivers have basalt and mudrock in their upper catchments associated with volcanism that formed the Atherton Tablelands. The coastal Malbon-Thompson Range and northern section of the Graham Range consist mainly of granites, whereas the southern section of the Graham Range and the Seymour Range and Southern Watershed sections consist predominantly of metamorphic material. The dominant soils in the floodplain are deep friable yellow or yellowish red loams, with lesser red loams. Lower areas of the Russell catchment that are inundated for considerable periods of the year also contain loamy organic soils with some areas of peat.

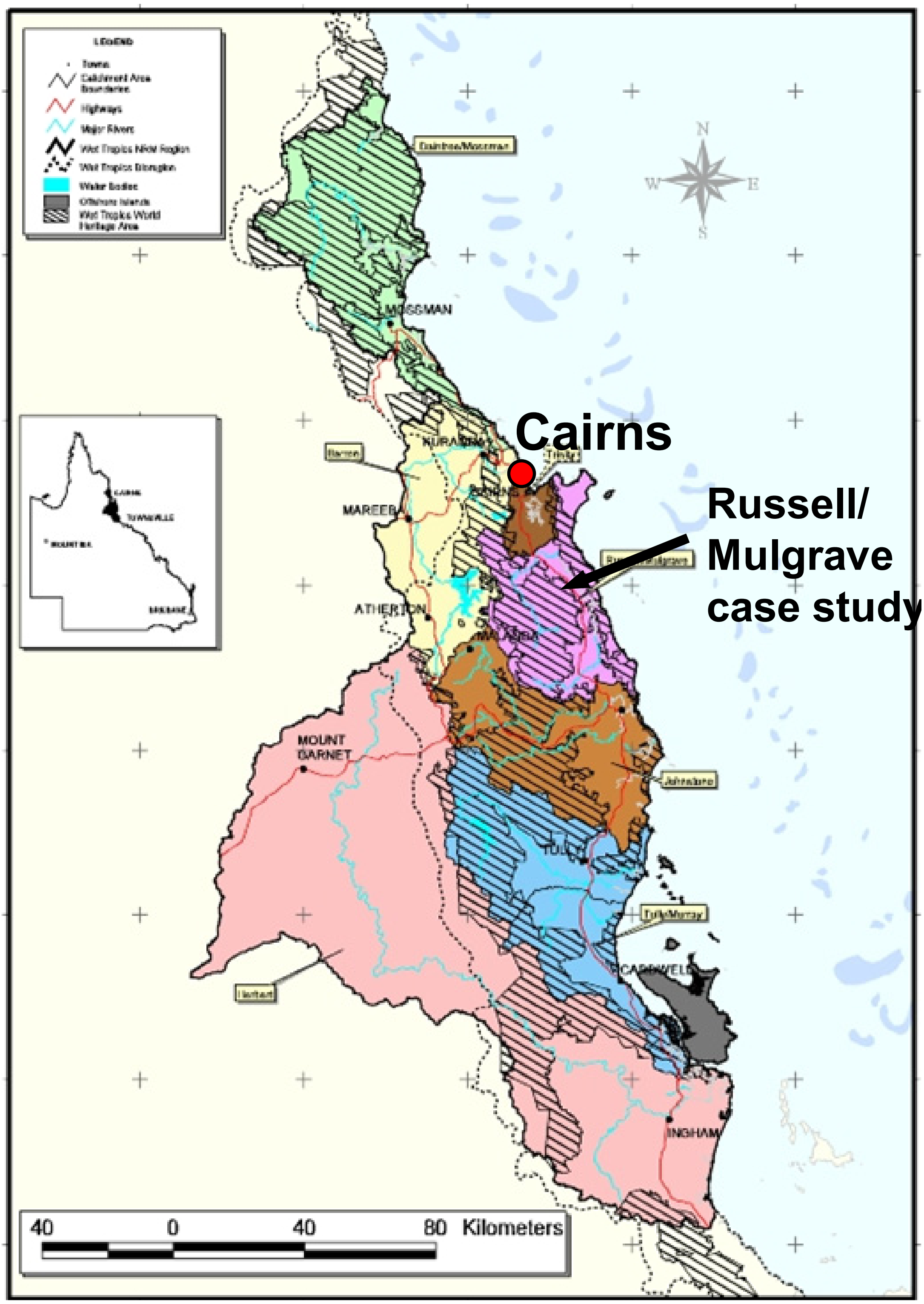


Figure 2.1. Wet tropics catchments and the Russell-Mulgrave catchment study area. Cross-hatching indicates the Wet Tropics World Heritage Area

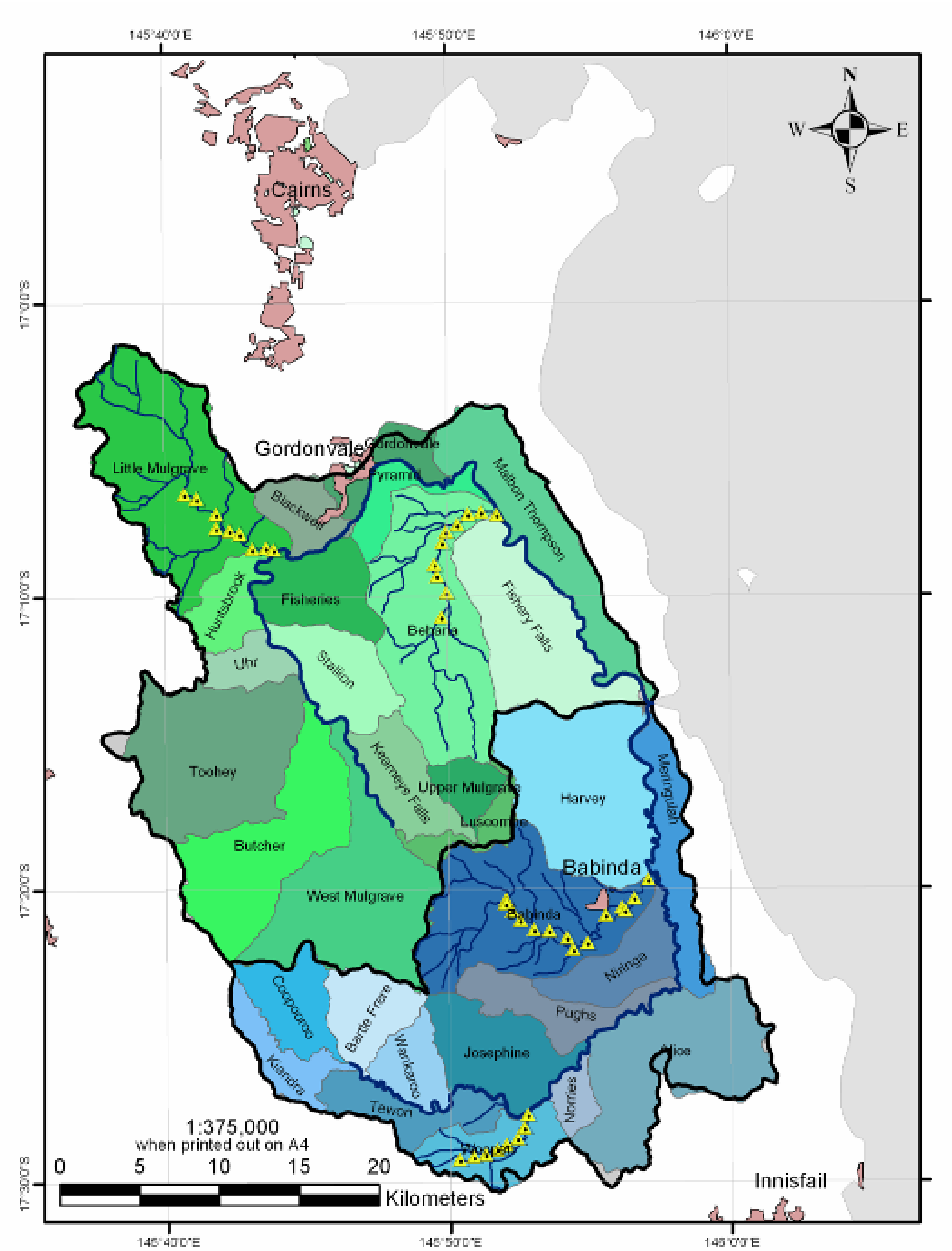


Figure 2.2. Russell-Mulgrave sub-catchments. Study sites are indicated by yellow triangles in this and subsequent figures.

Table 2.1. Summary rainfall and discharge statistics for east-flowing catchments in the wet tropics and for the Burdekin and Fitzroy rivers. Basin areas and gauged runoff data (1968 – 1994) were obtained from the Queensland Department of Natural Resources and Water. Rainfall data were obtained from the Bureau of Meteorology. Data sourced from Furnas (2003).

		Average annual			Annual Discharge (km^3)					Mean Annual Exports			
Basin Name	**Area (km^2)**	**Rainfall (mm)**	**Runoff (mm)**	**% Runoff**	**Average**	**Max.**	**Min.**	**Total (1968-1994)**	**Fine sed. (10^6 tonnes)**	**DIN export (t/y)**	**DON export (t/y)**	**PN export (t/y)**	**Total P export**
Daintree R.	2,192	2,492	1,575	63	1.26	3.56	0.11		0.05	215	100	183	53
Mossman R.	466	2,208	1,265	57	0.59	1.21	0.18		0.02	101	47	86	25
Barron R.	2,136	1,453	279	19[2]	0.81	2.66	0.16		0.03	139	64	118	34
Mulgrave-Russell R.	**1,983**	**3,016**	**1,836**	**61**	**3.64**	**7.21**	**1.32**	**94.6**	**0.14**	**622**	**289**	**529**	**153**
Johnstone R.	2,325	2,996	2,009	67	4.67	9.12	1.65		0.18	799	371	679	196
Tully R.	1,683	2,855	1,954	68	3.29	5.37	1.24		0.13	563	262	478	138
Murray R.	1,107	2,098	958	46	1.06	2.60	0.38		0.04	181	84	154	44
Herbert R.	9,843	1,506	407	27	4.01	11.99	0.53		0.54	686	319	583	168
Wet tropics[1]	**21,735**	**18,624**	**10,283**	**55[2]**	**19.33**	**43.72[3]**	**5.57[3]**	**521.9**	**1.13**	**3,306**	**1,536**	**2,810**	**811**
Burdekin R.	130,126	727	79	11	10.29	54.46	0.52	277.8	3.77	2,027	1,430	5,176	1,695
Fitzroy R.	142,537	735	43	6	6.08	23.22	0.18	164.2	2.23	1,198	845	3,058	1,001

[1] Main rivers only. [2] Underestimate due to abstraction from Tinaroo Falls Dam. [3] Maximum and minimum probably exaggerated (above values summed).

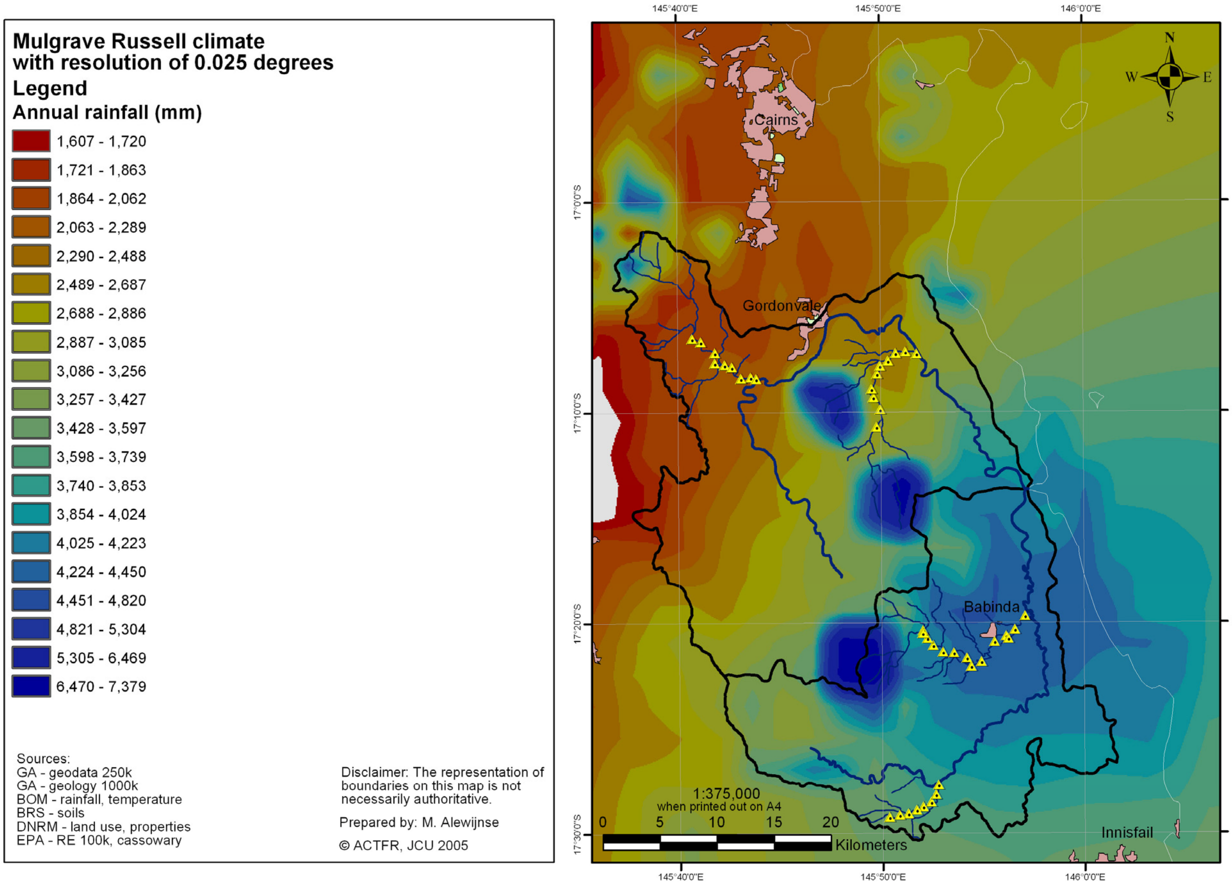


Figure 2.3. Russell-Mulgrave annual rainfall patterns.

The steep, mountainous parts of the catchments are largely protected within the Wet Tropics World Heritage Area as part of the Wooroonooran National Park. Some agricultural areas of the Atherton Tablelands are located in the upper reaches of the Russell and Mulgrave Rivers. The floodplains to the east of the range have been almost completely cleared for sugar cane and other agriculture. The Russell-Mulgrave catchment has the third largest area under sugar-cane production of the wet tropics basins, after the Herbert and Johnstone River catchments (Table 2.3). Additionally, extensive areas are used for growing bananas and other fruit crops, grazing and timber production. Small areas are used for flower production and large areas have been converted to grow turf in the Little Mulgrave River subcatchment.

The hydrology of the lower sections of the catchment has been significantly altered. There has been extensive drainage of wetlands and some redirection of surface flow through the construction of levee banks and road construction. Water is abstracted from Behana Creek for urban water supply.

Table 2.2. Landscape units of the Russell-Mulgrave catchment basin. Sourced from NQ Joint Board (Russell-Mulgrave Catchment Rehabilitation Plan, (date unknown)).

Upper (Atherton Tablelands) Section	The western rim of the catchment from Kalimna through Butchers Creek to Topaz, embracing a volcanic area characterised by undulating cleared uplands and headwater streams.
Central (Bellenden Ker Range) Section	The major part of the catchment taking in extensively forested Bellenden Ker Massif (mostly within Wooroonooran NP) and foothills, from just east of the Mt Haig portion of the Lamb Range across to Djarrugan (Walsh's Pyramid) in the north to the Twin Pinnacles of the Francis Range on the southern watershed, through which high gradient streams cut incised channels.
Lower Northern (Mulgrave Floodplain) Section	Comprising the depositional environment of the narrow coastal plain of the Mulgrave River from Gordonvale to Cucania, south of Deeral, which is extensively cultivated to the lower forested coastal ranges that confine the lower river reaches oriented parallel to the coast.
Lower Southern (Russell Floodplain) Section	That part surrounding the Russell River's lower reaches from Cucania south-west to Woopen Creek and south-east to the watershed east of Eubanangee Swamp, which is similar to the above section.
Northern Coastal (Malbon-Thomson) Range Section	Comprising the adjacent minor catchments draining the eastern fall of the southern part of the Malbon-Thompson Range and its southernmost lobe from Palmer Point to Flirt Point.
Southern Coastal (Graham & Seymour) Range Section	Including all of the Graham Range and that part of the Seymour Range that drains coastward into the Ella Bay swamps.
Southern (Francis Range Outlier) Watershed Section	Consisting of a hilly forested low range fanning out from Mt Chalmynia between Woopen Creek and Waugh's Pocket Roads.

2.3 Study sites

The four streams surveyed were the Little Mulgrave River and Behana Creek in the Mulgrave River catchment and Babinda Creek and Woopen Creek in the Russell River catchment (Figure 2.2). Babinda Creek rises near the summit of Mount Bartle Frere and Behana Creek rises near the summit of Mount Bellenden Ker. These streams, therefore, represent the two highest streams, or greatest altitudinal range, of any in Queensland. The Little Mulgrave River rises in the Lamb Range to the north of the Mulgrave River. Woopen Creek rises from Twin Peaks in the Francis Range to the south of the Russell River.

These streams were chosen because of the contrast in land-use management practices in their catchments. The Little Mulgrave River and Behana Creek have mostly intact riparian strips that are continuous, actively maintained, and steadily improved with replanting schemes initiated over the last decade or so. Land-use management in their catchments approaches current 'best management practice' (BMP) with regard to protecting river health. They are, therefore, particularly important because they represent near-natural systems that can be used as a reference (control) to compare with more affected systems in the Russell catchment. Across most of the lower floodplain sections of Babinda and Woopen Creeks the riparian vegetation is highly disturbed, dominated by invasive grasses and weeds such as Singapore daisy, *Sphagneticola trilobata* (Figure 2.5), with large trees occurring only in occasional patches (Figure 2.6) (MacKay *et al.*, Chapter 3). Bank erosion is obvious in many places and stretches of streams have been channelised by stands of para grass, *Urochloa mutica*, on banks and bars. Observations during rainfall events indicated that Babinda and

Woopen Creeks carry higher suspended sediment loads for longer periods than the Little Mulgrave River or Behana Creek. This suggested that water quality also differed in streams in the Mulgrave catchment compared with streams in the Russell catchment. It was predicted therefore, that the ecosystem health of Babinda and Woopen Creeks would be poorer than that of the Little Mulgrave River and Behana Creek, with measurable impacts on biodiversity in these systems.

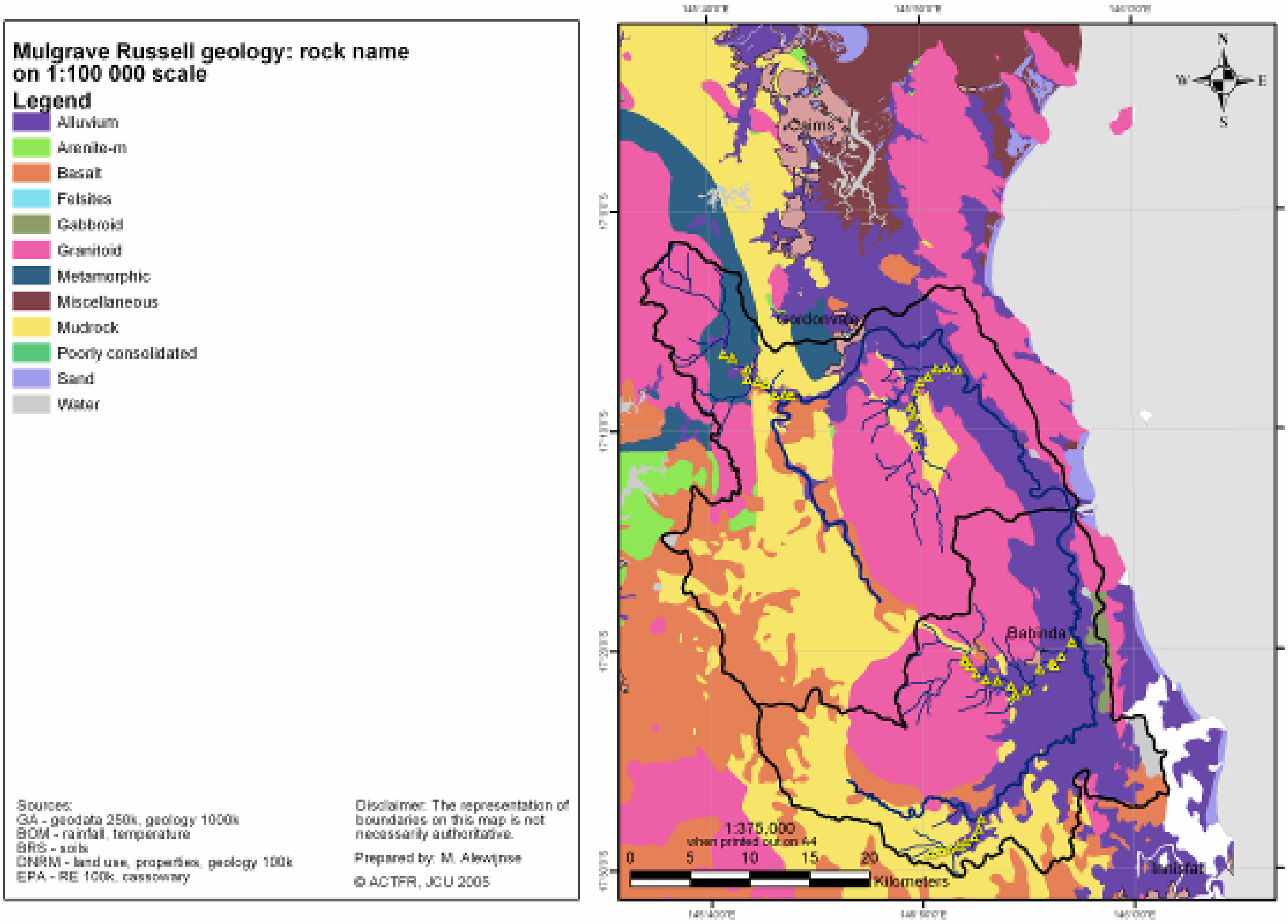


Figure 2.4. Geology of the Russell-Mulgrave catchment.

Table 2.3. Land-use cover in the catchments of the wet tropics. Data sourced from Furnas (2003). Note: Total catchment area includes residential, industrial and other land uses that are not shown.

Basin Name	Area (km^2)	Major land use (km^2) Sugar	Grazing	Timber	Horticulture	Other Cropping	Reserve
Daintree R.	2,192	38.3	569	803	0.9	0.6	693
Mossman R.	466	49.0	219	127	0.3	-	80
Barron R.	2,136	45.8	1,040	831	12.0	91.1	40
Russell-Mulgrave R.	**1,983**	**264.6**	**786**	**346**	**3.0**	**0.4**	**573**
Johnstone R.	2,325	344.8	969	613	20.7	4.9	279
Tully R.	1,683	162.2	350	1,031	25.9	-	
Murray R.	1,107	69.5	337	362	10.0	0.3	334
Herbert R.	9,843	668.6	7,202	991	0.0	36.5	642
Wet tropics	**21,735**	**1642.8**	**11,472**	**5104**	***72.8***	**133.8**	**2641**

The study area was restricted to the lowland floodplain sections of the study streams because this is where the major land-use impacts were occurring. The upper sections of all four streams are protected within reserves as part of the Wet Tropics World Heritage Area. Two sites in each stream were located upstream of reserve boundaries to assess biodiversity and physico-chemical parameters as each stream entered the floodplain. It was expected that these parameters would vary naturally owing to the strong longitudinal gradients in physical and biological attributes along these streams. Anthropogenic influences also increase with distance downstream as the area of agriculture in the catchment increases, so it was necessary to sample with adequate intensity to account for these gradients in subsequent analyses. Consequently, sampling sites were distributed at approximately 1 to 1.5 km intervals along each stream, from the base of the foothills to the confluence with the Mulgrave or Russell Rivers (Table 2.4; Figure 2.7).

Figure 2.5. Singapore Daisy, *Sphagneticola trilobata*.

Figure 2.6. View of Site 34 in Woopen Creek.

2.4 Methods

2.4.1 Catchment characteristics

Catchment characteristics were determined by GIS for the Russell-Mulgrave catchment and for the upstream catchment areas for each of the study sites. The upstream catchment area for each point was manually created using ESRI's ArcMap 9.1 software, a 25m digital elevation model from the Department of Natural Resources and Water (DNRW), and using 1:50 000 drainage line data from the Wet Tropics Management Authority (WTMA). All data were in a GDA94 geographic coordinate system.

After creating the upstream areas for each of the sample points, the shapefile was overlayed with the DNRW QLUMP land-use data from 2005, using the intersect command. This combined layer was then re-projected to the GDA94 MGA zone 55 projected coordinate system, to be able to calculate the surface area in square metres. Because of the manual creation of the subcatchment areas, with data that were sometimes limited in detail, an error of 0.1 km^2 can be expected.

Land use was described in terms of seven broad categories: conservation (including State Forest and National Parks), sugar cane, other cropping, grazing, residential/rural-residential, industrial and storage (Table 2.5). Sugar cane was separated from other crops as it was the dominant crop grown in the region (Russell *et al*. 1996). These data, combined with other physico-chemical data, contributed to the assembly of an environmental variables data matrix for subsequent analysis with other ecosystem components of this study. In this report these data were used to compare nutrient concentrations relative to the proportion of catchment area used for agriculture, similar to Bramley and Roth (2002), to contrast study streams with different riparian vegetation cover and land-use management practices. Riparian condition was assessed at 38 of the study sites using the assessment protocol of Werren and Arthington (2002),as described by MacKay *et al.* (Chapter 3).

Table 2.4. Locations of sample sites, and ecosystem descriptors measured at each. Additional water quality samples were collected at sites located in between the main study sites during canoe traverses of the Little Mulgrave River, Behana Creek and Babinda Creek.

Catchment	Stream	Site number	Latitude (dec.deg.)	Longitude (dec. deg.)	Catch area above site (km^2)	Geomorphology (this chapter)	Water Quality (this chapter)	Riparian Vegetation (chapter 3)	Aquatic Vegetation (chapter 3)	Macroinverteb-rates (chapter 4)	Fish (chapter 5)
Mulgrave River	Little Mulgrave River	**37**	17 06.486	145 40.719	65.30	✓				✓	
		1	17 06.650	145 41.150	67.12	✓		✓	✓	✓	✓
		2	17 07.180	145 41.835	72.95	✓	✓	✓	✓	✓	✓
		3	17 07.653	145 41.829	73.60	✓		✓	✓	✓	✓
		4	17 07.743	145 42.312	99.20	✓	✓	✓	✓	✓	✓
		5	17 07.846	145 42.667	101.01	✓	✓	✓	✓	✓	✓
		6	17 08.380	145 43.123	104.20	✓	✓	✓	✓	✓	✓
		7	17 08.333	145 43.600	106.12	✓	✓	✓	✓	✓	✓
		8	17 08.405	145 43.865	107.01	✓	✓	✓	✓	✓	✓
	Behana Creek	**27**	17 10.717	145 49.798	46.46	✓		✓	✓	✓	
		21	17 09.887	145 49.975	59.75	✓	✓	✓	✓	✓	✓
		22	17 09.315	145 49.651	63.11	✓	✓	✓	✓	✓	✓
		23	17 08.930	145 49.550	66.83	✓	✓			✓	
		24	17 08.184	145 49.829	69.10	✓	✓	✓	✓	✓	✓
		25	17 07.804	145 49.983	70.50	✓		✓	✓	✓	✓
		26	17 07.569	145 50.372	85.53	✓	✓	✓	✓	✓	✓
		28	17 07.213	145 50.755	94.81	✓	✓	✓	✓	✓	
		29	17 07.147	145 51.234	96.50	✓		✓	✓	✓	✓
		30	17 07.234	145 51.782	98.30	✓	✓	✓	✓	✓	✓
Russell River	Babinda Creek	**11**	17 20.377	145 52.054	14.92	✓				✓	
		40	17 20.527	145 52.019	17.01	✓	✓			✓	
		10	17 20.767	145 52.258	37.34	✓	✓	✓	✓	✓	✓
		9	17 21.085	145 52.543	39.26	✓	✓	✓	✓	✓	✓
		12	17 21.413	145 53.033	50.48	✓	✓	✓	✓	✓	
		13	17 21.450	145 53.563	60.07	✓		✓	✓	✓	
		14	17 21.679	145 54.211	62.64	✓	✓	✓	✓	✓	✓
		15	17 22.088	145 54.422	69.16	✓		✓	✓	✓	
		16	17 21.858	145 54.929	70.82	✓	✓	✓	✓	✓	✓
		17	17 20.926	145 55.579	80.55	✓	✓	✓	✓	✓	
		18	17 20.764	145 56.257	83.87	✓	✓	✓	✓	✓	✓
		19	17 20.334	145 56.579	87.93	✓	✓	✓	✓	✓	
		20	17 19.698	145 57.085	91.84	✓		✓	✓	✓	
	Woopen Creek	**38**	17 29.255	145 50.349	1.01	✓	✓	✓	✓	✓	
		39	17 29.156	145 50.842	3.02	✓	✓	✓	✓	✓	
		31	17 29.096	145 51.282	9.83	✓		✓	✓	✓	✓
		36	17 28.917	145 51.683	10.05	✓		✓	✓	✓	✓
		32	17 28.762	145 52.012	11.50	✓	✓	✓	✓	✓	✓
		33	17 28.543	145 52.416	11.76	✓	✓	✓	✓	✓	✓
		34	17 28.175	145 52.643	14.01	✓	✓	✓	✓	✓	✓
		35	17 27.715	145 52.765	26.76	✓	✓	✓	✓	✓	✓

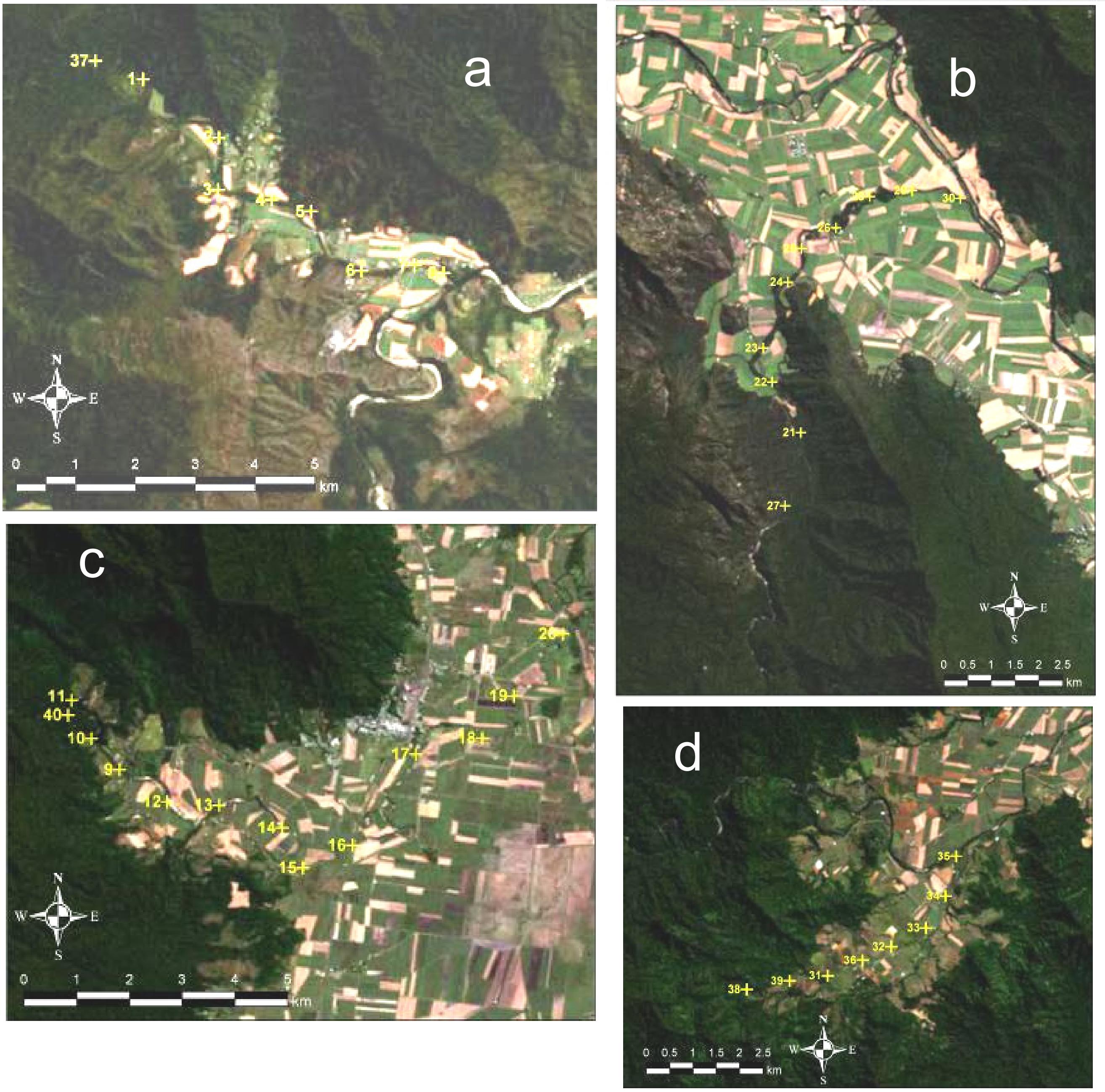


Figure 2.7. Aerial views of (a) the Little Mulgrave River, (b) Behana Creek, (c) Babinda Creek and (d) Woopen Creek, showing study site locations.

Table 2.5. Land-use categories used in GIS assessment.

Land-use Type	Abbreviation	Categories Included in Land-use Type
Conservation	CONSERV	1.1.3 National Park
		1.1.4 Natural feature protection
		1.3.0 Conservation, minimal use
		1.3.3 Conservation: remnant native cover
Sugar	SUGAR	3.3.5 Sugar
Other Cropping/Horticulture	OTH_CROP	4.3.0 Irrigated cropping
		4.4.1 Irrigated tree fruits
		5.1.0 Intensive horticulture
Plantation	PLANTAT	3.1.0 Plantation forestry
Grazing	GRAZE	2.1.0 Grazing natural vegetation
		3.2.0 Grazing modified pastures
Residential	RESID	5.4.0 Residential
		5.4.2 Rural residential
		5.5.3 Recreation and culture
Industrial/Commercial	INDUST	5.3.0 Manufacturing and industrial
		5.5.0 Services
		5.5.1 Commercial services
		5.5.2 Public services
		5.8.0 Mining
		5.9.0 Intensive uses: waste treatment and disposal
Reservoirs/dams	STORAGE	6.2.0 Reservoirs and dams

2.4.2 Hydrology

Stream gauging data were obtained from the DNRW gauge stations. No assessment of groundwater hydrology was undertaken in this study. Only limited hydrological data were available for the study streams. A single gauging station (111105A) currently operates on the upper section of Babinda Creek, located downstream from the Boulders near the base of the range, approximately 100 m downstream from study site 10. Another gauge was located in Babinda Creek near Babinda township (111102B), but ceased operating in 1988. A gauge in Behana Creek at Aloomba (111003C) ceased operating in 1971. There are flow data from Cairns Water (Cairns City Council) for the water intake at the Behana Creek gorge. However, this gauge was not operating during the study period. Rainfall data were obtained from the Bureau of Meteorology for rain gauge sites at Babinda Post Office, Deeral, Gordonvale, and for the top and bottom sites located on Mount Bellenden Ker.

2.4.3 Geomorphology and hydraulic parameters

The channel planform for each study stream was assessed from aerial photographs. Only rudimentary assessment was made to identify general patterns of sinuosity.

A preliminary geomorphic assessment was carried out at the 40 study sites. A pool-riffle-run sequence within a 100-m reach was surveyed at each site. This was modified to include a pool-riffle-cascade in upstream reaches and a pool-run sequence in downstream reaches where pool and riffles were spaced further than 100 m apart. This survey involved a combination of qualitative, semi-quantitative (ratings system) and quantitative (cross-sectional measurements and sediment particle size classification) measures in the four streams. Rapid assessment techniques combining elements of the geomorphic methods used in the State of the Rivers and AusRivas programs were employed. Additional bank stability rating scores were developed for this study. Observations were made of current *in situ* geomorphic characteristics and active processes, including the degree of undercutting of banks, flow diversions resulting from obstructions and the degree of recent erosion. An assessment of the riparian vegetation at the study sites was undertaken concurrently (MacKay *et al*., Chapter 3).

2.4.4 Stream channel geometry and hydraulics

Detailed channel cross sections were measured at 31 of the study sites to provide comparisons between sites and to calculate hydraulic dimensions for each site. Because no sites were gauged, bank-full dimensions and hydrological characteristics were estimated from channel geometry and site observations. Channel geometry also provided an indication of the dominant type of bank erosion occurring at each site (particularly in the case of large-scale mass failure because of cantilever failure or rotational slumping).

Various measures of stream power exist and there is some confusion among the different descriptors (Rhoads 1987). However, Brookes (1983; 1990) demonstrated a link between stream power per unit boundary area and bank stability, and this descriptor was used in this study. It was calculated for each site using the following equation:

$$\text{Stream power (Watts/m}^2\text{)} = (9800 \times \text{Discharge x Slope})/\ \text{Bank-full Width}$$

Where

$$\text{Discharge (m}^3\text{s}^{-1}\text{)} = \text{Velocity} \times \text{Cross Sectional Area of the channel}$$

Velocity was calculated using the equation:

$$\text{Velocity (ms}^{-1}\text{)} = \{(\text{Hydraulic Radius}^{0.66}) \times (\text{Slope}^{0.5})\}/\ \text{Manning's } n$$

Where

$$\text{Hydraulic Radius} = \text{Cross Sectional Area/ Wetted Perimeter}$$

Manning's *n* is a boundary roughness coefficient that accounts for the effects of edge friction and discharge. It was determined using a USDA program (USGS 2001) in conjunction with a series of pictures of reference streams with particular roughness coefficients (Barnes 1967). The slope used in these calculations was estimated by subtracting the elevation of the site from the elevation of the next upstream site and dividing this value by the stream distance between these sites.

Additional hydraulic parameters were measured during the survey of aquatic plants (MacKay *et al*., Chapter 3). They included the average water velocity and depth within quadrats positioned within riffles. These measurements were used to calculate Reynolds number and Froude number (Gordon *et al*. 2004). Water slope was measured as the change in relative height of the water surface over the entire 100 m site length, using a staff and dumpy level.

2.4.5 Bed substratum particle size and composition

At each site, large woody debris was recorded for the entire segment (reach). Riffle sediment characteristics were determined using a zigzag method (Bunte and Abt 2001), based on the technique developed by Wolman (1954) to describe coarse river bed materials. A systematic bank-to-bank course was chosen to pick up and measure the intermediate axis of 100 clasts (substratum particles) spaced regularly across the stream bed. Finer bed-particle-size distributions were determined from sediment samples collected at each site and dry-sieved in the laboratory using a minimum sieve mesh size of 4 ø (0.0625 mm), a maximum mesh size of -6 ø (64 mm), with a 1 ø interval between mesh sizes to separate coarse fragments, and finally tested by hydrometer to separate fines that passed through the 4 ø mesh. Particles larger than - 6 ø were measured manually with vernier callipers (Rowell 1994; Gordon *et al*. 2004). Sediments were classified using a modified Wentworth Scale described in Table 2.6 (Wentworth 1922; Gordon *et al*. 2004). The percentage of each size class was calculated using the sediment size analysis program, *GRADISTAT* Version 4.0, to provide simple grain size analysis of riffle samples. Grain size parameters were calculated geometrically (in microns) and logarithmically (using the phi scale) (Krumbein and Pettijohn 1938). The median particle size (d_{50}), the 10th percentile (d_{10}) and the 90th percentile (d_{90}) were also determined at each site. Measurements of substratum particle composition within 1.0 m^2 quadrats were undertaken as part of the aquatic plant survey (MacKay *et al*., Chapter 3).

2.4.6 Longitudinal survey

A reach-scale visual assessment of the Little Mulgrave River, Behana Creek and Babinda Creek was undertaken by traversing the entire stream segment across the floodplain by canoe, concurrently with water quality sampling. This was not done for Woopen Creek, which was too shallow for the canoe. During this survey the pool-run-riffle sequence was mapped, but because of the non-regular meandering nature of these streams, this could not provide any useful information as to the geomorphic health and condition (relative distances between riffles etc.) and is not reported. Other parameters recorded included the degree of undercutting of left and right banks, riparian connectivity, the presence of large woody debris, weed infestation and degree of erosion, at approximately 100-300 m intervals. These characteristics were rated using a nominal score. A 1-5 scale was used for each, with 1 indicating good condition and 5 indicating poor condition. These data were aggregated into a combined rating, which is presented in this report to illustrate trends in overall geomorphic condition.

Table 2.6. Sediment particle size classification adopted for use in GRADISTAT from Wentworth (1922) (see Gordon *et al*. 2004).

Sediment description		Grain size	
		mm	Phi (ø)
Boulder	Very large	>2048	>-11
	Large	>1024	>-10
	Medium	>512	>-9
	Small	>256	>-8
Cobble	Large	>128	>-7
	Small	>64	>-6
Gravel	Very Coarse	>32	>-5
	Coarse	>16	>-4
	Medium	>8	>-3
	Fine	>4	>-2
	Very Fine	>2	>-1
Sand	Very coarse	>1	>0
	Coarse	>0.5	>1
	Medium	>0.25	>2
	Fine	>0.125	>3
	Very fine	>0.064	>4
Silt	Very coarse	>0.032	>5
	Coarse	>0.016	>6
	Medium	>0.008	>7
	Fine	>0.004	>8
	Very fine	>0.002	>9
Clay	Clay	<0.002	<9

2.4.7 Water Quality

Water quality samples were collected at selected study sites (Table 2.4). In addition, a longitudinal series of water quality samples was collected in each stream during the survey by canoe. This allowed a large number of samples, distributed along the longitudinal stream gradient, to be taken over a short period of time in each stream. In Woopen Creek, a series of water quality samples was collected at a number of sites at accessible locations. All samples were collected in acid-washed polyethylene bottles and were stored temporarily on ice and then frozen within two hours of collection. All samples were analysed in the Australian Centre for Tropical Freshwater Research (ACTFR) analytical laboratory at James Cook University, Townsville. Field measurements at each site included temperature, pH, electrical conductivity and dissolved oxygen, measured using Hydrolab H20 multi-parameter probes and/or a hand-held YSI 556 MPS multiprobe. These parameters were also measured using Greenspan sensors during the survey of aquatic plants (MacKay *et al*., Chapter 3).

Several sites were sampled on more than one occasion, providing indication of temporal variation during the study period and associated with rainfall and increased flows that occurred midway through the survey. Sampling was intended to correspond with low flow periods but rainfall during the survey increased flow to bank-full, so some samples were collected at low flow and others during receding flood flows. This has been considered during analyses and all plots indicate the sample series of each data point. Table 2.7 provides details of sample dates that correspond to labels used on figures. Only results of field parameters and nutrients, nitrogen and phosphorus are presented in this report, as they are most directly relevant to the biological components measured as river health indicators during this study. A more complete treatment of water quality issues and dynamics is presented in the Water Quality report of the Catchment to Reef Program.

Table 2.7. Water quality sample series and dates of samples. The codes are used to indicate sampling date in figures.

Stream	*Sample series*	*Data sampled*
Little Mulgrave River	1	04/07/2005
	2	05/07/2005
Behana Creek	1	29/06/2005
	2	14/07/2005
	3	16/07/2005
Babinda Creek	1	23-27/06/2005
	2	02/07/2005
	3	08/07/2005
	4	12/07/2005
Woopen Creek	1	13/07/2005

2.5 Results

2.5.1 Land use

Catchment characteristics and land-use cover estimates (area upstream of each site in the study streams) are presented for each stream in Tables 2.8 – 2.11. The upper catchments of all streams were largely pristine, with intact dense rainforest cover. The Little Mulgrave River had the largest catchment of all the study streams and most of its catchment was upstream of land-use influences, protected within the Wet Tropics World Heritage Area. The Little Mulgrave River was unique in that it still retained a considerable length of stream upstream of private land tenure that had a relatively low slope. In the other study streams (and most streams in the wet tropics) the low-relief areas have been developed and most of the protected areas upstream of land-use influences are steep.

Table 2.8. Land-use area in the Little Mulgrave River sub-catchment. Land-use categories are defined in Table 2.5. Data source: DNRW's QLUMP land-use data from 2005.

	Study site								
	37	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**
Total area above site (km²)	65.3	67.1	73.0	73.6	99.2	101.0	104.2	106.1	107.0
CONSERV	65.3	67.1	72.7	73.2	97.1	98.3	100.9	102.2	102.7
SUGAR	-	-	0.13	0.22	1.07	1.61	2.02	2.60	2.94
OTH_CROP	-	-	0.07	0.07	0.12	0.12	0.14	0.14	0.14
PLANTAT	-	-	-	-	-	-	-	-	-
GRAZE	-	-	-	-	-	-	-	-	-
RESID	-	-	0.04	0.06	0.87	0.90	0.98	1.04	1.04
INDUST	-	-	-	-	-	-		-	-
STORAGE	-	-	-	-	-	-	-	-	-

Land use on the floodplains of Behana and Babinda Creeks was dominated by sugar cane production. Sugar production was also prominent in the catchments of the Little Mulgrave River and Woopen Creek, but the area under sugar cane was much smaller and other land uses covered similar areas within these catchments. For example, grazing and crops such as orchard fruits and bananas covered more area than sugar in the Woopen catchment. In the Little Mulgrave River, the area used for residential developments was a third of the area used for sugar cane. Considerable areas adjacent to the Little Mulgrave River have been converted from sugar cane to turf growing and it is not clear whether this has been picked up in the QLUMP data.

2.5.2 Hydrology

Figure 2.8 shows the major drainage channels in the Russell-Mulgrave catchment, with the four study streams highlighted and study sites marked. These streams are classic dendritic streams in terms of their drainage patterns (Gordon *et al*. 2004). The daily flow volumes for Babinda Creek, at gauge station 111105A, for the period from January 2000 to present, are plotted on Figure 2.9, which shows that large flow events occur several times annually and that these large flows persist only for short periods of time. This is seen more clearly in Figures 2.10a and 2.10b, which show the daily flow volumes for this gauge station over the two years prior to this study, and during the study period, respectively. Figure 2.10a also shows the flow volumes resulting from Cyclones Larry and Monica in 2006. The flow volumes associated with these two cyclones were large, but not unusual for this stream (Figure 2.9). These figures also show that although large flow events associated with cyclones occur during the wet season, they can occur very late in the season; furthermore, moderate flow events can occur through the year. For example, during this study, heavy rainfall produced small spates in all four study streams and caused minor flooding; and another flood occurred just after our field survey (Figure 2.10b). Moderate flow events occurred for several more months and in 2006 they occurred throughout the year (Figure 2.10a).

Table 2.9. Land-use area in the Behana Creek sub-catchment. See Table 2.8 for further explanation.

	Study site									
	27	**21**	**22**	**23**	**24**	**25**	**26**	**28**	**29**	**30**
Total area above site (km^2)	46.5	59.8	63.1	66.8	69.1	70.5	85.5	94.8	96.5	98.3
CONSERV	46.5	59.8	63.0	65.63	67.1	67.49	80.0	83.0	83.2	83.2
SUGAR	-	-	-	1.19	2.02	3.00	5.42	11.51	12.99	14.74
OTH_CROP	-	-	-	-	-	-	-	-	-	-
PLANTAT	-	-	-	-	-	-	-	-	-	-
GRAZE	-	-	-	-	-	-	-	-	-	-
RESID	-	-	-	-	-	-	0.12	0.19	0.26	0.26
INDUST	-	-	-	-	-	-		0.05	0.07	0.07
STORAGE	-	-	-	-	-	-	0.02	0.02	0.02	0.02

Table 2.10. Land-use area in the Babinda Creek sub-catchment. See Table 2.8 for further explanation.

	Study site												
	11	**40**	**10**	**9**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**
Total area above site (km^2)	14.9	17.0	37.3	39.3	50.5	60.1	62.7	69.2	70.8	80.5	83.9	87.9	91.8
CONSERV	14.7	17.0	37.0	38.9	49.0	56.9	57.7	61.1	61.2	66.6	67.7	68.1	69.3
SUGAR	-	-	-	-	0.57	1.97	3.60	5.23	6.73	10.70	11.66	15.57	17.87
OTH_CROP	-	-	-	0.07	0.31	0.35	0.49	0.49	0.49	0.63	0.66	0.66	0.67
PLANTAT	-	-	0.11	0.11	0.11	0.16	0.16	0.16	0.16	0.16	0.18	0.18	0.18
GRAZE	-	-	0.20	0.20	0.45	0.53	0.53	0.96	0.96	0.96	0.96	0.96	0.96
RESID	-	-	0.02	0.02	0.02	0.02	0.16	0.19	0.19	0.38	1.44	1.44	1.78
INDUST	-	-	-	-	-	-	-	-	-	0.08	0.21	0.21	0.34
STORAGE	-	-	-	-	-	-	-	-	-	-	-	-	-

Table 2.11. Land-use area in the Woopen Creek sub-catchment. See Table 2.8 for further explanation.

	Study site							
	38	**39**	**31**	**36**	**32**	**33**	**34**	**35**
Total area above site (km^2)	1.0	3.0	9.8	10.1	11.5	11.8	14.0	26.8
CONSERV	1.0	2.8	9.4	9.5	9.6	9.6	10.7	19.8
SUGAR	-	-	-	0.03	0.31	0.40	1.21	2.28
OTH_CROP	-	-	0.11	0.26	0.83	0.97	1.22	1.83
PLANTAT	-	-	-	-	-	-	-	-
GRAZE	-	0.24	0.30	0.30	0.72	0.75	0.90	2.96
RESID	-	-	-	-	-	-	-	-
INDUST	-	-	-	-	-	-		0.12
STORAGE	-	-	-	-	-	-	-	-

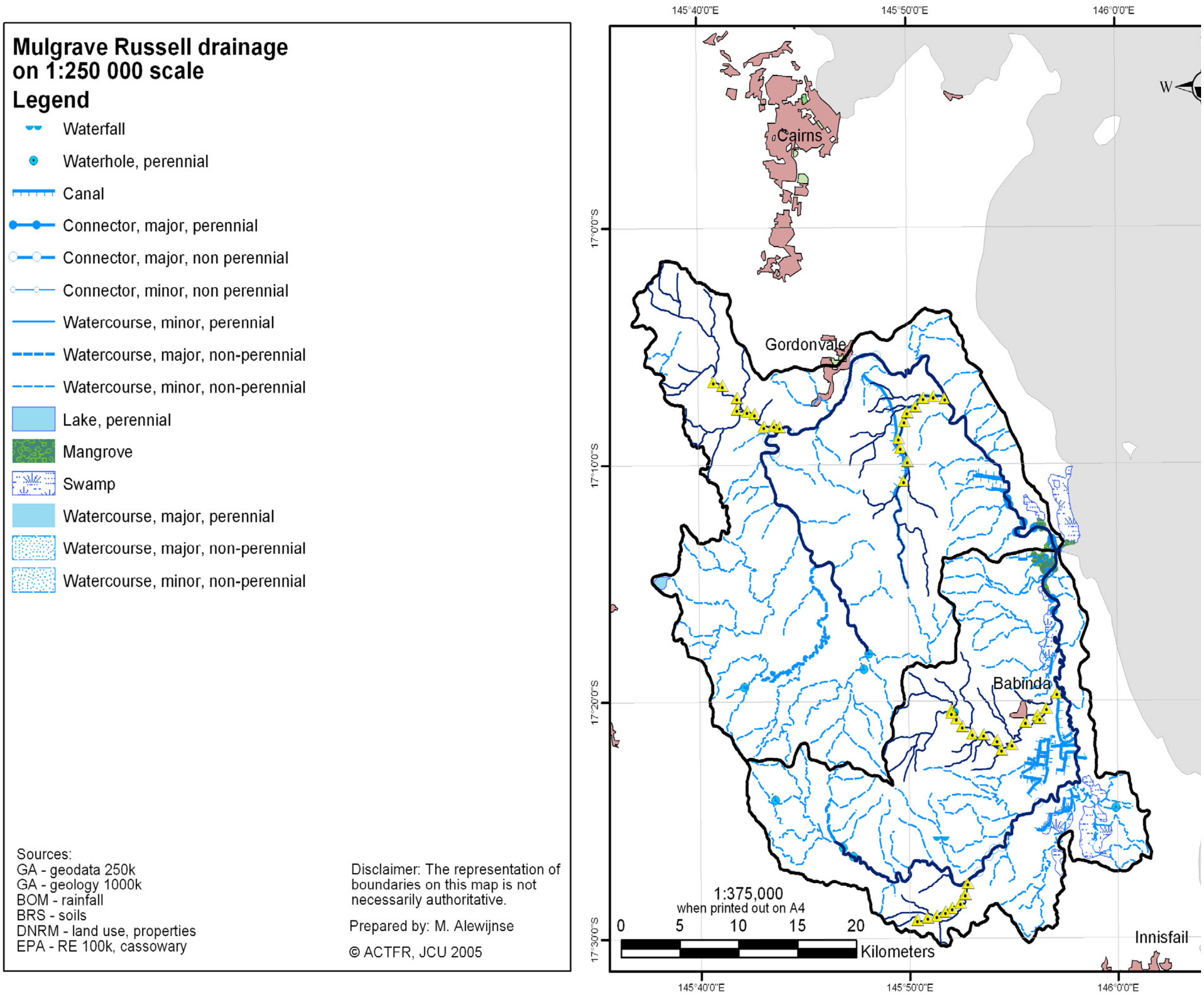


Figure 2.8. Russell-Mulgrave catchment drainage

Flow data for the sampling period were not available for the other streams. However, flow records for a gauge that used to operate on Behana Creek at Aloomba (111003C) confirm that Behana Creek also experiences frequent high flows with flow levels increasing and decreasing rapidly (Figure 2.11). Comparison of daily rainfall at Babinda Post Office, Deeral and Gordonvale shows that there was a gradient in rainfall intensity from Babinda northward (Figure 2.12). However, after heavy rainfall during the survey, flows in all four study streams were observed to approach bank-full. Rainfall was greatest in the upper catchments, especially associated with Mounts Bartle Frere and Bellenden Ker, so, although less rainfall may have fallen on the floodplains in the northern sections of the study area, significant quantities are likely to have fallen on the upper catchments contributing to the high flows observed in all streams. Figure 2.13 contrasts the precipitation at the top and bottom of Mount Bellenden Ker.

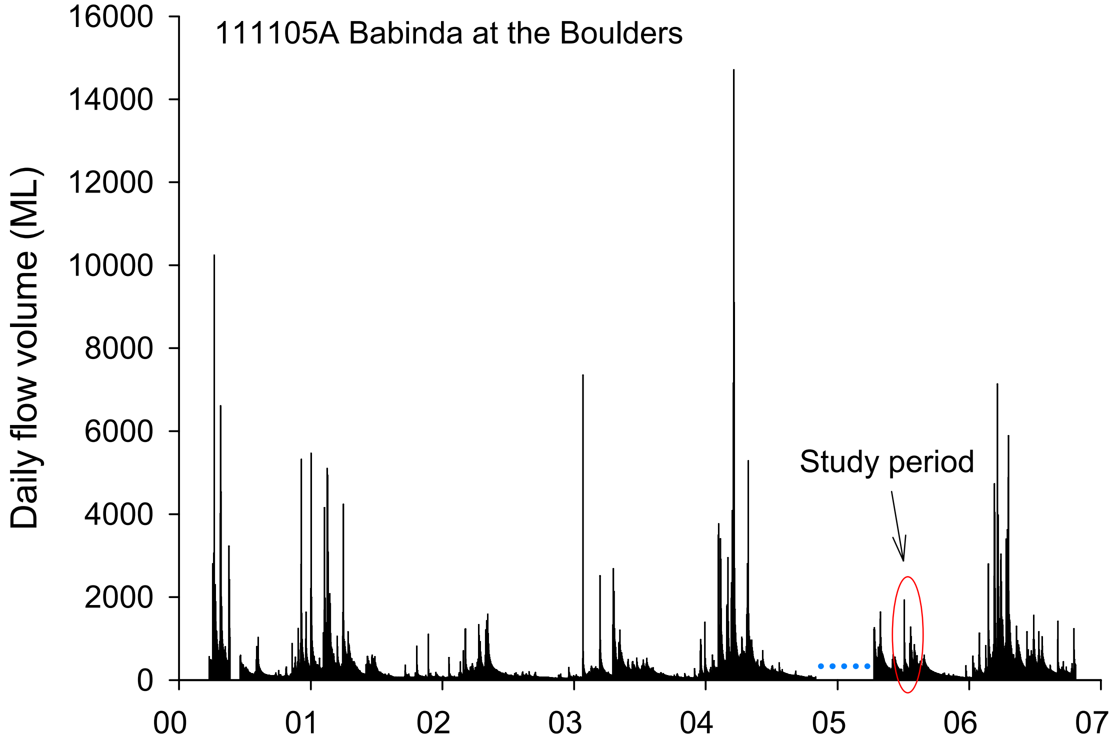


Figure 2.9. Daily flow volumes (2000 – 2006) at gauge station number 111105A in Babinda Creek, located at the Boulders (study site 10). Dotted line indicates a period of missing data.

2.5.3 Geomorphology

All four study streams had strong longitudinal gradients in geomorphic features. This was most striking for stream width and sediment particle sizes (Figures 2.14 – 2.21). In the upper reaches, bed materials consisted of coarse cobbles and boulders. In the lower reaches, the bed materials generally became finer, reducing to gravels and sands towards the mouths of the longer streams, Babinda and Behana Creeks. An interesting feature of Behana and Babinda Creeks is that they narrowed downstream rather than widened, in contrast to the classic model of streams widening with distance downstream. Figure 2.14a and 2.16a show how the bank-full width of Behana and Babinda Creeks narrowed after initially widening at the base of the range. The narrowing of these streams did not appear to be the result of excessive incision of the downstream sections, but is more likely explained because these streams over-top their banks very close to the range during floods and the full stream discharge is not contained in the channel during high flows. In our study streams, stream power per unit area reduced within the channel with distance downstream owing to the decrease in slope (Figure 2.14b,c and 2.16b,c) and, thus, the potential for geomorphic work in the channel reduced downstream. Reduction in stream slope also reduced velocities within the channel, resulting in the gradient in sediment particle size distribution downstream, as larger particles gradually deposited with slowing velocity (Figure 2.14d and 2.16d).

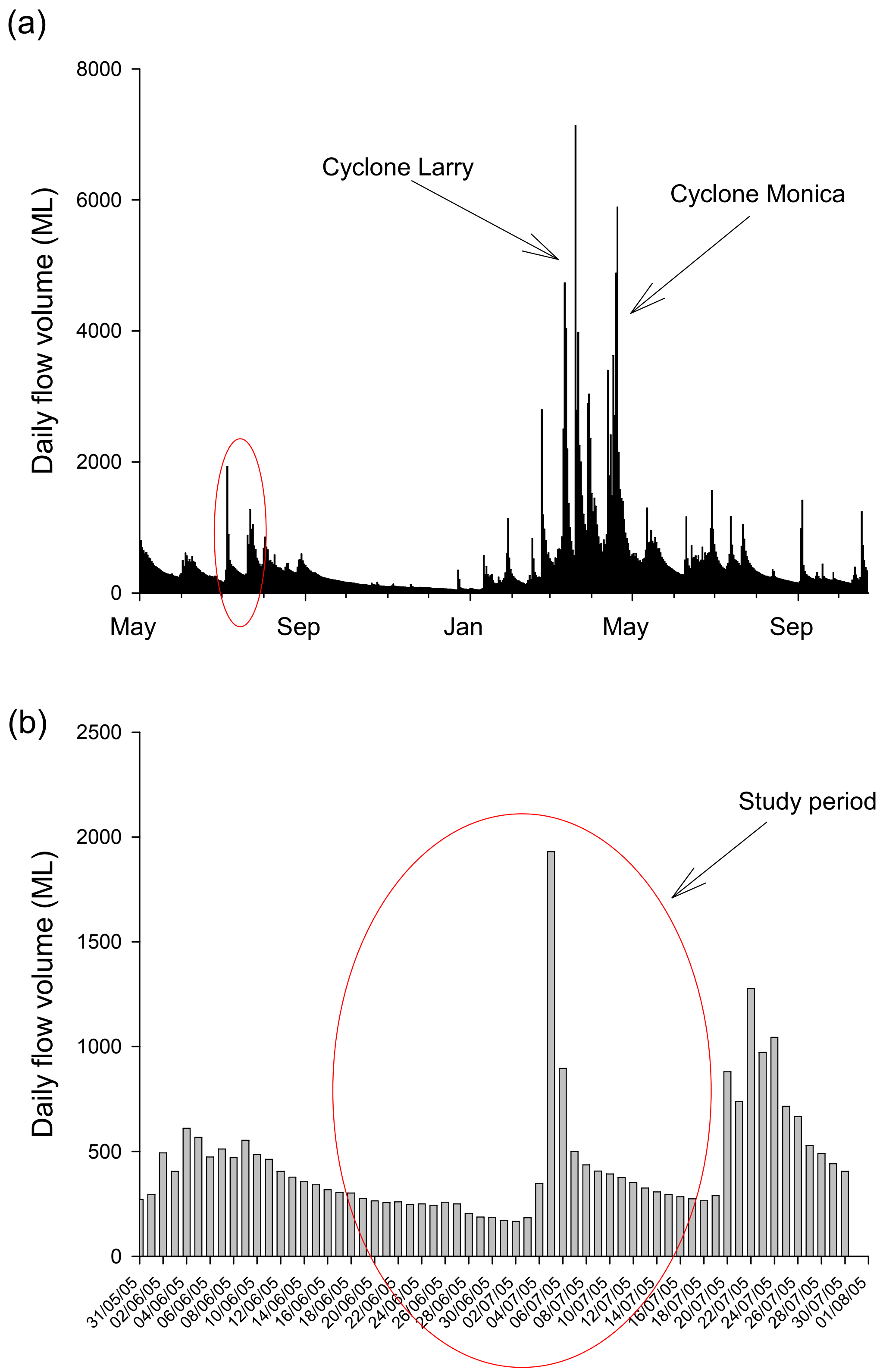


Figure 2.10. Daily flow volumes for Babinda Creek at gauge station 111105A during the study period.

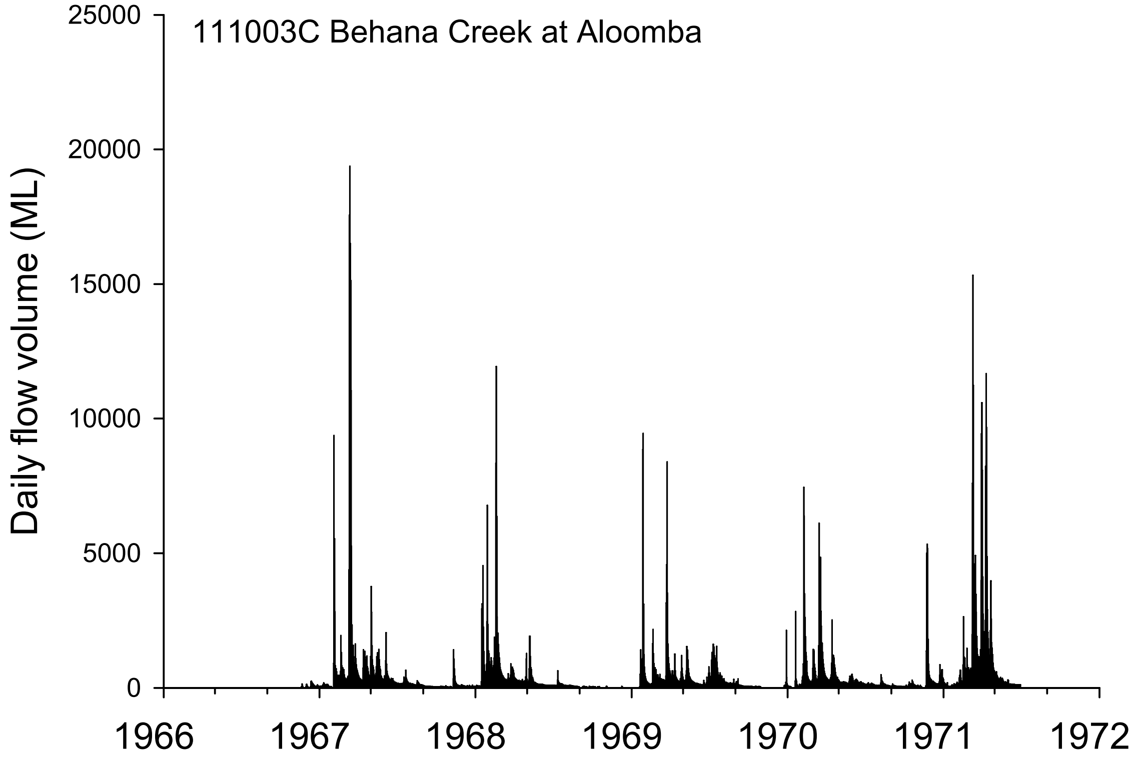


Figure 2.11. Daily flow volumes for Behana Creek gauge 111003C, 1996-1971.

Woopen Creek did widen with distance downstream, possibly because it had a much smaller and lower-elevation upper catchment and so produced relatively smaller flood flows that could be contained within the channel. Woopen Creek also had a more gradual change in slope as it entered the floodplain (Figure 2.20b), in contrast to the abrupt change in Behana Creek (Figure 2.14b) and, particularly, Babinda Creek (Figure 2.16b).

The Little Mulgrave River narrowed, widened and then narrowed again (Figure 2.18a) because the channel was laterally constrained by the valley sides in some sections, resulting in increased velocities and bed incision. Bed slope steepened locally in these sections as a result of localised bed lowering (knick points) (Figure 2.20b). The cross sectional profiles in Figure 2.21 illustrate the narrowing of the channels, and bench formation within the bank-full confines that indicate past or present incision.

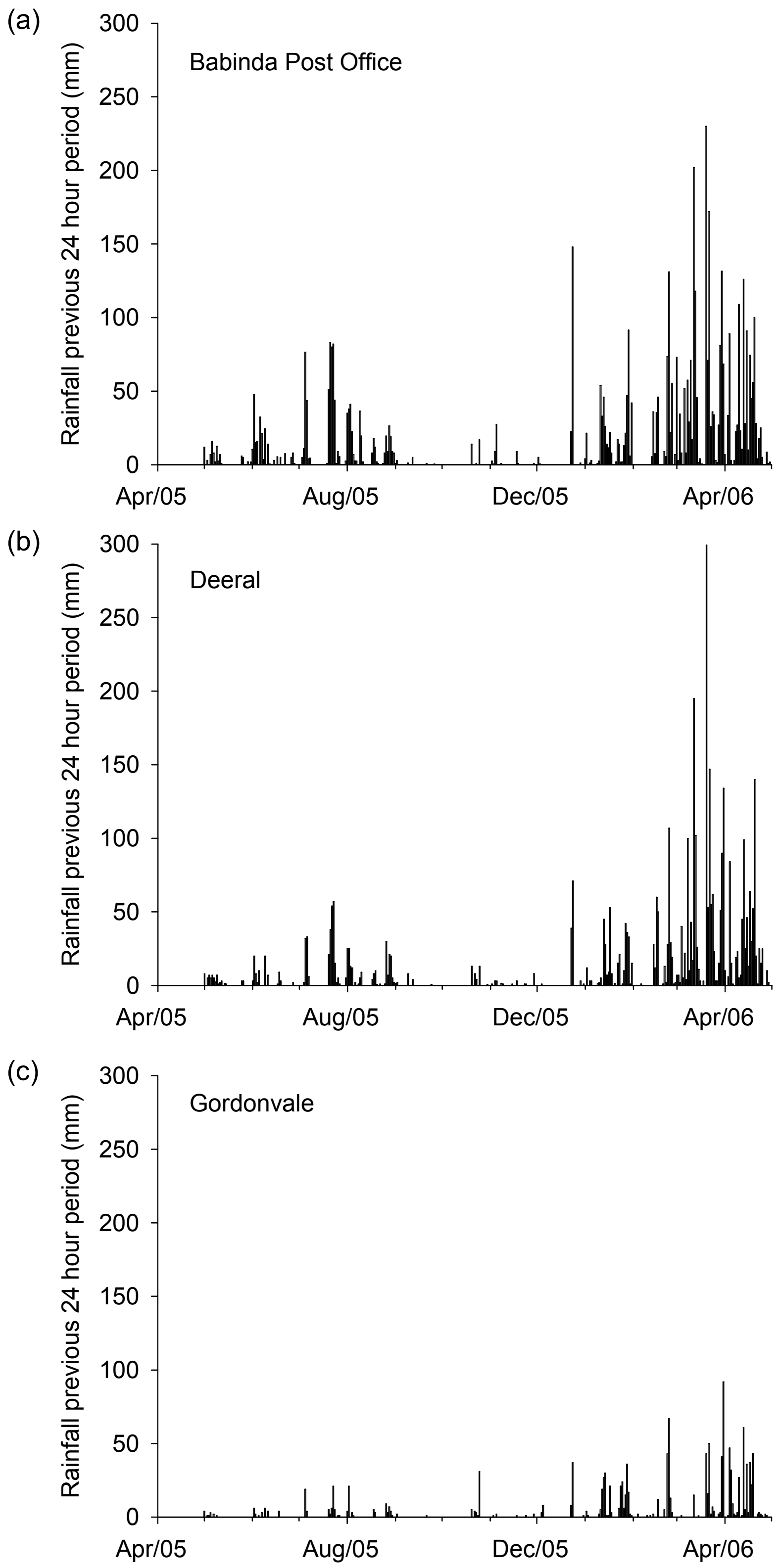


Figure 2.12. Daily rainfall recorded at (a) Babinda Post Office, (b) Deeral and (c) Gordonvale during the study period.

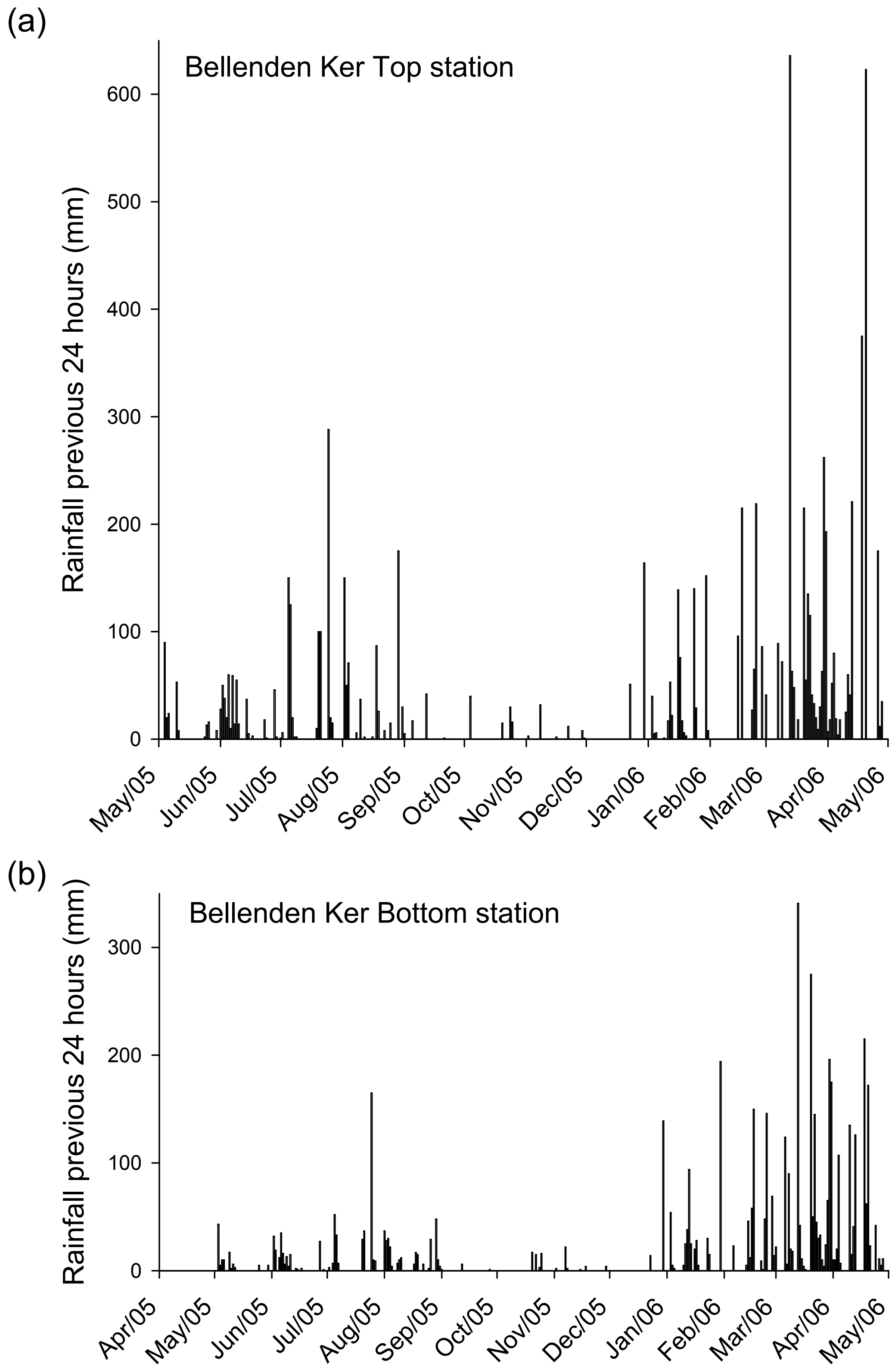


Figure 2.13. Daily rainfall recorded at (a) Bellenden Ker Top station and (b) Bellenden Ker Bottom station, from August 2005 to May 2006.

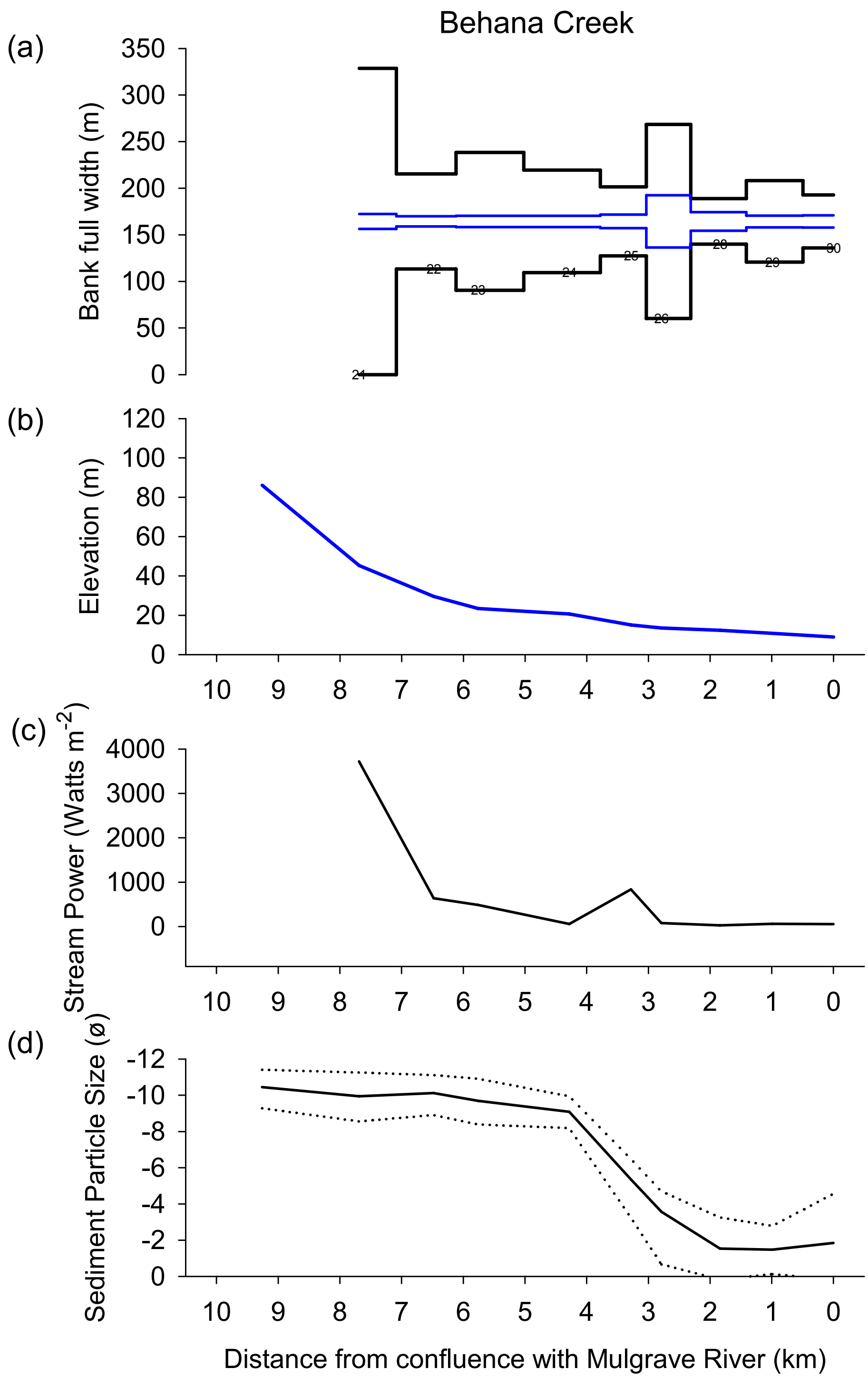


Figure 2.14. Longitudinal patterns of geomorphic and hydraulic attributes of Behana Creek: (a) bank-full width, (b) elevation, (c) stream power and (d) median (d_{50}), d_{10} and d_{90} sediment particle sizes.

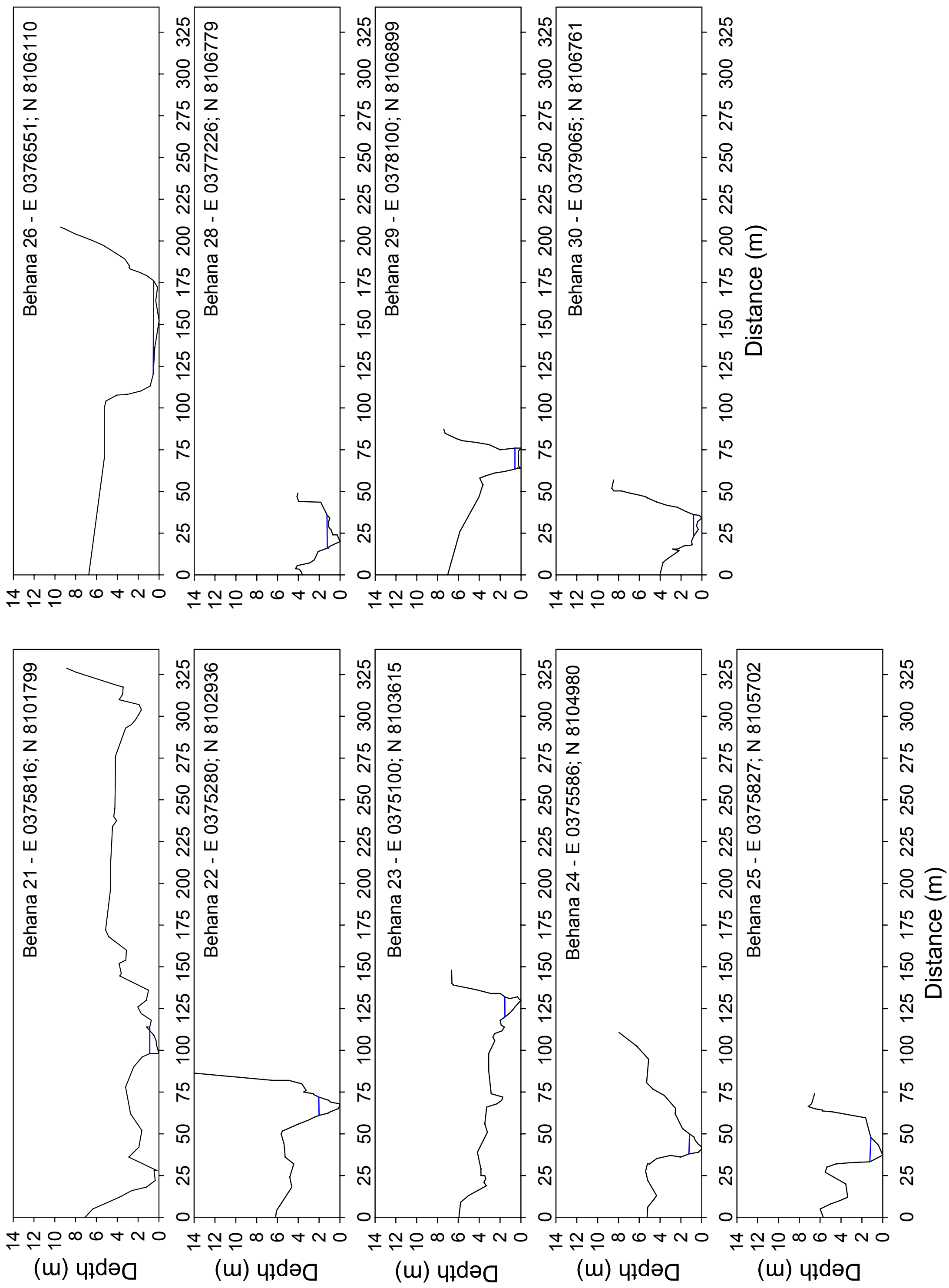


Figure 2.15. Cross-sectional profiles of study sites in Behana Creek.

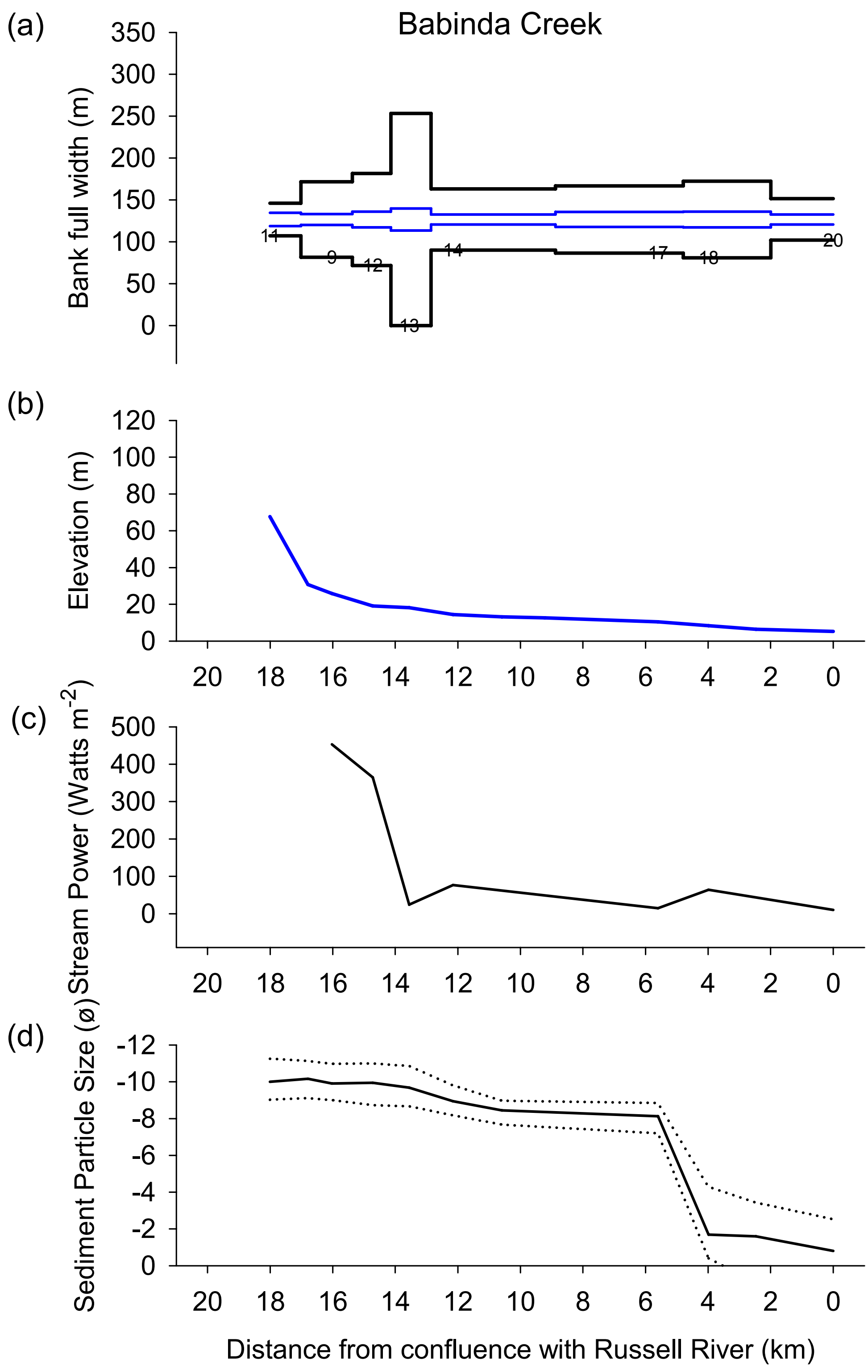


Figure 2.16. Longitudinal patterns of geomorphic and hydraulic attributes of Babinda Creek: (a) bank-full width, (b) elevation, (c) stream power and (d) median (d50), d10 and d90 sediment particle sizes.

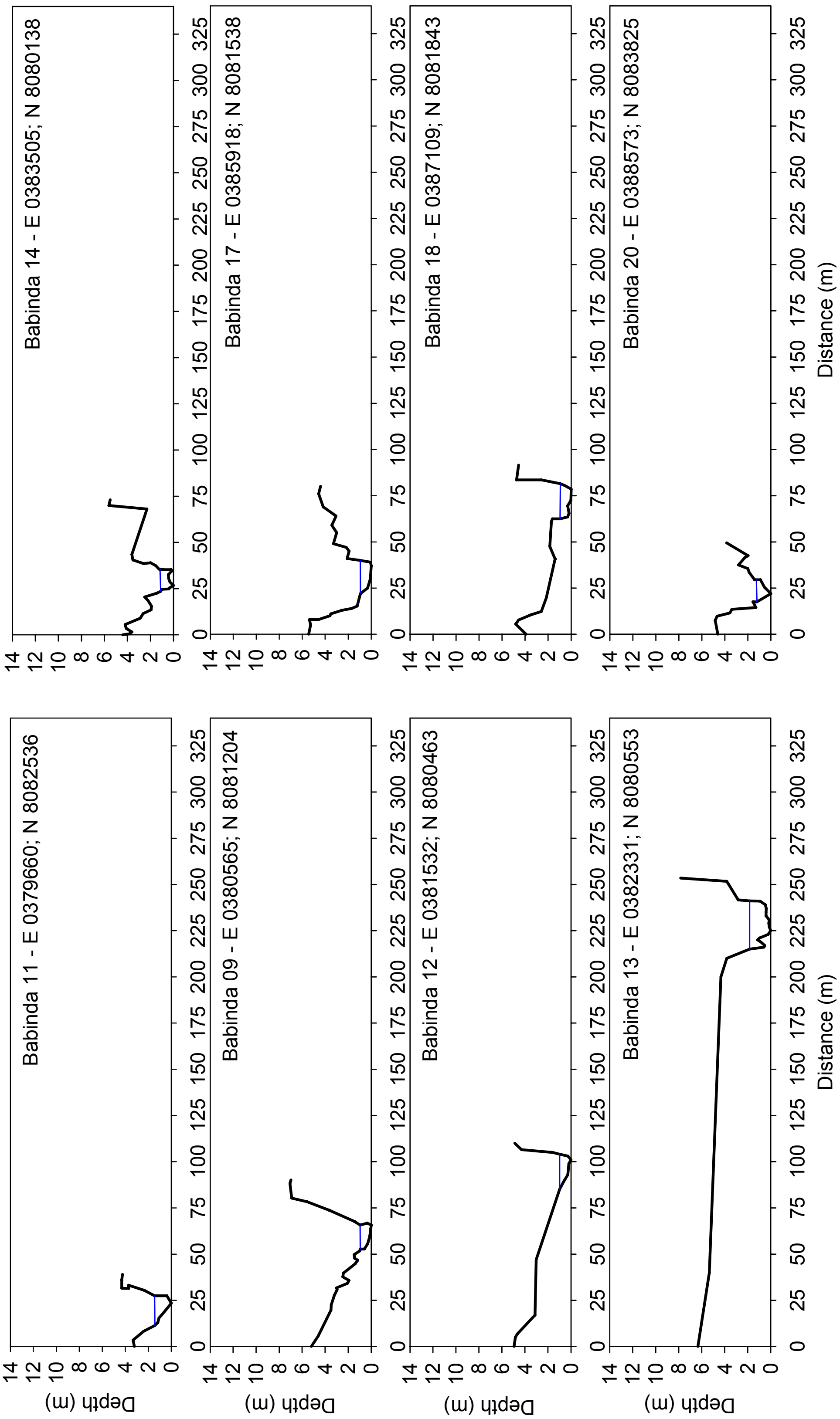


Figure 2.17. Cross-sectional profiles of study sites in Babinda Creek

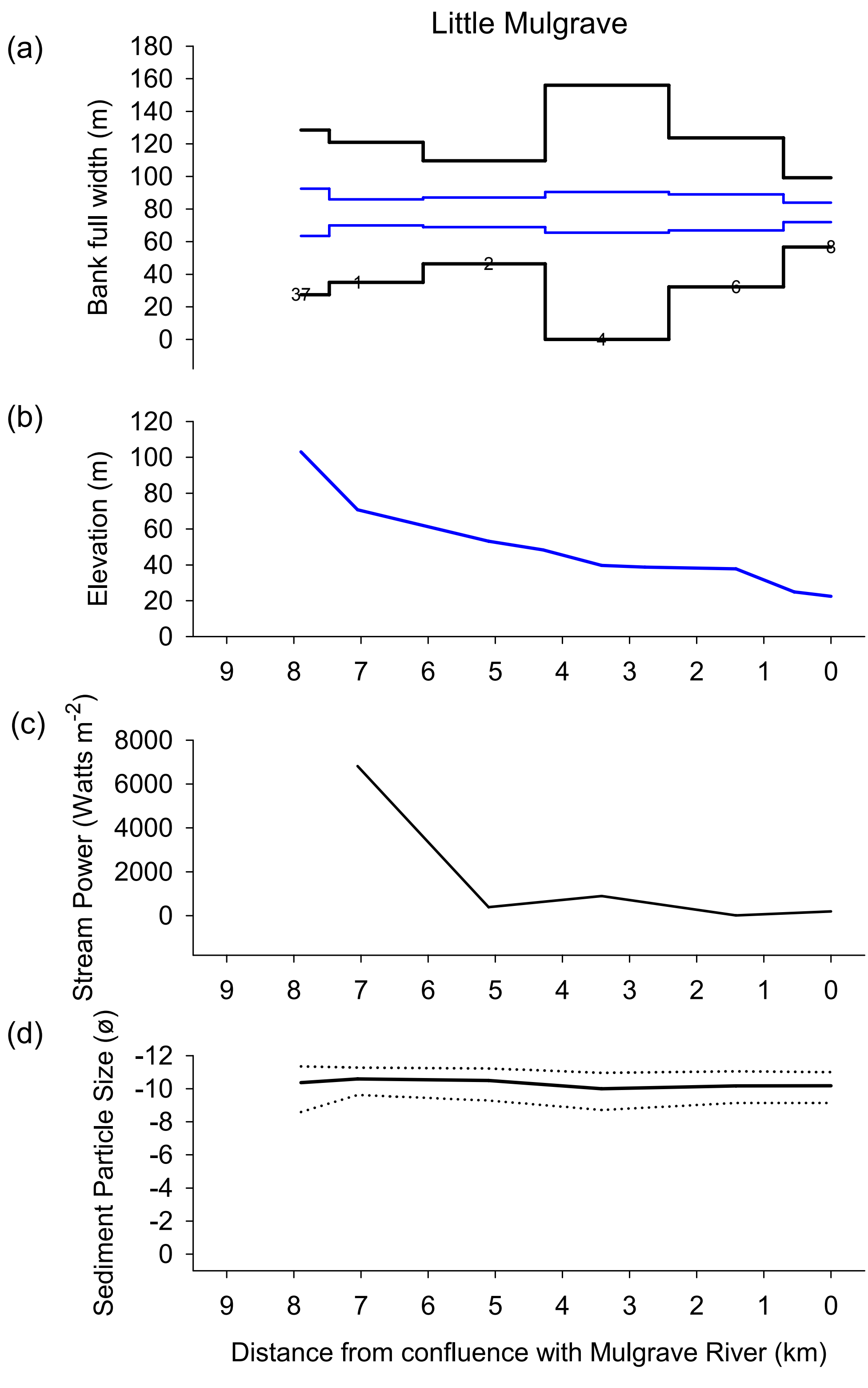


Figure 2.18. Longitudinal patterns of geomorphic and hydraulic attributes of the Little Mulgrave River: (a) bank-full width, (b) elevation, (c) stream power and (d) median (d50), d10 and d90 sediment particle sizes.

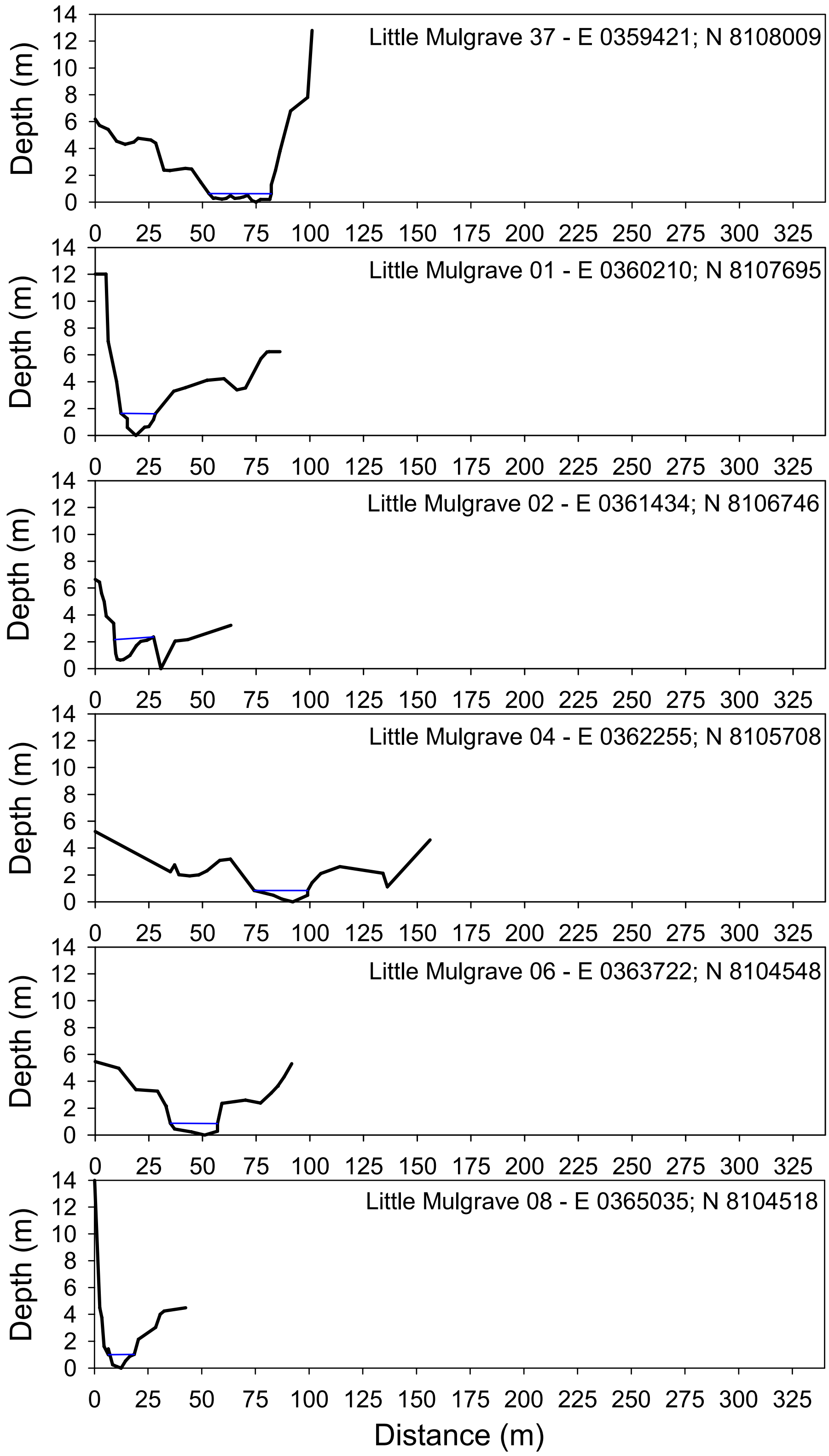


Figure 2.19. Cross-sectional profiles of study sites in the Little Mulgrave River.

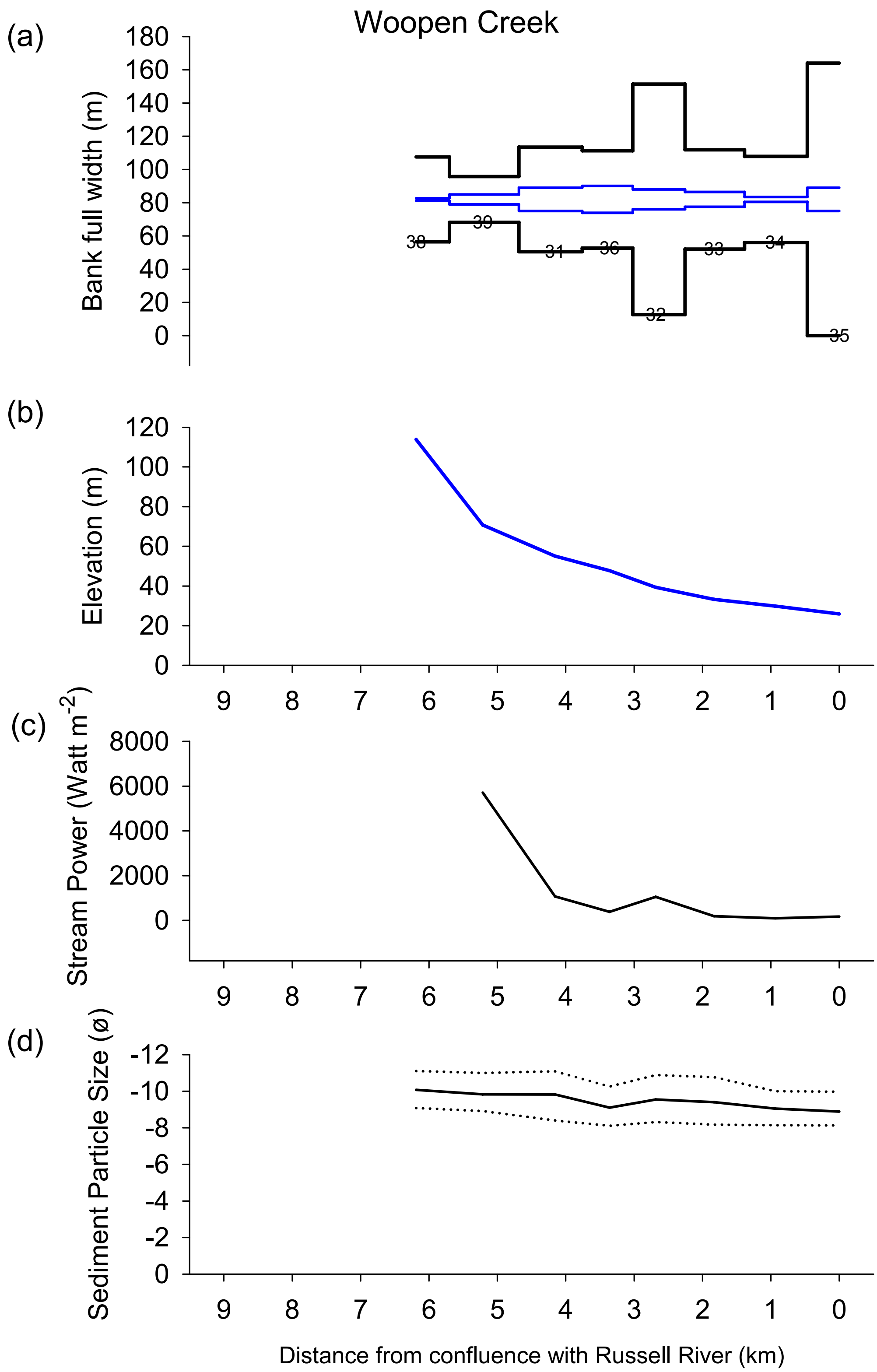


Figure 2.20. Longitudinal patterns of geomorphic and hydraulic attributes of Woopen Creek: (a) bank-full width, (b) elevation, (c) stream power and (d) median (d50), d10 and d90 sediment particle sizes.

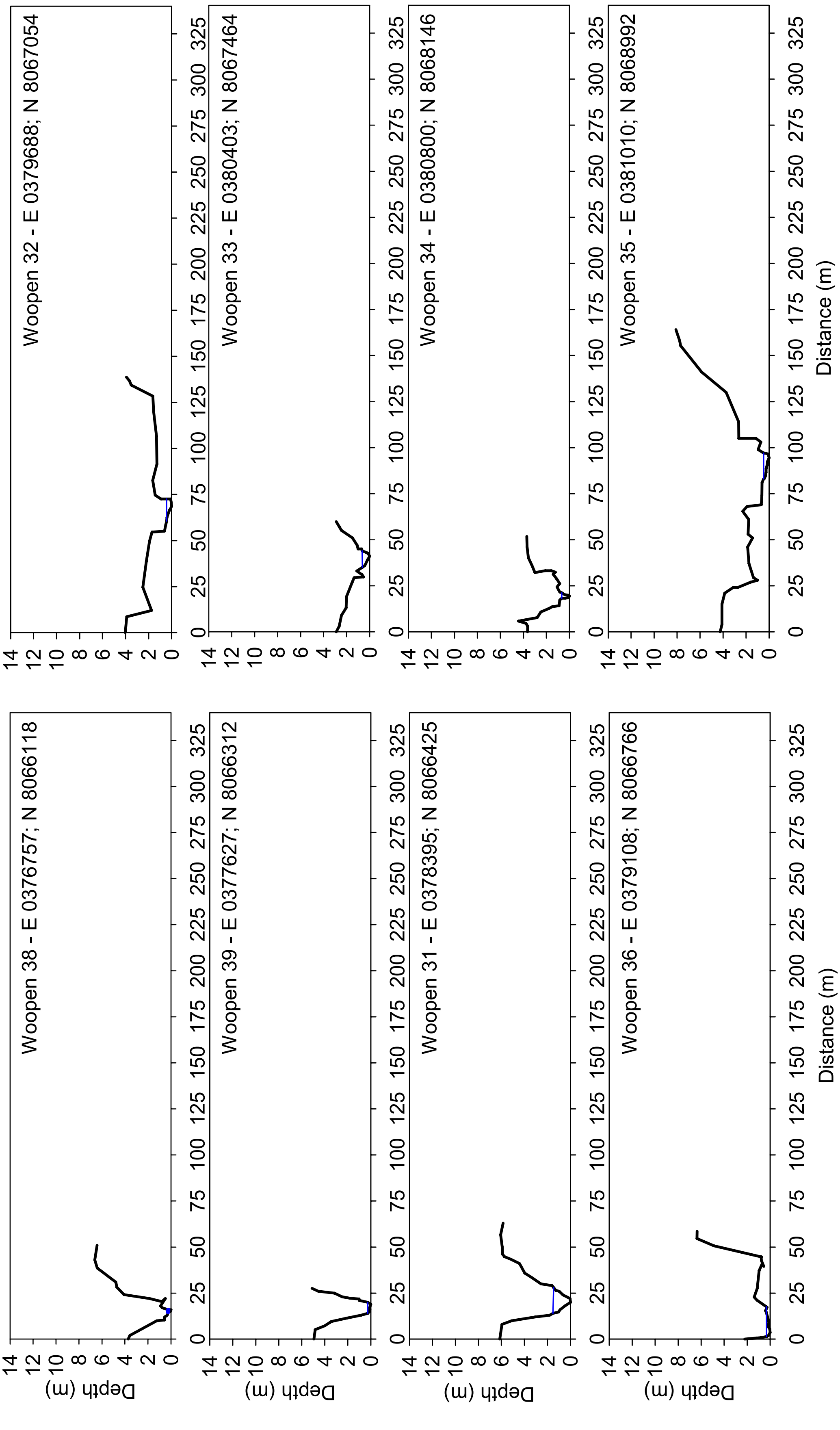


Figure 2.21. Cross-sectional profiles of study sites in Woopen Creek

The overall geomorphic condition also decreased downstream in all streams, with Woopen and Babinda Creeks being in worse condition than the Little Mulgrave River and Behana Creek (Figure 2.22). Although deep incision was not apparent, there was a high degree of channelisation in many sections of Babinda Creek and to a lesser extent in Woopen Creek, where riparian vegetation consisted almost entirely of Singapore daisy, *Sphagneticola trilobata*, or para grass, *Urochloa mutica*. The wetted channel of these sections was relatively narrow, geomorphic complexity was low and water velocity was relatively high. In Woopen Creek, elevated flows resulting from channelisation did not appear to cause incision, possibly because the flows were still not great enough to remove the cobble and coarse gravel substratum. There was also no incising of the channel in the upper sections of Babinda Creek. However, in the lower reaches, where sediment particles consisted of finer gravels and sands, there was some stream incision as a result of channelisation. In these sections, channelisation had increased velocities sufficient to transport these sediments and the channel had deepened (Figure 2.23). Water depth was generally uniform and much deeper than comparable reaches in Behana Creek, and water velocities were higher (Figure 2.24).

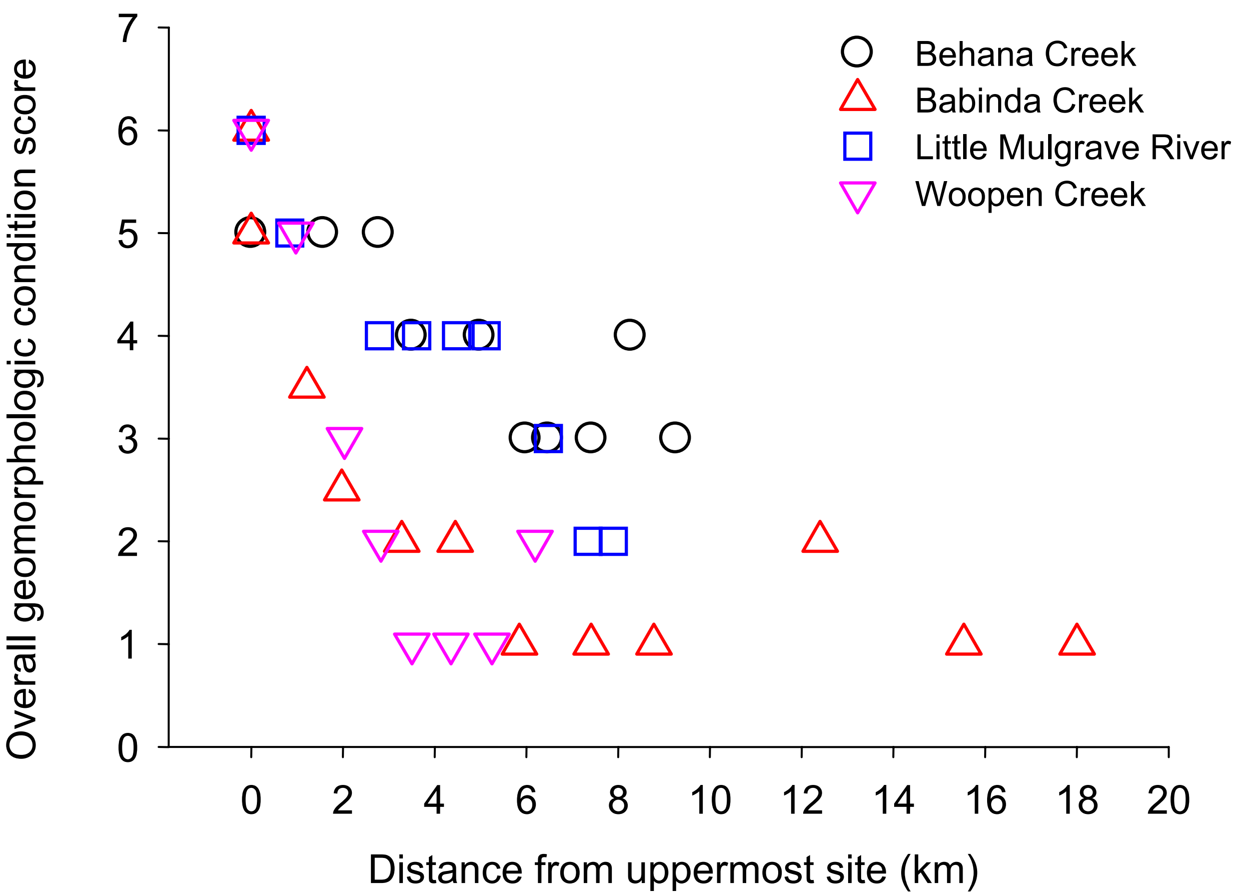


Figure 2.22. Relationship between overall geomorphic condition (scale 0-7 = poor to good) and distance from the uppermost site for the four study streams.

Figure 2.23 View of lower reach of Babinda Creek. Site 19.

Figure 2.24 View of lower reach of Behana Creek. Site 29.

2.5.4 Water Quality

The four streams were all perennial and had good flows originating from pristine forested upper catchments. As a consequence, water quality in all study streams was generally good during base flows. The range of concentrations of nutrients was considerably lower than that reported by Bramley and Roth (2002) in streams on the Herbert River floodplain. However, NO_x (nitrate + nitrite) and total phosphorus concentrations exceeded the 2006 Queensland Water Quality Guidelines (QWQG) (QEPA 2006) for wet tropics lowland streams in all study streams.

Water quality parameters for each stream are described in Figures 2.25 to 2.28. Dissolved oxygen concentrations in all streams were close to saturation as a result of the high flow and turbulence associated with riffles. Water temperature increased with distance downstream as the streams became more open and flows slowed. Water temperature ranged between 17.2 ˚C in the upper reaches of Babinda Creek to 20.9 ˚C in the lower reaches of Behana Creek. pH was generally lower in Behana and Babinda Creeks than in the Little Mulgrave River and Woopen Creek. pH tended to decline with distance downstream in Babinda Creek. Conductivity increased with distance downstream, but the variance between streams was greater, with conductivity highest in the Little Mulgrave River and Woopen Creek. Behana Creek had the lowest conductivity.

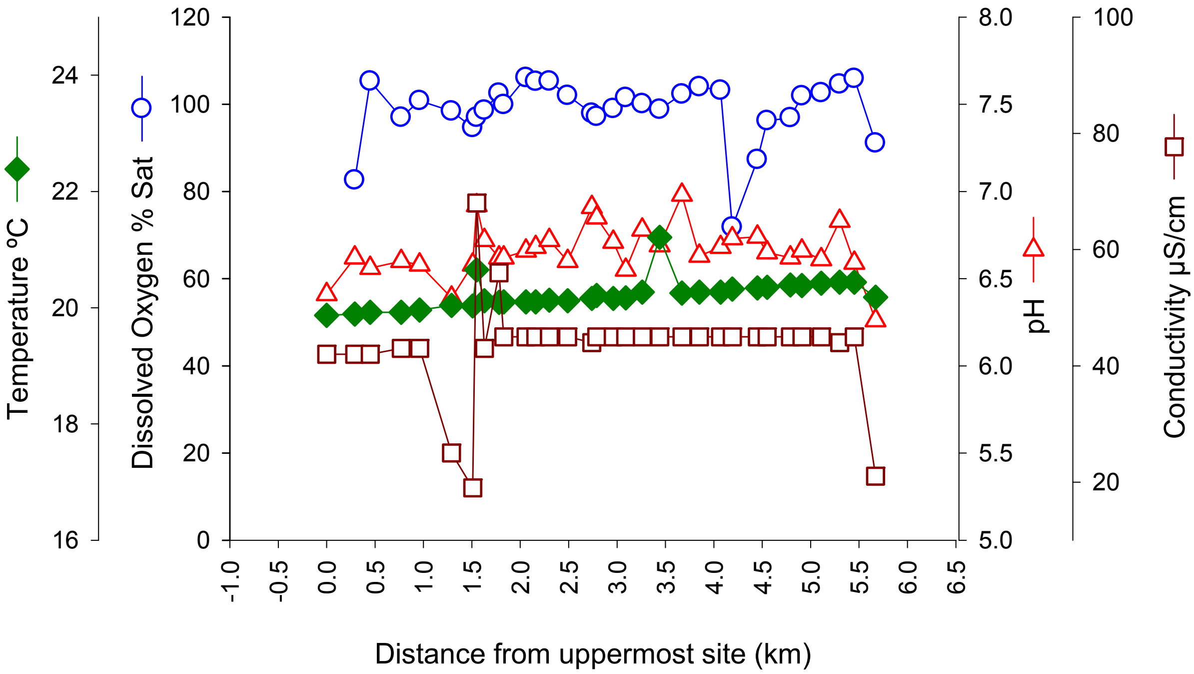


Figure 2.25. Physico-chemical variables (temperature, dissolved oxygen, pH and conductivity) measured during canoe traverses for the Little Mulgrave River.

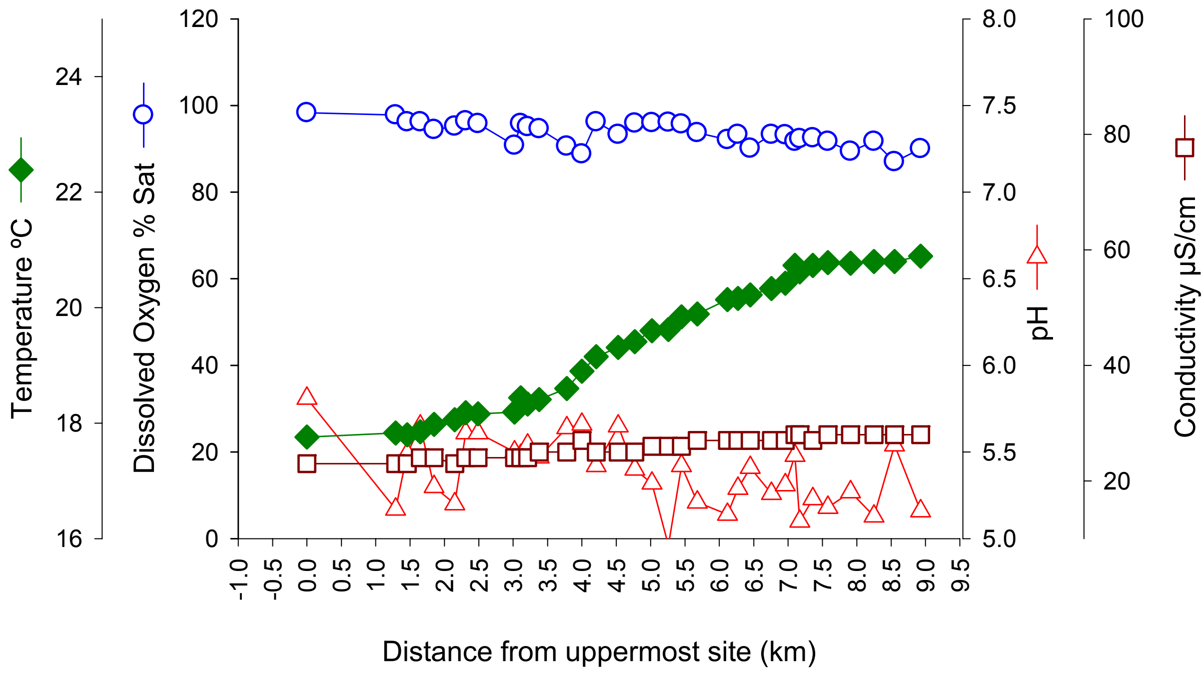


Figure 2.26. Physico-chemical variables (temperature, dissolved oxygen, pH and conductivity) measured during canoe traverses for Behana Creek.

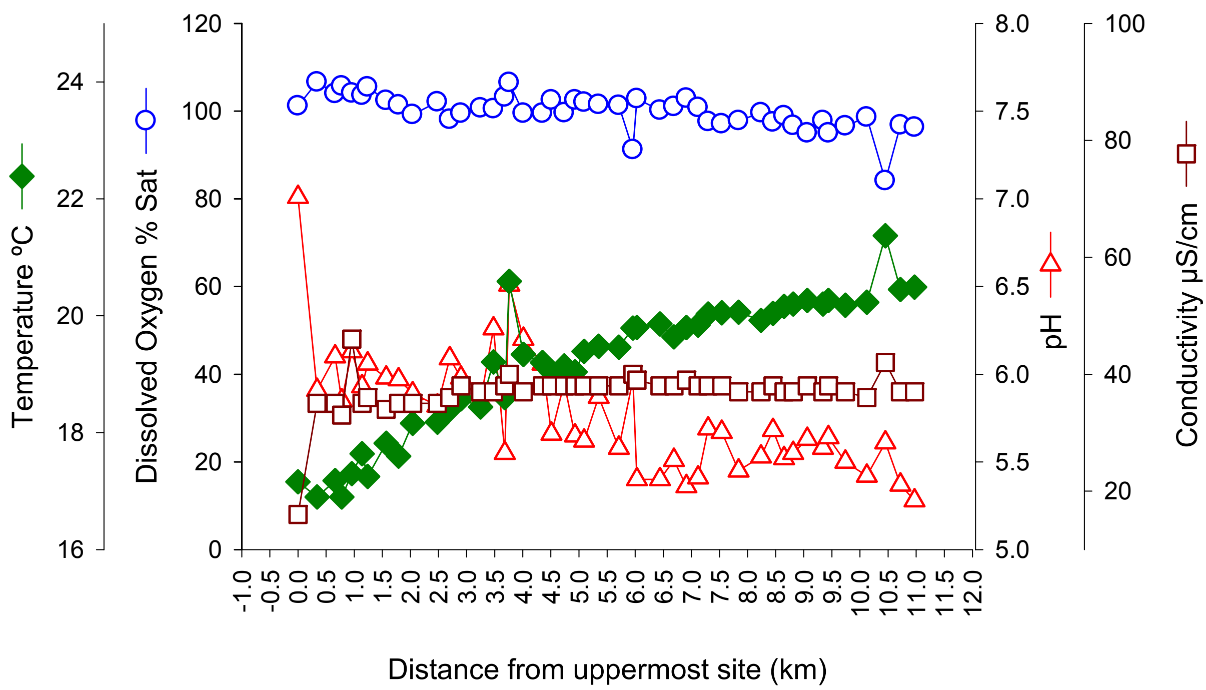


Figure 2.27. Physico-chemical variables (temperature, dissolved oxygen, pH and conductivity) measured during canoe traverses for Babinda Creek.

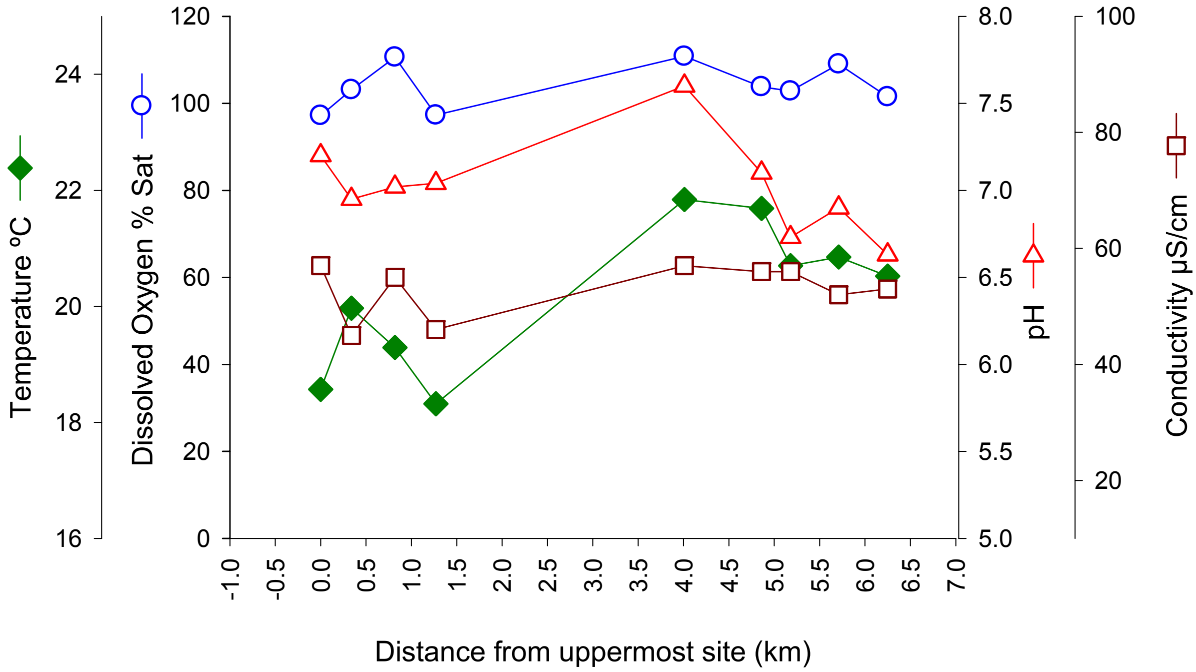


Figure 2.28. Physico-chemical variables (temperature, dissolved oxygen, pH and conductivity) measured at sample sites in Woopen Creek.

2.5.5 Nitrogen

Nitrogen concentrations were elevated in all study streams during the time water samples were collected, with total nitrogen concentration increasing with distance downstream (Figures 2.29 to 2.32). There was generally a four-fold increase in total nitrogen concentration from the most upstream site to the most downstream site in all of the study streams.

The most consistent patterns were observed in the filterable species of nitrogen, particularly nitrate and nitrite. Figure 2.33 contrasts the Nitrate + Nitrite (NO_x) concentrations in the four study streams. In Behana and Babinda Creeks NO_x concentrations increased rapidly with distance downstream (Figure 2.33a,b). NO_x concentrations exceeded QWQG 2006 after only a few kilometres from the national park boundary. However, in Babinda Creek concentrations declined at the very downstream sites. These sites are all downstream of the Babinda township and sugar mill. Mats of blue-green algae were observed on the benthic substratum and excessive growth of filamentous algae was attached to para-grass stands and other substrata at these downstream sites. Algal growth of these types was not observed at any other sites in the study area. The decline in NO_x concentration may therefore be due to absorption by this algal growth. Figure 2.34 shows that there was a very consistent trend in NO_x concentrations up to the sites located below the sugar mill intake, even when samples were taken on different dates.

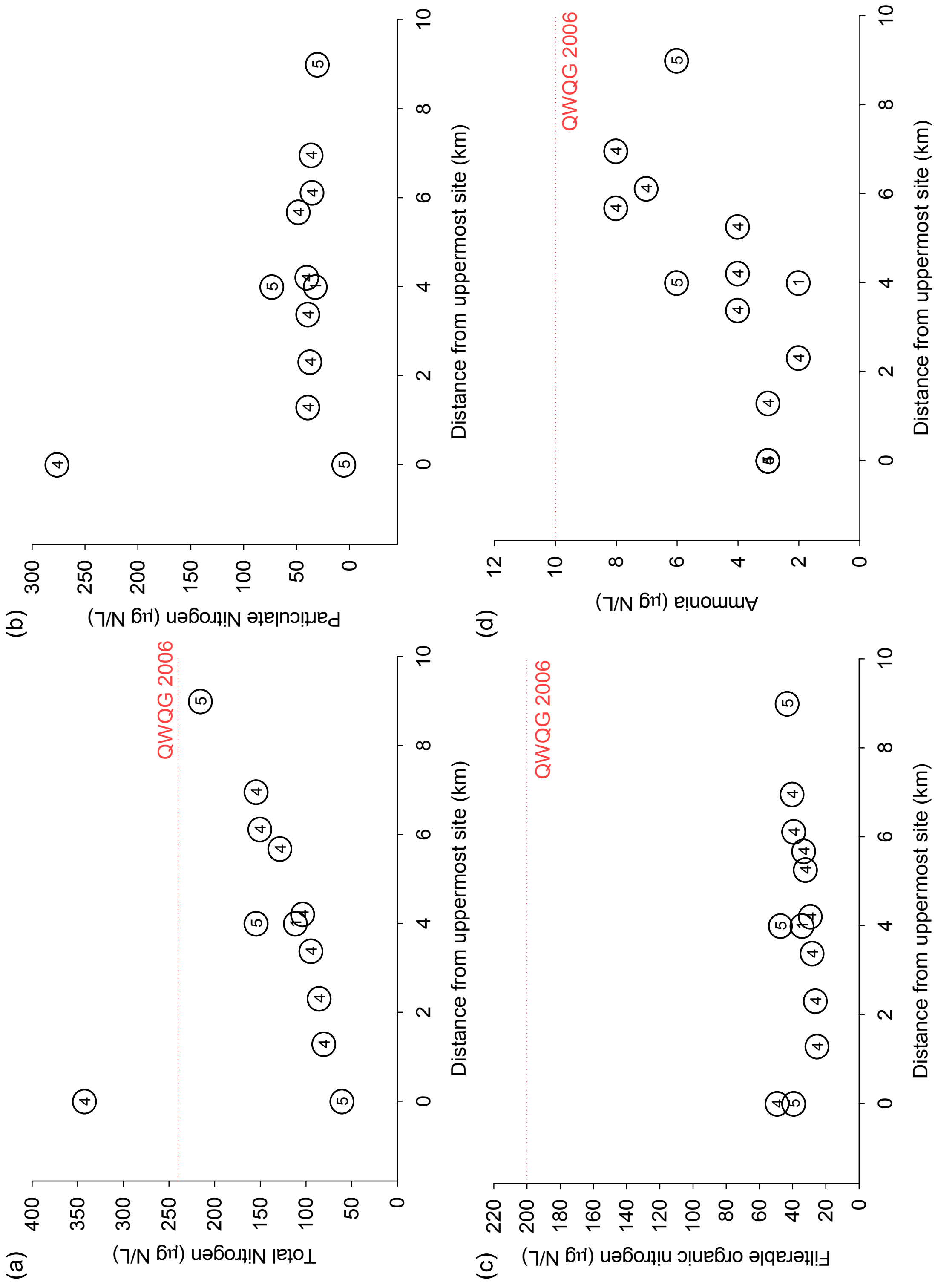


Figure 2.29. Nitrogen values measured in Behana Creek: (a) total nitrogen, (b) particulate nitrogen, (c) filterable organic nitrogen, (d) ammonia. Numbers indicate sample series described in Table 2.7. The horizontal lines represent the Queensland Water Quality Guideline (2006) concentrations.

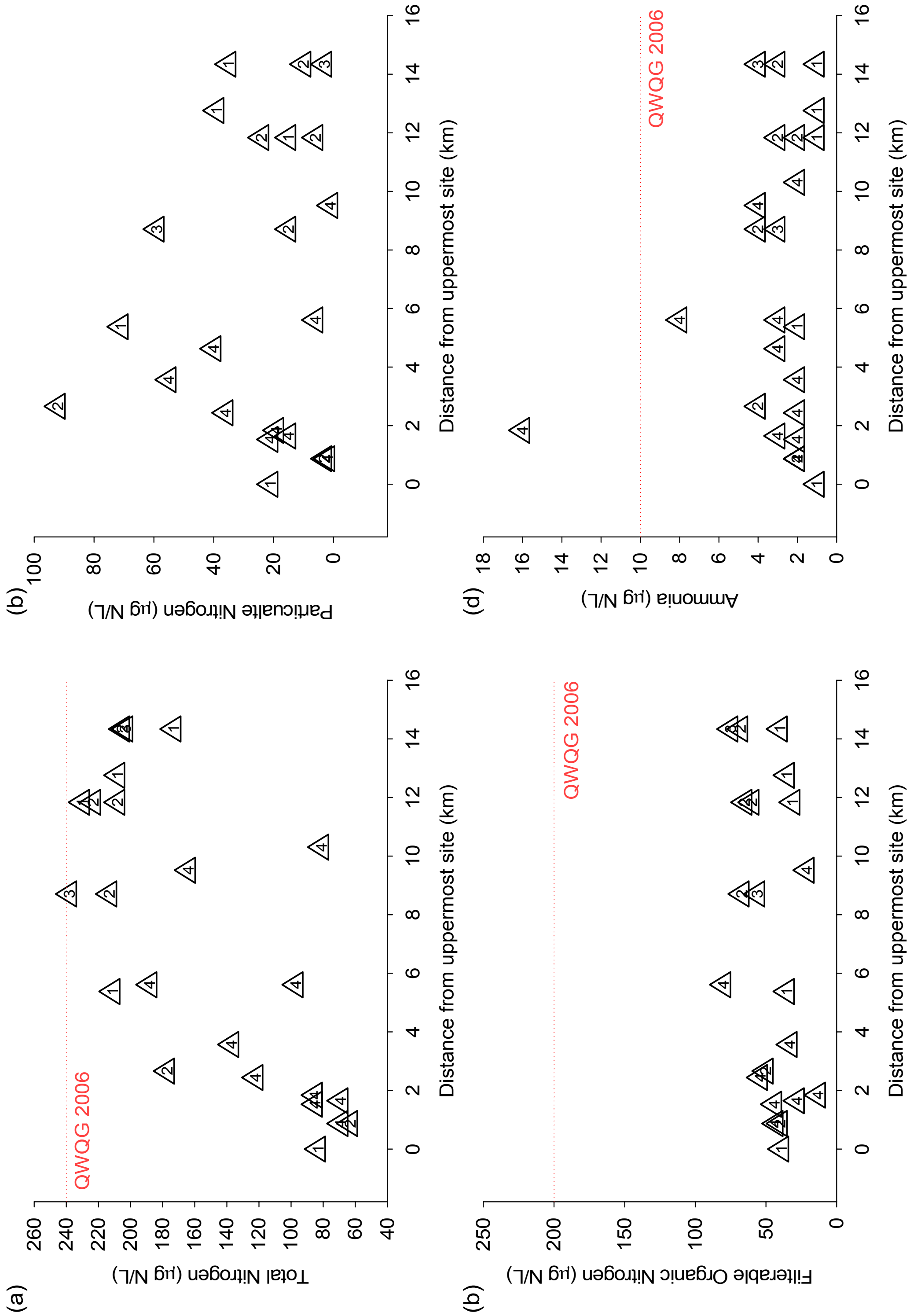


Figure 2.30. Nitrogen values measured in Babinda Creek: (a) total nitrogen, (b) particulate nitrogen, (c) filterable organic nitrogen, (d) ammonia. Numbers indicate sample series described in Table 2.7. The horizontal lines represent the Queensland Water Quality Guideline (2006) concentrations.

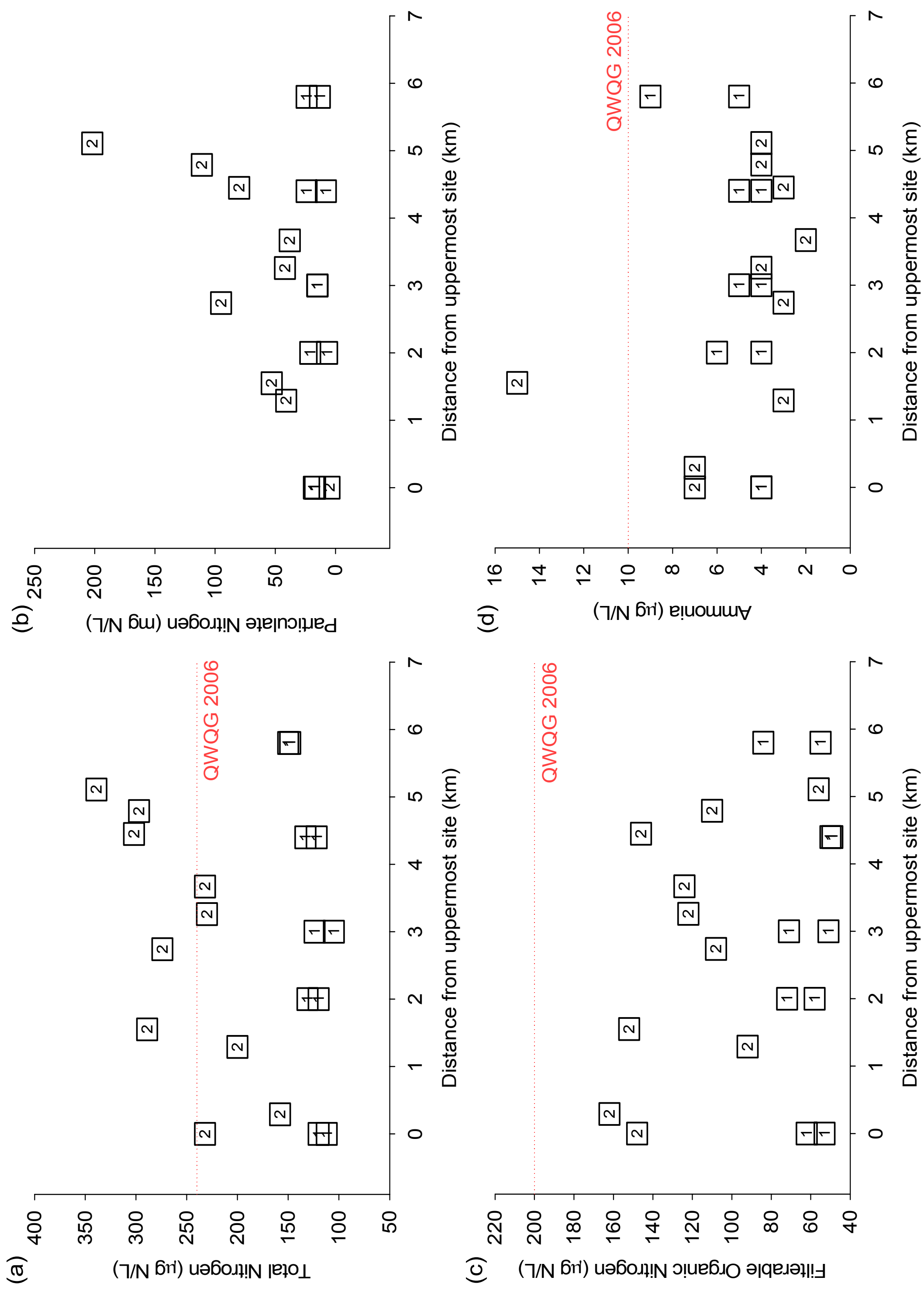


Figure 2.31. Nitrogen values measured in the Little Mulgrave River: (a) total nitrogen, (b) particulate nitrogen, (c) filterable organic nitrogen, (d) ammonia. Numbers indicate sample series described in Table 2.7. The horizontal lines represent the Queensland Water Quality Guideline (2006) concentrations.

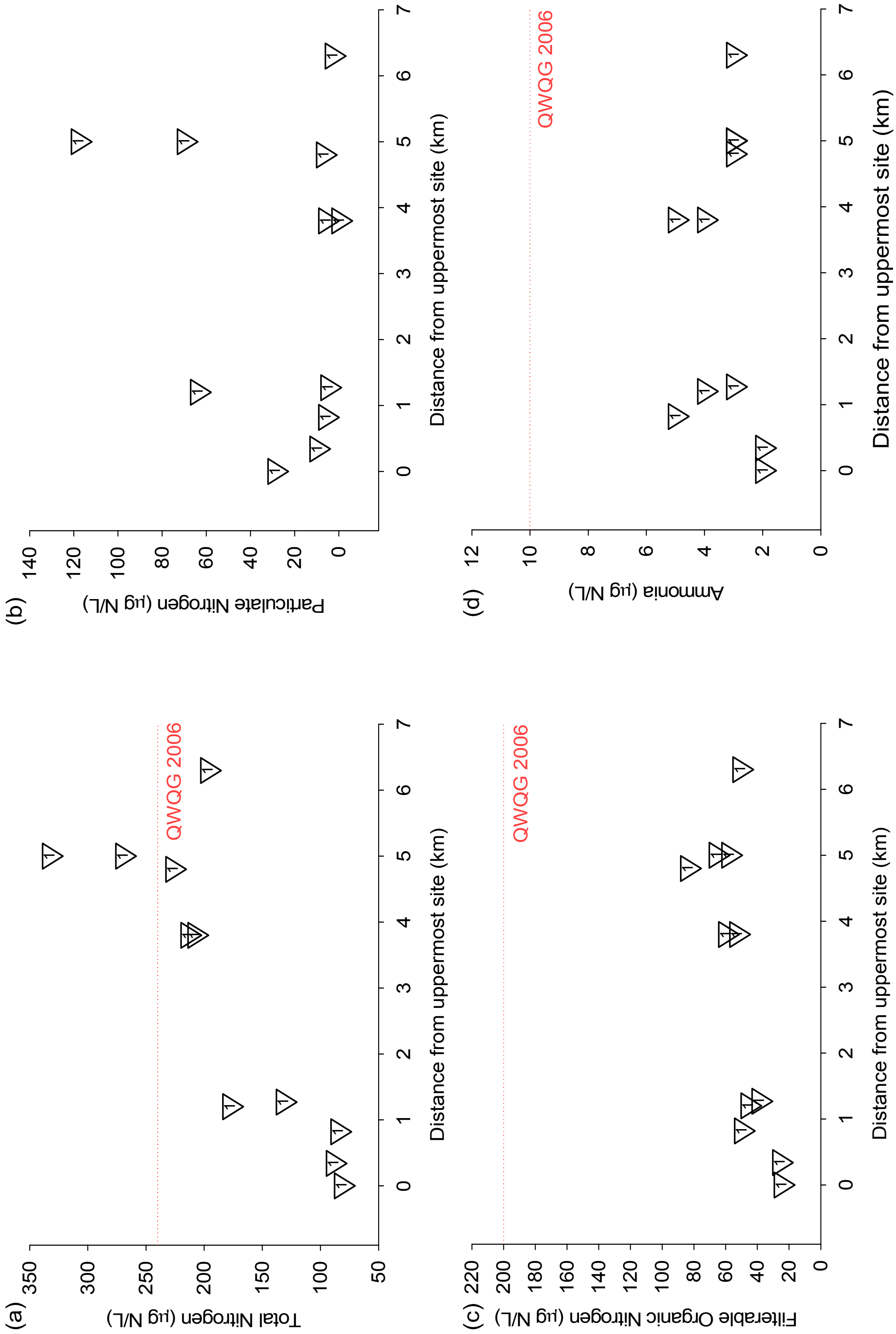


Figure 2.32. Nitrogen values measured in Woopen Creek: (a) total nitrogen, (b) particulate nitrogen, (c) filterable organic nitrogen, (d) ammonia. Numbers indicate sample series described in Table 2.7. The horizontal lines represent the Queensland Water Quality Guideline (2006) concentrations.

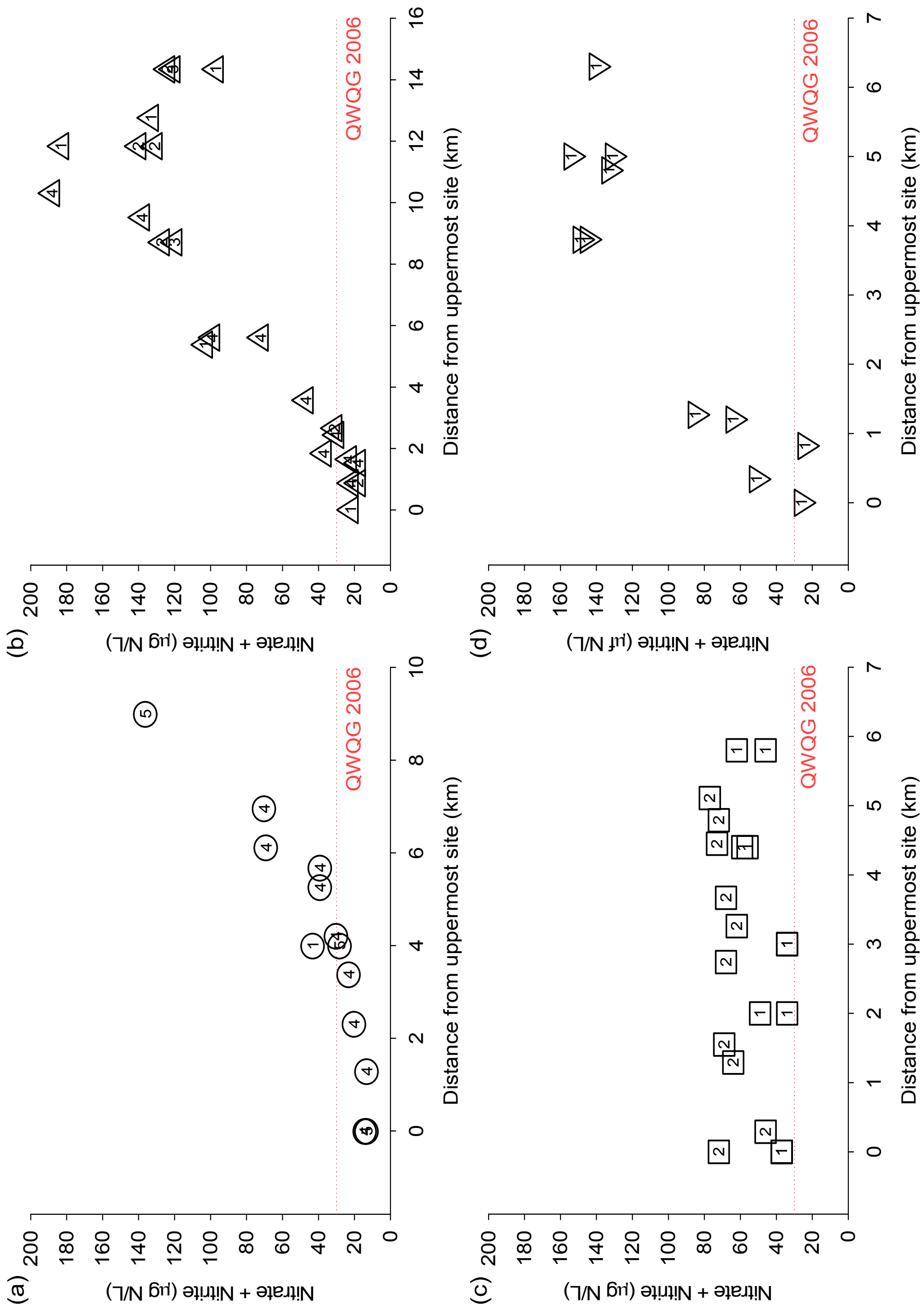


Figure 2.33. Nitrate + nitrite (=NOx) concentrations for (a) Behana Creek, (b) Babinda Creek, (c) the Little Mulgrave River, and (d) Woopen Creek. The horizontal lines represent the Queensland Water Quality Guideline (2006) concentrations.

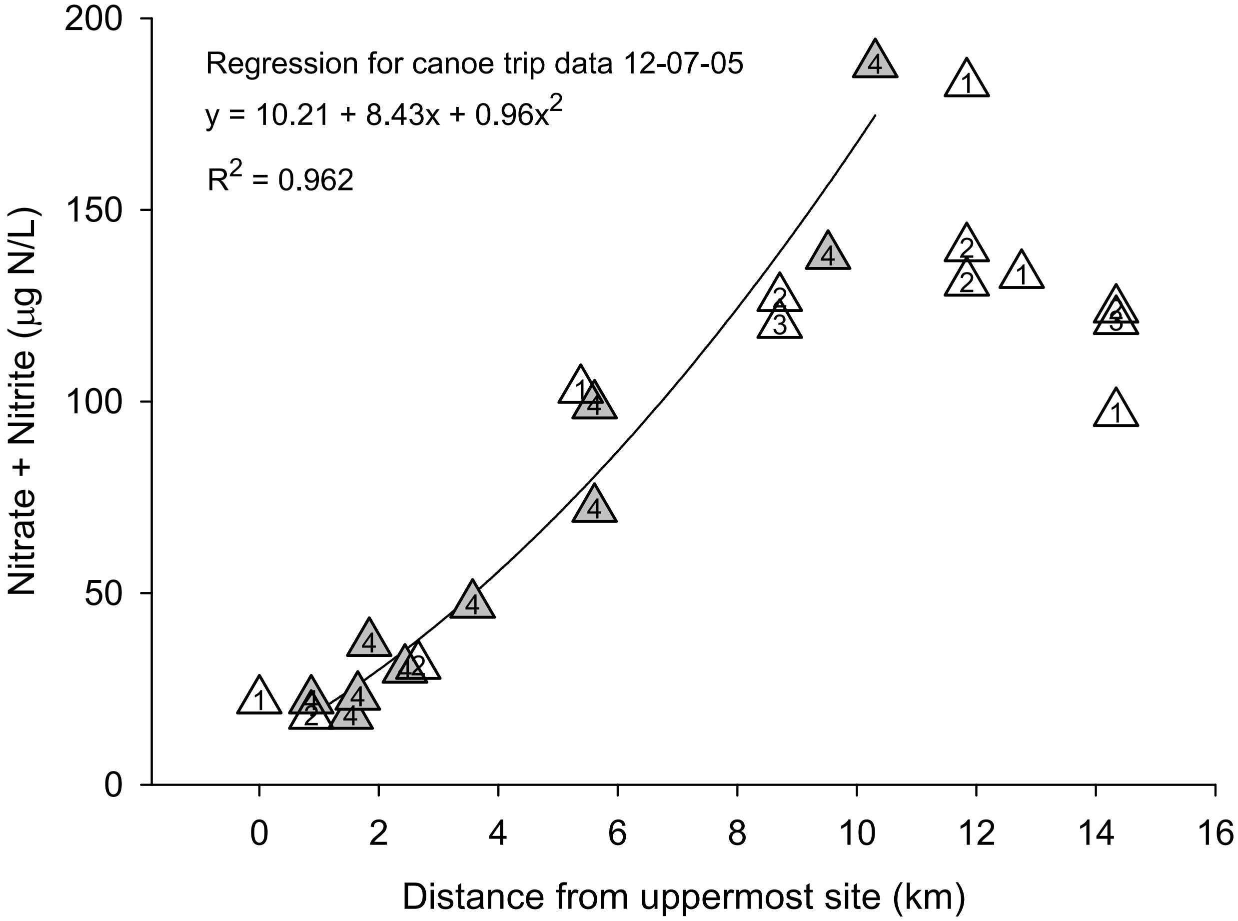


Figure 2.34. The concentration of nitrate + nitrate (NO_x) in Babinda Creek at sites along the stream continuum from the base of the range to the confluence with the Russell River. Numbers indicate sample series. Regression is a linear quadratic of data collected during sampling series 4 (canoe traverse – shaded symbols).

Figure 2.35a compares the relationship between distance downstream and NO_x concentrations in Behana and Babinda Creeks. Babinda Creek had consistently higher NO_x concentrations than Behana Creek, but the rate of increase with distance downstream was lower. Figure 2.35b shows the same relationship for the Little Mulgrave River and Woopen Creek. Both these streams had higher concentrations in the upper reaches than Behana or Babinda Creeks. The NO_x concentrations increased substantially in Woopen Creek but only increased slightly in the Little Mulgrave River.

The differences in the patterns of NO_x concentrations between streams can largely be explained by differences in the proportion of land-use types in the catchments. Figure 2.36a shows the relationship between the area of agriculture and the NO_x concentrations in the four streams. Figure 2.36b shows the relationship between the proportion of fertilised agricultural area in the catchment and the NO_x concentrations in the four streams. These figures show that concentrations of NO_x are greater at sites with a greater area of agriculture in their catchment. However, it is the proportion of fertilised agricultural area in their catchment that best predicts the NO_x concentrations in all streams. This relationship makes it possible to compare streams and thus to compare directly the effects of different land-use practices and riparian vegetation cover. For example, Behana Creek has a much greater riparian vegetation cover along the stream length and has lower concentrations of NO_x than the other three study streams.

Figure 2.37 compares the NO_x concentration in Behana and Babinda Creeks, taking into account the relative proportions of fertilised agriculture in the catchment area of each sampling site. The downstream sites in Babinda Creek, where nutrient concentrations decline, possibly as a result of algal growth, have been excluded from this comparison. An analysis of covariance, using the proportion of fertilised agricultural area in the catchment as a covariate, shows that Behana Creek has significantly lower concentrations of NO_x compared with Babinda Creek. Figure 2.38 illustrates a similar comparison for the Little Mulgrave River and Woopen Creek. Because the Little Mulgrave River has a large upper catchment, the proportion of the lower catchment used for agriculture does not vary greatly and sites tend to cluster together. In contrast, Woopen Creek has a relatively small upper catchment and so, as sites are located further downstream, the area of agriculture represents a significant proportion of the total catchment area and the influence of land use rises quickly, resulting in higher nutrient concentrations. Figure 2.39a shows that the area of agriculture increases at similar rates in all four subcatchments with distance downstream. However, Figure 2.39b shows that the proportion of catchment used for agriculture increases at a greater rate in Woopen Creek and more slowly in the Little Mulgrave River. This is because of the relative sizes of their undeveloped upper catchments and the relative influence of dilution from the upper catchment on nutrient concentrations. The differences in the proportions of the catchment used for agriculture in Behana Creek and Babinda Creek are most likely because of differences in the shapes of their lower catchments.

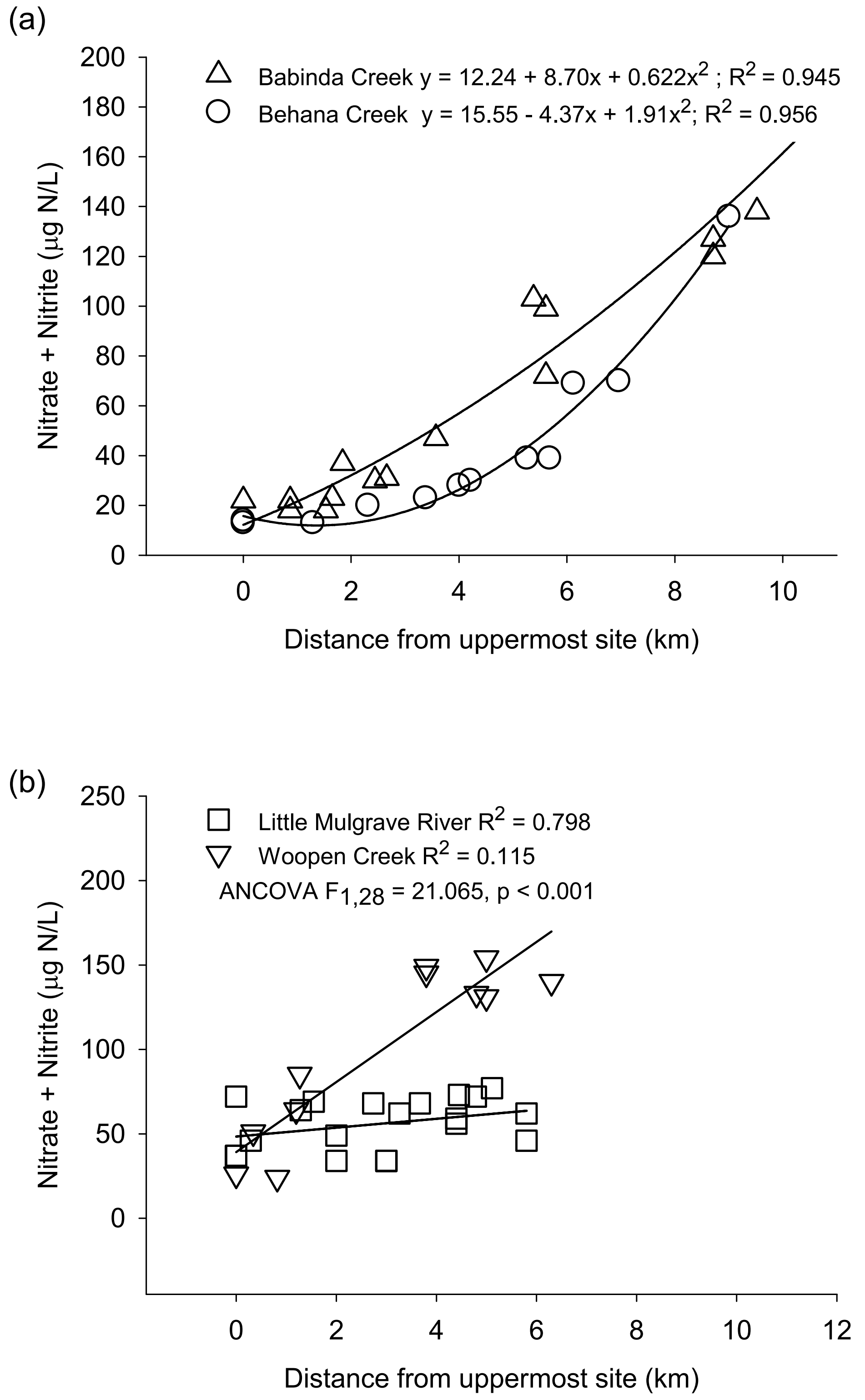
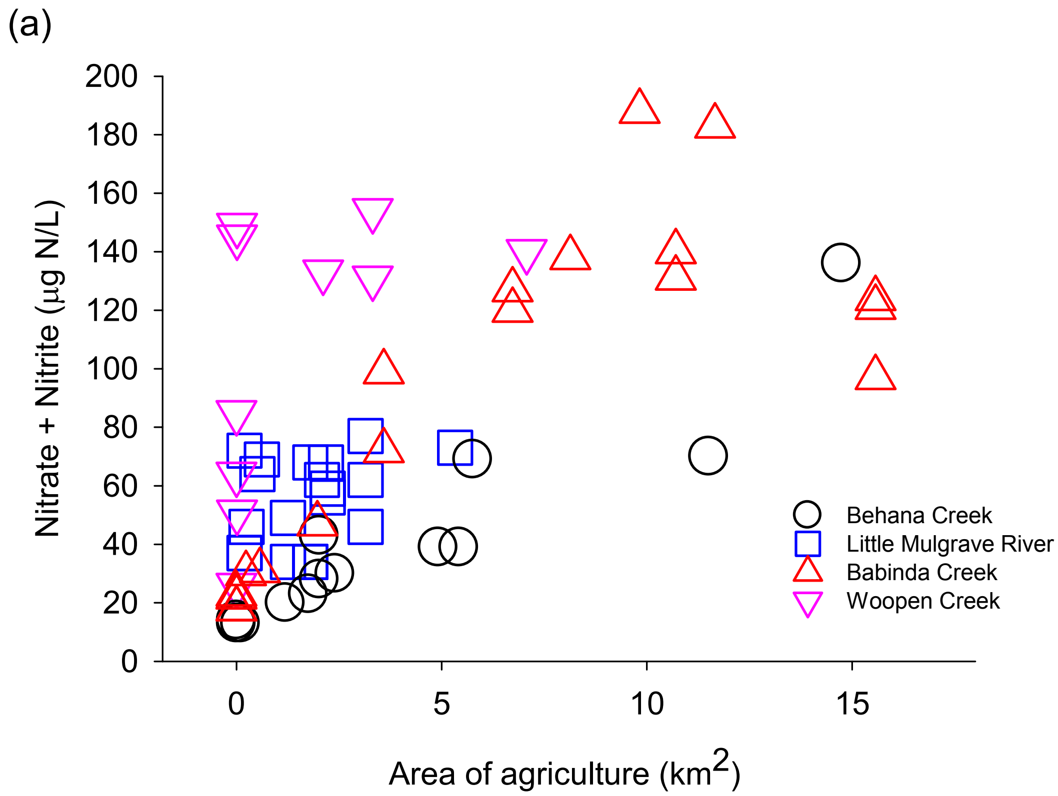


Figure 2.35. Relationship between the concentration of nitrate and nitrite (NO_x) and the distance from the upper-most sample site (at the base of the range): (a) Behana and Babinda Creeks, linear quadratic regression displayed; (b) Little Mulgrave River and Woopen Creek. Regression based on linear equation y1 = y0 + a(x). The significant difference between streams is shown by the results of an analysis of covariance (ANCOVA) using the distance from uppermost site as a covariate.



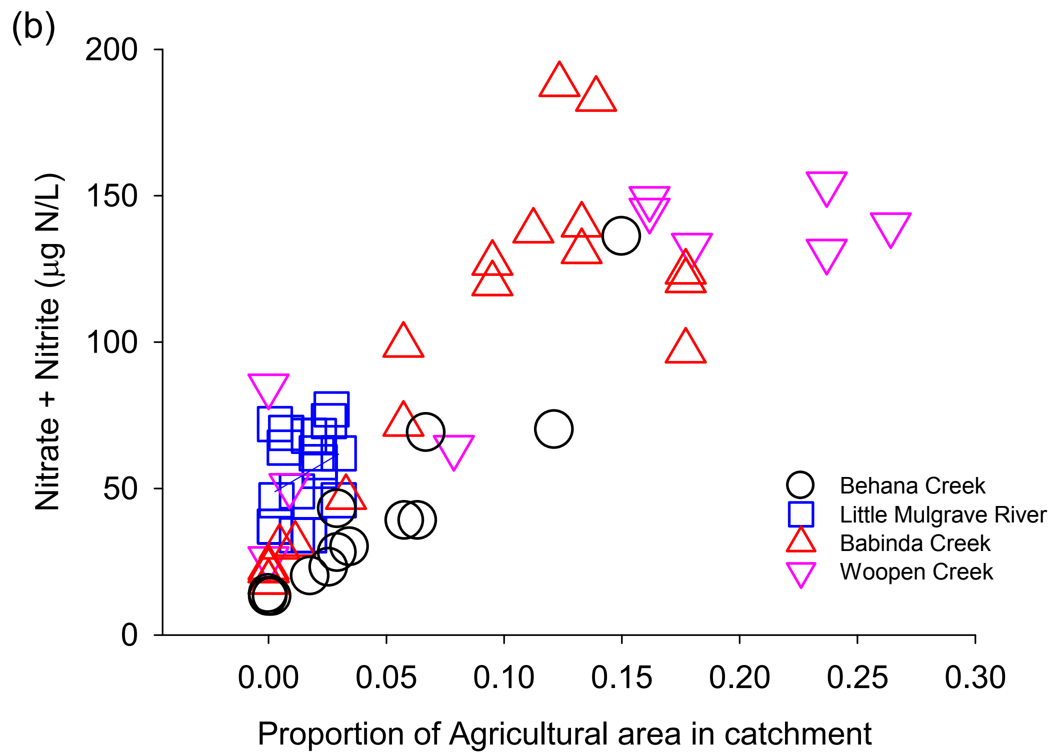


Figure 2.36. Relationship between the nitrate + nitrite (NO_x) concentrations and (a) area of agriculture in catchment, and (b) the proportion of catchment used for agriculture for the four study streams.

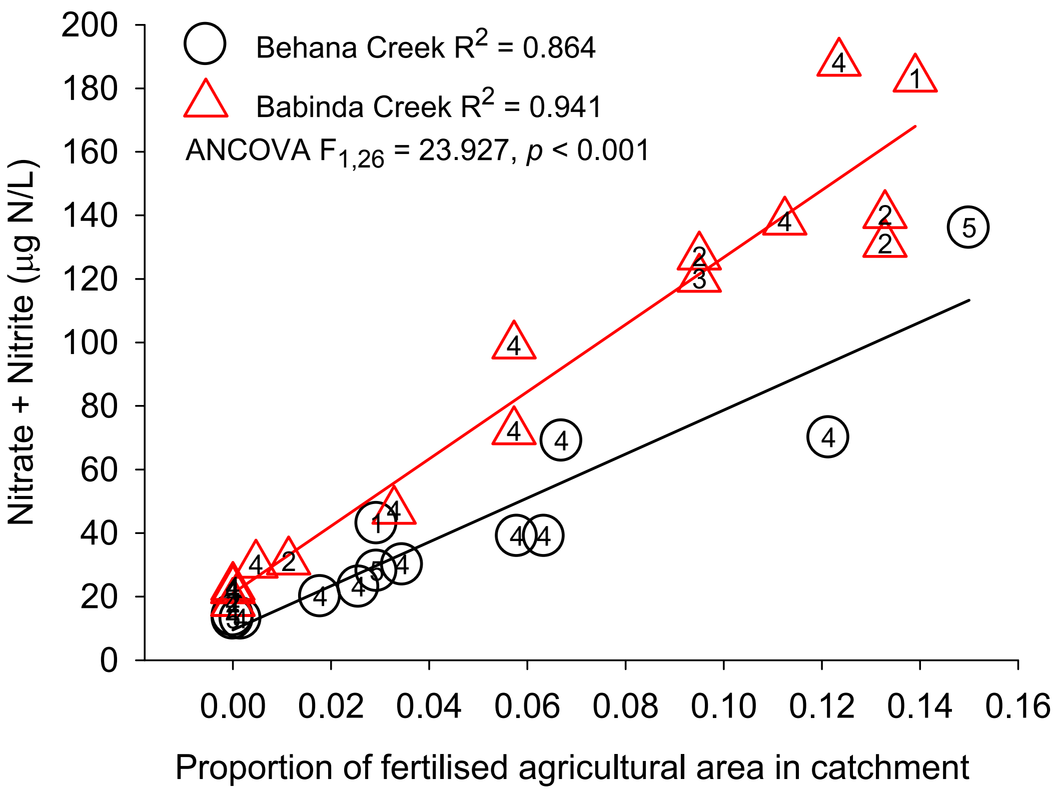


Figure 2.37. Relationship between concentration of nitrate + nitrite (NO_x) at sites along Behana and Babinda Creeks and the proportion of the catchment area currently used for agriculture. Regressions are based on the linear equation y1 = y0 + a(x). The significant difference between streams is shown by the results of an analysis of covariance (ANCOVA) using the proportion of agricultural area in catchment as a covariate.

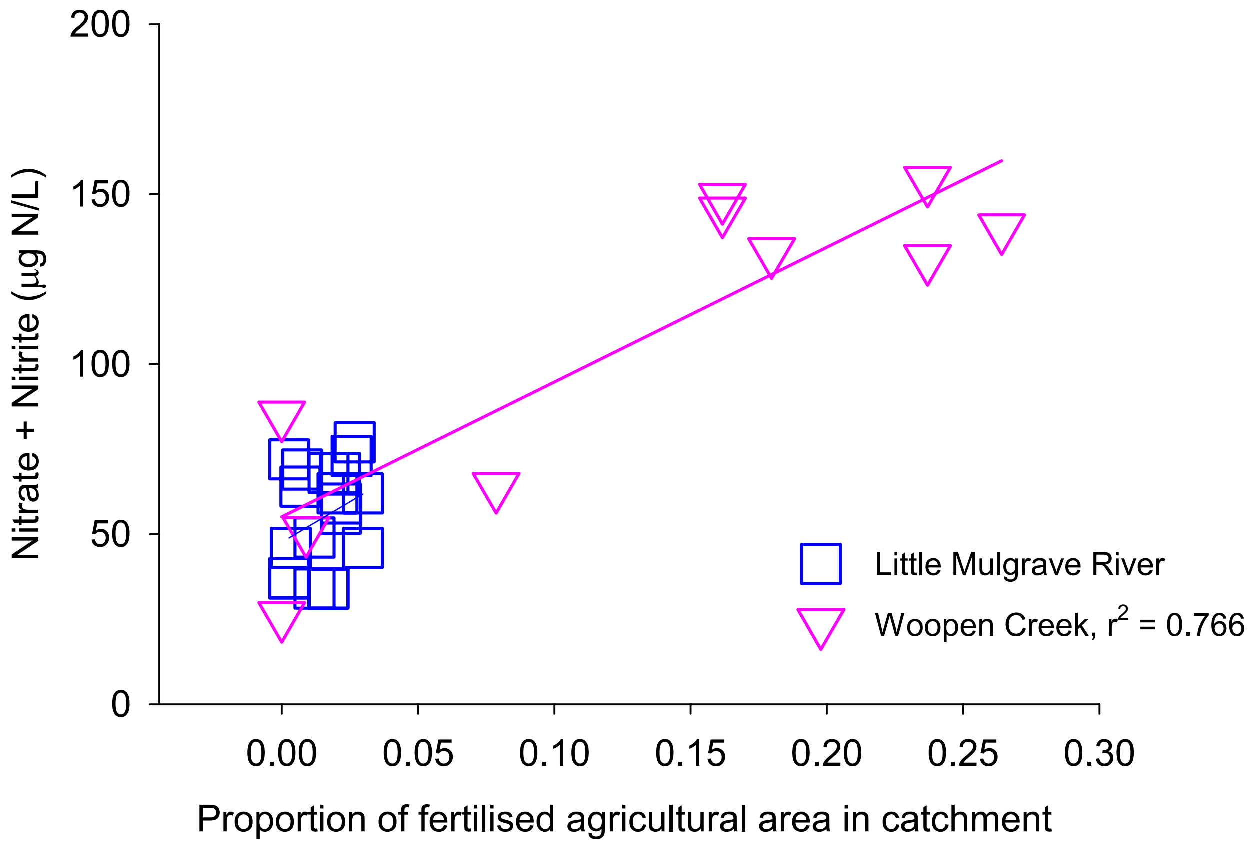


Figure 2.38. Relationship between concentration of nitrate + nitrite (NO_x) at sites along the Little Mulgrave River and Woopen Creek and the proportion catchment area currently used for agriculture. Regressions are based on the linear equation y1 = y0 + a(x).

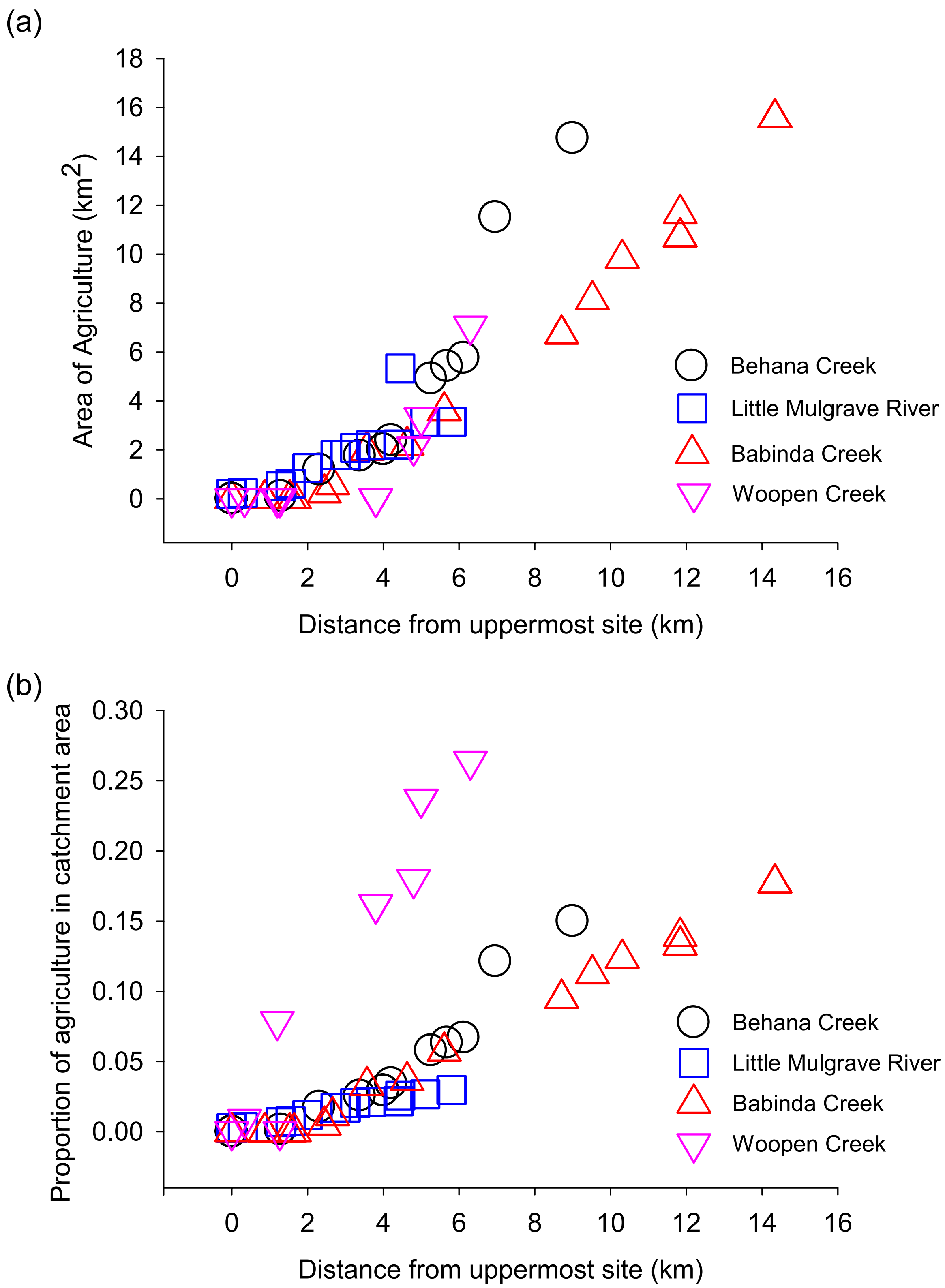


Figure 2.39. Relationship between the distance downstream from the uppermost study site and (a) the area of agriculture in the catchment, and (b) the proportion of catchment area used for agriculture for the four study streams.

The effect of rainfall and spate during study period

The lower concentrations of NO_x in Behana Creek compared with Babinda Creek are likely the result of the differences in riparian cover and land-use management practices in their catchments. However, given the differences in rainfall in the two areas, this assumption is inconclusive: for example, the differences could be the result of differences in the hydrology of the two catchments, or to the timing of sample collection relative to the spate these streams experienced during the sampling period. A flow event on July 5 allowed us to examine this. Figure 2.40 shows the effect of the July 5 high flows on nutrient concentrations in the Little Mulgrave River. It was coincidental that water samples were collected from this stream just before and just after this event. Samples collected on July 4 had much lower nutrient concentrations than those taken on July 5. These results also show that particulate nutrients increased at an exponential rate with distance downstream during a flood event, but were near-constant during low flows. NO_x increased with distance downstream during both high and low flows. However, the concentration of filterable reactive phosphorus decreased during high flows with distance downstream, but increased during low flows.

The data presented in Figure 2.37, showing the difference in the concentration of NO_x in Behana Creek compared with Babinda Creek, represent samples taken at a number of occasions, some before the spate and some one week after (as indicated by the sample series codes in the figure symbols). The increase in the concentration of NO_x with distance or the proportion of fertilised agriculture in the catchment was a feature of samples taken before and after the spate. The concentration of NO_x in samples from the Little Mulgrave River, collected before the spate, also increased with distance (Figure 2.37c) but at a lower level than immediately after the spate.

2.5.6 Phosphorus

Concentrations of phosphorus were variable within and between streams (Figures 2.41 to 2.44). Concentrations of total phosphorus exceeded QWQG 2006 guidelines at nearly all sites in all four study streams as a result of high concentrations of particulate phosphorus and filterable organic phosphorus. Concentrations of filterable reactive phosphorus only exceeded these guidelines consistently at sites in the lower reaches of Woopen Creek. The Little Mulgrave River had the highest concentration of total phosphorus of all the study streams (Figure 2.45). However, these high concentrations were due to samples being collected during high flows associated with the spate that occurred mid-way through the survey. Concentrations of phosphorus in samples taken in the Little Mulgrave River before this spate were more comparable to Woopen Creek and the higher concentrations recorded in Babinda Creek, but were higher than those recorded in Behana Creek. There was a notable peak in total phosphorus concentrations in Babinda Creek a few kilometres downstream from the uppermost site, associated mainly with particulate phosphorus. This peak soon receded, indicating that it was due to an isolated source. It may have been because of the input from the Double Barrel Creek tributary that is just upstream from these sites and has considerable area of its catchment used for banana farming and orchard crops.

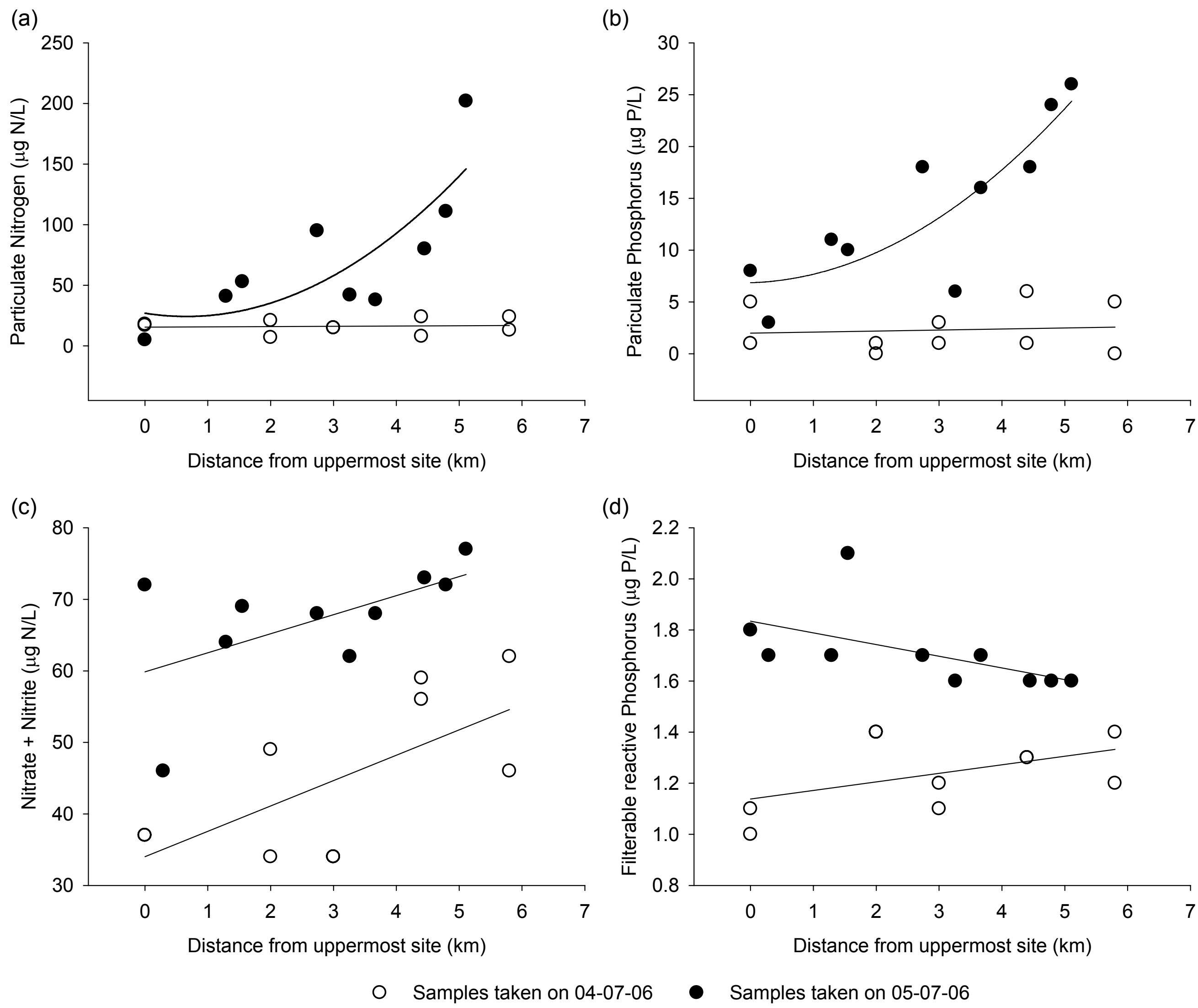


Figure 2.40. Nutrient concentrations in the Little Mulgrave River before and after a spate. Open circles (○) represent samples taken before spate on July 4, 2005. Filled circles (●) represent samples taken after the spate, on the July 5, 2005. (a) Particulate nitrogen concentrations, (b) particulate phosphorus concentrations, (c) nitrate + nitrite (NO_x), and (d) filterable reactive phosphorus concentrations.

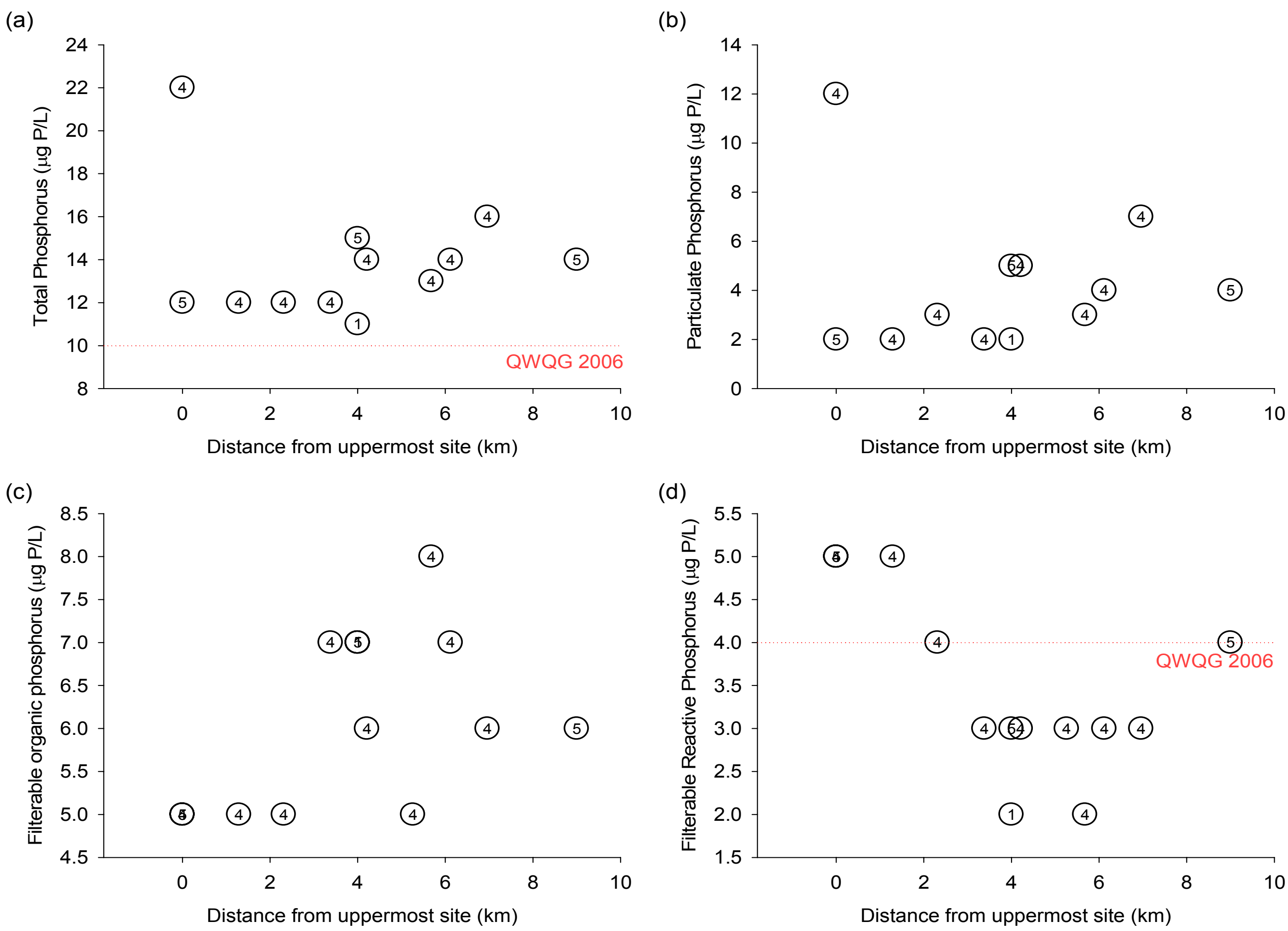


Figure 2.41. Phosphorus values measured in Behana Creek: (a) total phosphorus, (b) particulate phosphorus, (c) filterable organic phosphorus, (d) filterable reactive phosphorus. Numbers indicate sample series described in Table 2.7. The horizontal lines represent the Queensland Water Quality Guideline (2006) concentrations.

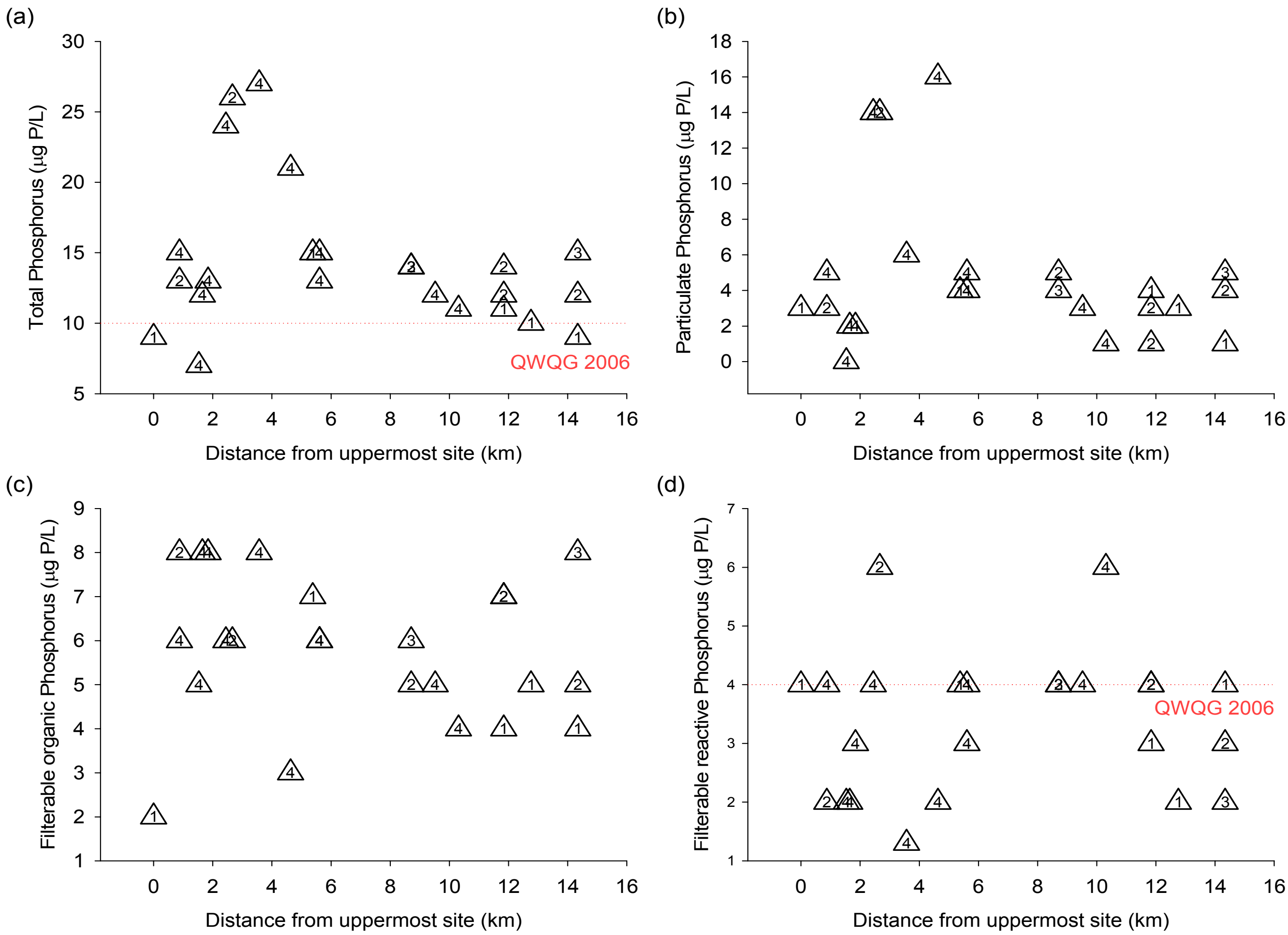


Figure 2.42. Phosphorus values measured in Babinda Creek: (a) total phosphorus, (b) particulate phosphorus, (c) filterable organic phosphorus, (d) filterable reactive phosphorus. Numbers indicate sample series described in Table 2.7. The horizontal lines represent the Queensland Water Quality Guideline (2006) concentrations.

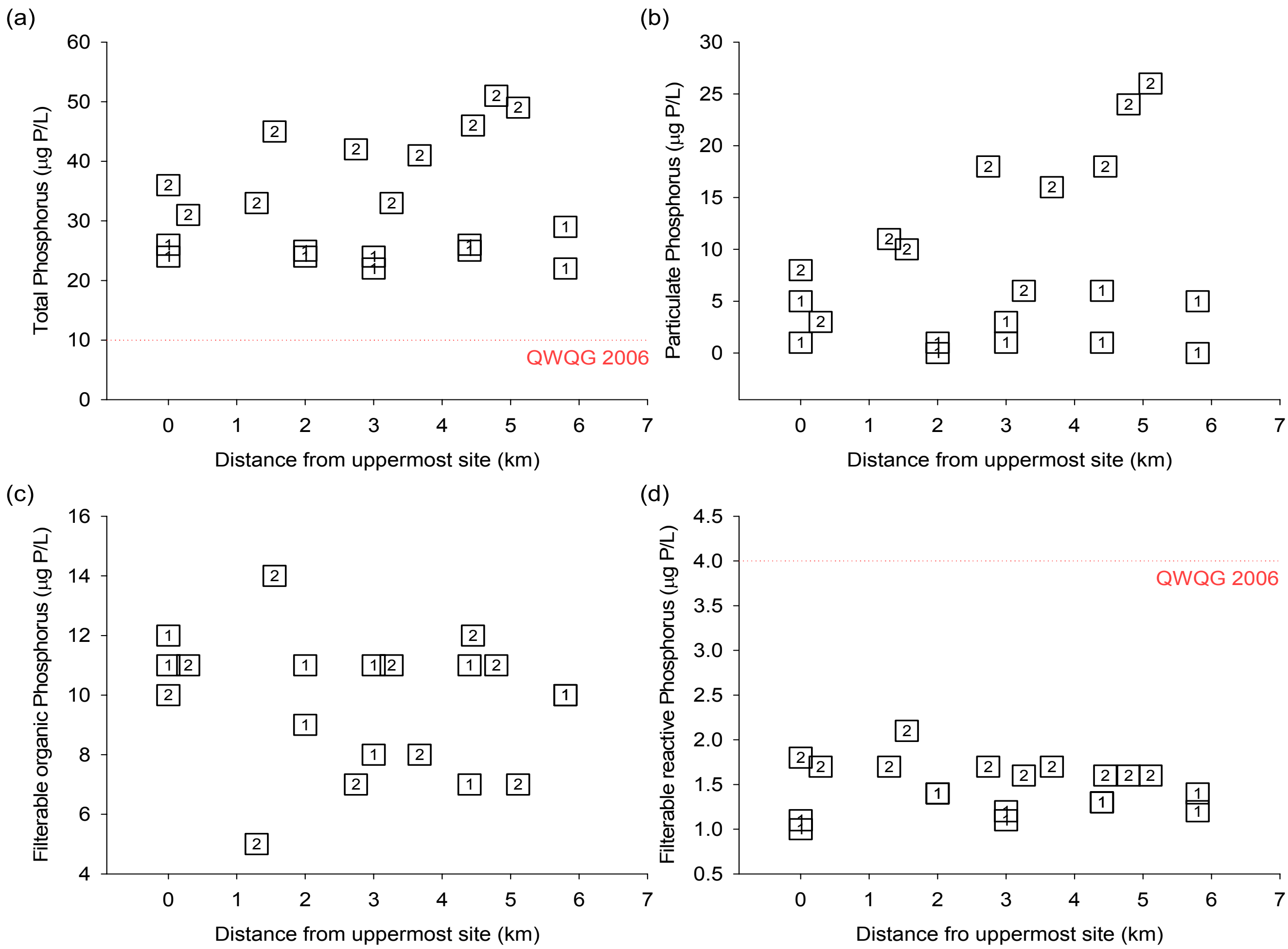


Figure 2.43. Phosphorus values measured in the Little Mulgrave River: (a) total phosphorus, (b) particulate phosphorus, (c) filterable organic phosphorus, (d) filterable reactive phosphorus. Numbers indicate sample series described in Table 2.7. The horizontal lines represent the Queensland Water Quality Guideline (2006) concentrations.

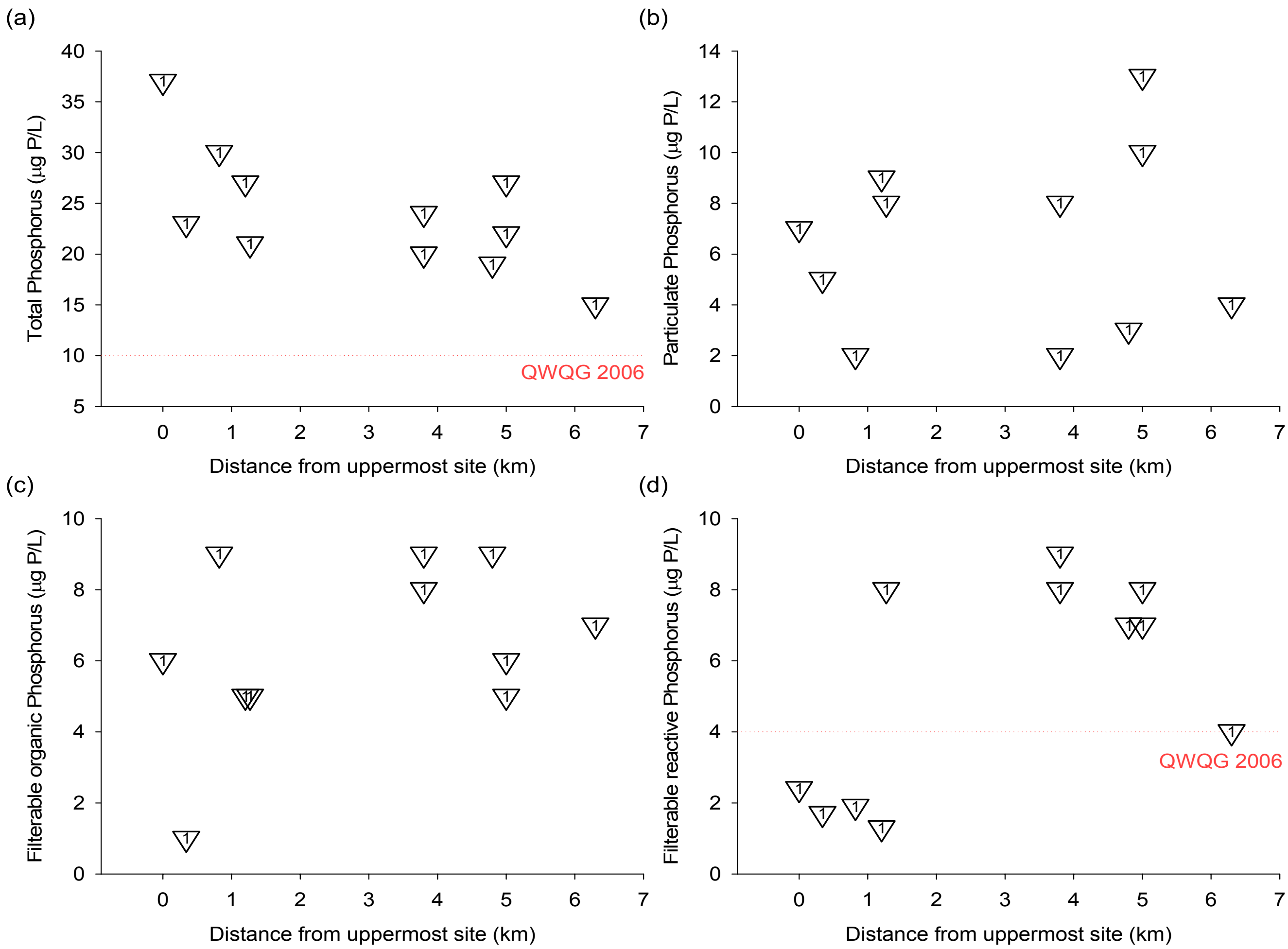


Figure 2.44. Phosphorus values measured in Woopen Creek: (a) total phosphorus, (b) particulate phosphorus, (c) filterable organic phosphorus, (d) filterable reactive phosphorus. Numbers indicate sample series described in Table 2.7. The horizontal lines represent the Queensland Water Quality Guideline (2006) concentrations.

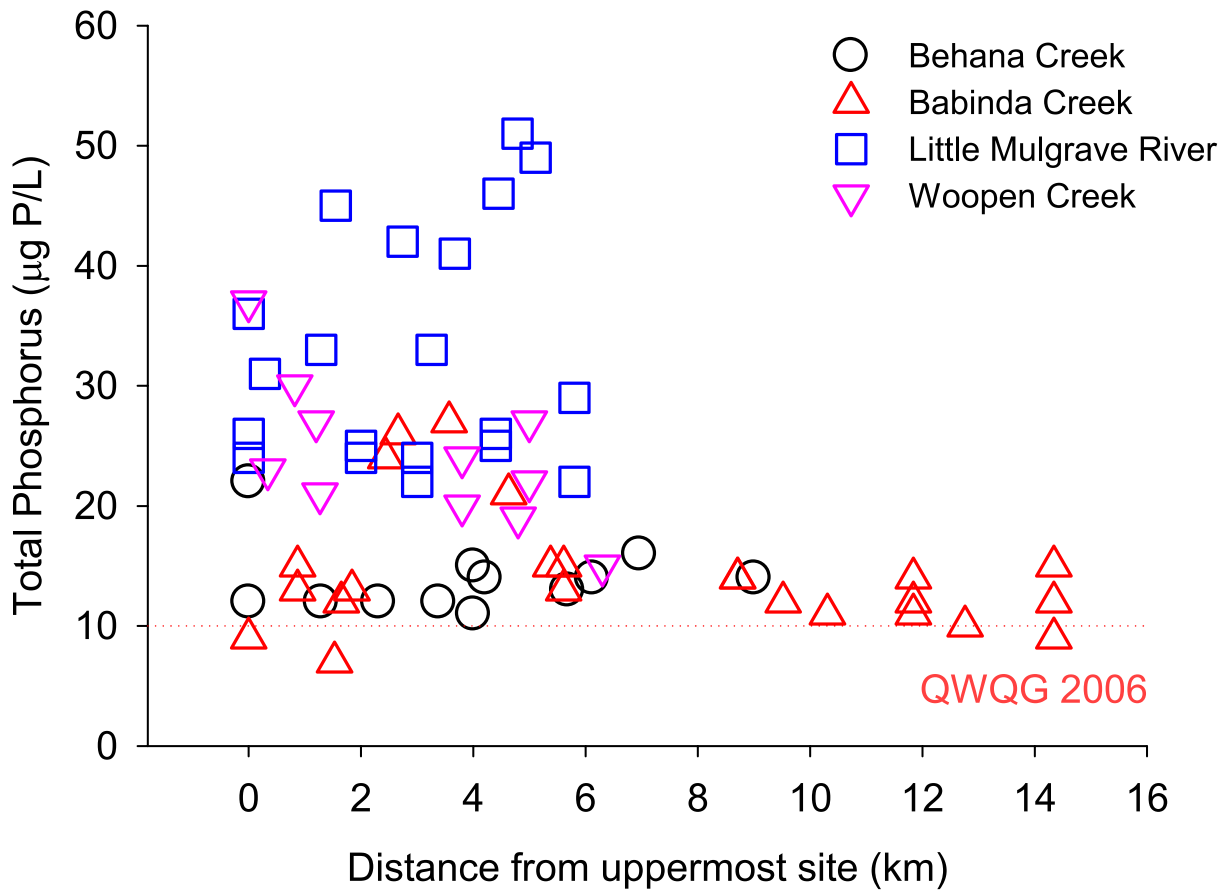


Figure 2.45. Total phosphorus concentrations in the four study streams. The horizontal line represents the Queensland Water Quality Guideline (2006) concentrations.

2.6 Discussion

The purpose of this component of the study was to provide a background physico-chemical description of the study area and study streams and provide an assessment of the types and extent of physico-chemical disturbances, to support the analyses of biotic distributions described in subsequent chapters. This report, therefore, provides a detailed description of the study streams but is only preliminary in that it only touches on some of the physico-chemical processes that are operating in these systems. The bio-physical processes affecting water quality in wet tropics streams have been addressed in more detail in the Water Quality report of the Catchment to Reef Program.

The Russell-Mulgrave catchment area is typical of the central wet tropics coastal region. Its climate is wet and is dominated by the influence of the Mt Bartle Frere – Mt Bellenden Ker mountain massif. All of the sub-catchments surveyed in this study had broad similarities. They generally had steep upper catchments with similar geology and vegetation cover and land-use influences, with sugar-cane production the main agricultural activity. All the streams in this study were perennial and could be classified as ‘flashy’ in that their hydrographs respond very quickly to rainfall and are susceptible to frequent spates. They all had strong longitudinal gradients in substratum type, channel geometry and nutrient concentrations. However, they also differed in several ways. For example, Woopen Creek was much smaller in catchment area and channel dimensions than the other three sub-catchments. The Little Mulgrave had a larger catchment area upstream from where human land-use influences appear, which included a long section of low-relief stream. The catchments of all the other study streams had been developed to the base of the range, with the protected sections rising steeply. The contrast between the riparian cover in the Little Mulgrave River and Behana Creek and that in Babinda Creek and Woopen Creek was striking, with the latter streams degraded as a consequence of riparian clearing, bank destabilisation and weed invasion. This degradation was reflected in the scores of overall geomorphic condition of the study sites in these streams. Behana and Babinda Creeks were the most similar to each other in size and land-use types and potentially provide a very useful comparison between streams with different land management practices and riparian vegetation cover.

Only Babinda Creek demonstrated any meander in its plan form. This contributed to the stream being longer than Behana Creek, even though the linear distance across the floodplain was similar. Stream length will influence the amount of stream habitat area in a catchment, the travel time of water in a drainage system and the availability of sediment for transport. However, Babinda Creek was disturbed along most of its lowland sections. Riparian vegetation had been dramatically altered, with large trees replaced by herbaceous vegetation such as Singapore daisy and para grass, as well as large stands of bamboo and other weeds. As a consequence of the change in riparian structure, geomorphic conditions of the banks were generally low in Babinda Creek. Mass bank failures were evident along many reaches of the stream. This was not because the infestation of weeds had necessarily increased the susceptibility of the banks to erosion. Instead, stands of para grass would protect a bank or bar and deflect flows to the opposing bank, increasing its susceptibility. Where both banks had dense stands of either para grass or Singapore daisy, the flow had become channelised and velocities had increased. This appears to have caused incision in the lower reaches of Babinda Creek, creating a uniformly deep, featureless sandy channel, with swift flows. In contrast, the lower reaches of Behana Creek were quite shallow with flows switching from side to side and bars were present. Although there were stands of para grass in Behana Creek, the edge habitats were quite varied and included tree roots and other structure. In Babinda Creek this habitat had been almost completely overgrown and consisted of inundated stands of Singapore daisy or para grass, and habitat diversity was generally low.

The narrowing of the channel geometry in Behana and Babinda Creeks demonstrates a distinctive characteristic of many wet tropics streams. These streams emerge from the mountain range with high velocity and volume, and during high flows the sudden change in slope results in them 'spilling out' over the floodplain as soon as they are no longer constrained by their valley. The result is a reduction in potential bank-full discharge and within-channel stream power. As a consequence, the full stream discharge does not contribute to channel formation. Some streams of the Illawarra region of New South Wales have also been shown to exhibit downstream reductions in channel size, attributed largely to the rapid decline in slope and stream power, the availability of the floodplain overbanking and the cohesive nature of the bank sediments (Nanson and Young 1981). Kapitzke *et al*. (1998) also found that several coastal streams and rivers of northern Queensland (e.g. Russell and Mulgrave rivers), exhibit a downstream decrease in channel dimensions on the floodplain close to the coast, attributed largely to underlying geology, sediment type and geological formations adjacent to the channel margins. In the streams in this study, at calculated bank-full conditions, stream power per unit area within the channel reduces as the slope and velocity decline across the floodplain, so the potential for geomorphic work within the channel is reduced with distance downstream. The decrease in velocity has also created a distinct gradient in sediment particle sizes long the stream, with the upper sections characterised by large boulders and the lower sections of the longer streams (Behana and Babinda) by sands.

The type of material found in the bed and bars of the stream reflects the depth and slope of the bank-full flow. Measurements of bed materials are typically conducted to help characterise the stream's ability to carry different size sediments. However, in this study, the emphasis was on the influence of bed materials on habitat structure and the distribution of the biotic components of the system. The gradient in bed particle sizes will have a strong effect on in-stream habitat types for the biota, especially benthic plants and invertebrates, but also many benthic fish. The sediment particle sizes will also influence the retention of organic material such as leaf litter or fallen branches and the ability of primary producers to attach and grow. Thus, the bed sediments also influence the metabolism and trophic dynamics of the stream section. Large rocks and boulders and large woody debris also influence local current velocities, sediment deposition and scour dynamics, providing additional physical and habitat diversity.

As well as strong geomorphic gradients in these streams there were very distinct water quality gradients, with temperature and nutrient concentrations increasing downstream, and pH decreasing. All four study streams had elevated nutrient concentrations, although these were relatively low compared with levels recorded in streams on the Herbert River floodplain (Bramley and Roth 2002; Pearson *et al*. 2003). Nonetheless, several nutrients exceeded concentrations recommended in the QWQG 2006 and were well above concentrations that would induce a biological response (Pearson and Connolly 2000). NO_x concentrations were high in all streams and increased rapidly downstream, reflecting the proportion of agriculture in the catchment area above the site from where samples were collected. Therefore, the shape of the catchment and the relative dilution from the upstream catchment had a large influence on the nutrient concentrations observed at a site.

The decline in NO_x concentration at the downstream sites on Babinda Creek was interesting and appears to be the result of absorption by excessive growth of mats of blue-green algae observed on benthic substrata, including the sand bed, and long masses of filamentous algae attached to para grass stands and other substrata at these downstream sites. It was beyond the scope of our study to investigate the cause of the growth of these algae but is interesting that it was not observed at any other sections of stream in the study area and that these sites were directly downstream from the Babinda township and the Babinda sugar mill.

Phosphorus concentrations also exceeded QWQG 2006 in all streams except Behana Creek. This was mainly because of high concentrations of particulate and filterable organic phosphorus, possibly a product of recent runoff. Filterable reactive phosphorus concentrations were generally low and only exceeded guidelines in the lower reaches of Woopen Creek. The peak in total phosphorus in Babinda Creek downstream from the confluence of Double Barrel Creek indicates a localised source that is then diluted or absorbed further downstream. This may originate from orchard and banana crops that are grown in the catchment of Double Barrel Creek and demonstrates the different water quality influences of different land uses. Sugar-cane production uses little or no phosphorus fertilizer and the low phosphorus concentrations recorded in Behana Creek reflects the fact that sugar cane is the only crop grown in that catchment.

The comparisons between water samples collected in the Little Mulgrave River before and after heavy rainfall also demonstrate that rainfall has a large influence on the concentrations of nutrients at a site at any time, and the patterns of the downstream nutrient concentration gradient in these streams. It was expected that greater quantities of particulate and dissolved nutrients would be transported to the stream during rainfall. However, it was interesting that, even during low flows, NO_x concentrations increased with distance downstream, indicating a chronic leaking of dissolved inorganic nitrogen from adjacent land. In contrast, the change in the gradient of filterable reactive phosphorus with rainfall from an increasing slope to a decreasing slope with distance downstream is most likely the result of dilution by rainfall.

The longitudinal gradient in stream nutrients and the effect of rainfall highlight that it is impractical to compare the nutrient concentrations between sites and attribute a cause, without considering a number of contributing factors. For example, there is a significant gradient in rainfall in the lowland sections of the Russell-Mulgrave catchment area, with rainfall decreasing north from Babinda towards Gordonvale. There is also a big difference between precipitation rates falling on the peaks of the ranges compared with the lowland floodplains. This means that there is generally consistent dilution in the study streams coming from the pristine upper parts of their catchments, but the influence of floodplain drainage may vary throughout the study area. It is not straightforward to conclude whether higher rainfall in the floodplain results in higher rates of transport of contaminants to the streams draining the floodplain or if it may have a dilution effect. If rain falls during harvest or fertilizer application, or after a dry spell, then greater rainfall may transport large quantities of nutrients and oxygen-demanding organic matter. However, if rainfall persists for long periods, dilution may eventually occur. Persistent high rainfall may also saturate the soil, which will affect runoff dynamics and groundwater movement. Persistently saturated soil may also become anoxic, affecting the decomposition of nutrients and organic matter (including sugars from cane juice) with significant implications for water quality (Pearson *et al.* 2003). Therefore, comparisons of water quality between Babinda Creek, which has higher rainfall falling on its floodplain, and the other three study streams need to be treated with some caution.

Nevertheless, the difference between the nutrient concentrations in Behana and Babinda Creeks is very interesting given that they have very different riparian cover and land-use management practices. Land-use management in Behana Creek approaches current best management practice (BMP) with regard to protecting river health and reducing the transport of agricultural contaminants to the Great Barrier Reef lagoon. It is, therefore, encouraging that nutrient concentrations were lower in Behana Creek. The comparison between Behana and Babinda Creeks represent a large-scale test of the effectiveness of riparian vegetation and BMP on reducing agricultural contamination. Tracking the influence of agriculture along each stream and accounting for differences in the relative proportion and distribution of land-use cover appears to be a powerful means to examine these effects on a large scale. However, this comparison is currently unreplicated and may be confounded by differences in rainfall and hydrology in the two floodplains of these catchments.

For the purposes of this study it is sufficient to show that the concentrations of nutrients were higher in Babinda Creek. However, it is important to point out that although nutrient concentrations were lower in Behana Creek, they were still at levels that might be expected to induce a biological response. This means that the effect of agricultural nutrients on the biotic components in the stream ecosystems in this study is difficult to test because all streams have elevated nutrient concentrations.

2.7 Conclusions and Recommendations

The descriptions provided here offer a sound basis for comparing biological attributes among streams. They clearly demonstrate downstream gradients in several characteristics of the study streams, including stream power, substrata type, temperature, conductivity and nutrient concentrations. These gradients were consistent with the hydrogeomorphology of the streams and the influence of agriculture on them. Contrasts between streams, in particular, indicated strong land-use influences. How these relate to the biota of the streams is dealt with in the following chapters. But it seems very likely that our detailed approach to stream descriptions will be vital in disentangling potential influences on the ecological integrity of the streams. This approach would clearly benefit any future health assessment of streams in the region and elsewhere.

2.8 Acknowledgements

We are indebted to Barry Butler, Jon Brodie and John Faithful for advice on water quality issues, the ACTFR laboratory for processing of water samples, Allan Hooper of DNRW for supplying flow data, Alana Grech and the Bureau of Meteorology for providing rainfall data, Paul Duncasson for laboratory processing of sediments, and of course to the late Garry Werren for his constructive inputs in the early stages of the project. We wish to especially acknowledge the assistance of Bruce Corcoran and other landholders who are showing remarkable leadership in the management of lands that are providing products and conservation values in tandem.

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Chapter 3

Aquatic Macrophytes as Indicators of Catchment Land-use and Water Quality in Wet Tropics Streams

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3.1 Introduction

Macrophyte assemblage structure in undisturbed streams and rivers varies naturally across gradients of resource availability and hydraulic/hydrologic disturbance (Biggs 1996). Forested headwater streams are generally characterised by low resource (nutrient and/or light) availability. These streams are devoid of macrophytes or are colonised by non-vascular macrophytes such as mosses and liverworts (Westlake and Dawson 1975; Sheath *et al.* 1986; Howard-Williams *et al.* 1987; Everitt and Burkholder 1991; Dawson *et al.* 1999; Scarlett and O'Hare 2006). Mosses and liverworts commonly have low light compensation points (the point at which photosynthesis balances respiration and net CO_2 exchange is zero) and are morphologically suited to the relatively high hydraulic stresses (high stream gradients, high water velocities and coarse substrata) that occur in headwater streams (Biggs 1996; Suren *et al.* 2000). With increasing distance downstream, resource availability increases, streambed slopes decrease, stream substrata become finer and vascular macrophytes dominate (Holmes and Whitton 1977; Dawson 1988; French and Chambers 1996). In addition, a greater diversity of growth forms may occur when compared with forested headwater streams (Baattrup-Pedersen *et al.* 2006). Whilst greater resource availability may promote greater macrophyte growth in middle to lower reaches of streams and rivers, the distribution and abundance of macrophytes throughout any given stream reach may nonetheless vary considerably, reflecting habitat heterogeneity and the occurrence of disturbance events, such as floods and droughts, that periodically remove macrophytes (Bilby 1977; Sand-Jensen and Madsen 1992; Biggs 1996).

Resource and disturbance gradients in river catchments may be altered by changes to catchment land use, particularly through agriculture and urbanisation (e.g. Davies-Colley 1997; Harding *et al.* 1999; Riley *et al.* 2003; Snyder *et al.* 2003), which may in turn influence macrophyte assemblage structure. Changes in macrophyte assemblage structure often associated with changed catchment land uses include increased macrophyte abundance, changes in species dominance, altered species richness, and the invasion and establishment of alien species (Bunn *et al.* 1998; Carr and Chambers 1998; Demars and Harper 1998; King and Buckney 2000; Sosiak 2002). Macrophytes may therefore be useful indicators of landscape and riparian disturbance in the wet tropics region of north Queensland, where declining water quality, particularly increased nutrient and sediment loads, may be affecting the health of nearshore reef systems in the Great Barrier Reef Lagoon (e.g. Neil *et al.* 2002; Brodie and Mitchell 2005; McKergow *et al.* 2005; O'Reagain *et al.* 2005). Macrophytes have not been widely used as biomonitoring tools for Australian streams and rivers despite the recognition of their potential for use in stream bioassessments (Cranston *et al.* 1995; ANZECC and ARMCANZ 2000; Whittington 2000; Mackay *et al.* 2003). However, macrophytes have been used successfully to monitor the trophic status of European streams and rivers (Haslam 1982; Newbold and Holmes 1987; Robach *et al.* 1996; Demars and Harper 1998; Holmes *et al.* 1998; Kelly and Whitton 1998; Thiebaut *et al.* 2002). The use of macrophytes as bioindicators of trophic status assumes that predictable relationships exist between assemblage attributes and physico-chemical habitat (Carbiener *et al.* 1990; Robach *et al.* 1996; Ali *et al.* 1999). To date predictable relationships between macrophyte assemblage structure and environmental parameters have not been widely established for Australian lotic ecosystems, although conceptual models relating these attributes have been developed (Biggs 1996; Riis and Biggs 2003; Mackay *et al.* 2003).

Predicted macrophyte assemblage structure following land-use changes

Predicted changes in macrophyte assemblage structure following landscape alterations are summarised in Table 3.1. Potential causes of change in macrophyte assemblage structure following landscape changes investigated in this report include increased nutrient availability from diffuse or point sources in agricultural and/or urban areas (Brodie and Mitchell 2005) and loss of riparian vegetation, resulting in reduced shading and increased water temperatures.

Increased nitrogen and phosphorus loads to streams arising from fertiliser application, sewage effluent and other anthropogenic activities (Young *et al*. 1996) are often cited as a cause of excessive algal or macrophyte growth in streams and other waterways (King and Buckney 2000; Sosiak 2002; Wade *et al*. 2002; Carr *et al*. 2003; Perna and Burrows 2005). Nitrogen and phosphorus are essential macronutrients for plant growth (Salisbury and Ross 1985). The extent to which nutrient addition can promote macrophyte growth in wet tropics streams is dependent upon light availability, background nutrient concentrations and stream hydraulics. In shaded environments such as headwater streams, nutrient addition may not stimulate macrophyte growth greatly because of shading by the riparian canopy. Similarly, nutrient addition to streams where background nutrient concentrations are already sufficient for plant growth (e.g. lowland streams) are not predicted to greatly stimulate macrophyte growth but may drive changes in the abundances of other stream autotrophs such as epiphytic algae. Thus increased nutrient concentrations are predicted to have the greatest impact in streams with little riparian canopy cover and low background nutrient concentrations.

Increased nutrient availability, and habitat disturbance in general, may facilitate the establishment and proliferation of alien species (Bunn *et al*. 1998; King and Buckney 2000; Douglas *et al*. 2005; Perna and Burrows 2005). The environmental constraints hypothesis (*sensu* Galatowitsch *et al*. 1999) suggests that the invasion of some species (e.g. alien taxa) was formerly prevented by resource limitation. Removal of a resource constraint enables species to invade habitats from which they were previously excluded. In tropical regions of Australia, particularly prominent components of the alien flora are grasses (Poaceae) (Russell *et al*. 1996; Williams and West 2000; Ferdinands *et al*. 2005). Many alien grass species were introduced intentionally as pasture grasses, including a number now considered noxious environmental weeds in Australia (Ferdinands *et al*. 2005). These taxa frequently invade riparian habitats and stream margins (Houston and Duivenvoorden 2002; Werren 2002). For example, para grass (*Urochloa mutica* [Forsskk.] Nguyen) is an introduced grass species known to have detrimental impacts on aquatic ecosystems of the Australian tropics (Bunn *et al*. 1997; Bunn *et al*. 1998; Ferdinands *et al*. 2005). While increases in nutrient availability may facilitate the establishment and proliferation of alien species there are few supporting data to demonstrate specific relationships between increased nutrient inputs and establishment of alien weed species common to the wet tropics. This was highlighted recently as a research need for Australian tropical rivers (Brodie and Mitchell 2005). Other mechanisms may also be invoked to explain the successful establishment of alien species, such as the absence of natural enemies and increased light availability caused by the loss of riparian canopy cover.

Table 3.1. Summary of predicted responses of aquatic macrophytes to increased nutrient and light availability in the wet tropics region.

Impact	Macrophyte Assemblages Attributes and Responses	Potential Indicators/Metrics	Hydraulic Constraints
Reference Condition	• Low macrophyte abundance in headwater streams, relatively greater abundance in lowland streams. • Low species richness in headwater streams (due to light and nutrient limitation) relative to lowland streams. • Predominance of non-vascular macrophyte species (e.g. Bryophytes) over vascular macrophyte species in headwater streams • Higher proportions of vascular (submerged, floating and emergent) macrophytes in lowland streams. • Low abundances of alien species present. • Distribution of macrophytes patchy in undisturbed streams.	• Total macrophyte cover. • Species richness. • Proportion of taxa that are native/alien. • Morphological attributes (proportion of taxa growing submerged, floating, emergent).	• Hydraulic conditions (e.g. coarse substrata, high water velocities) may naturally limit the growth of submerged vascular macrophytes in headwater streams.
Increased nutrient availability	• Minor increase in macrophyte abundance (i.e. cover or biomass) in headwater streams if riparian canopy remains intact; larger increases if riparian canopy disturbed. • Minor increase in macrophyte abundance in lowland streams (since background concentrations may already be sufficient for macrophyte growth). • Dominance by alien vegetation with higher nutrient requirements than native vegetation (or that use available nutrients more efficiently than native vegetation). • Change in species richness; loss of taxa that thrive in low nutrient environments. • Competitive dominance by one or few taxa with high nutrient requirements. • Increased abundance of submerged vascular macrophytes in upland areas even where riparian canopy cover high. • Increased algal growth under high nutrient conditions that could smother macrophytes. • Increased abundance of grasses (Poaceae).	• Total macrophyte cover. • Species richness. • Proportion of taxa that are native/alien. • Morphological attributes (proportion of taxa growing submerged, floating, emergent). • Proportion of taxa belonging to Poaceae (grasses).	• Coarse substrata may limit growth of submerged vascular macrophytes in headwater streams by limiting root penetration
Loss of riparian zone integrity	• Increased macrophyte abundance, particularly in headwater streams that are light limited. • Increased growth rates due to increased water temperatures. • Loss of shade tolerant taxa (e.g. submerged bryophytes). • Increased abundance of emergent macrophytes in headwater streams, particularly Poaceae (e.g. para grass). • Effects more pronounced in shaded areas, especially upland areas where shade-tolerant bryophytes may dominate.	• Total macrophyte cover. • Species richness. • Proportion and abundance of non-vascular taxa.	• Effects may be dependent upon substratum stability; e.g. sandy areas of lower Logan River (southeast Queensland) devoid of submerged macrophytes.

Riparian vegetation limits macrophyte growth through shading and reduced water temperatures. The extent to which riparian vegetation can shade macrophytes is dependent upon stream width, the height and structure of the riparian canopy, and channel characteristics such as orientation and bank height (Russell *et al*. 1996; Bunn *et al*. 1999a). Loss of riparian vegetation can substantially increase light availability and hence the potential for increased macrophyte abundance, altered species richness and dominance, and loss of shade-tolerant taxa (Davies-Colley 1997; Bunn *et al*. 1998; Bunn *et al*. 1999b; Wilcock *et al*. 2002; Pusey and Arthington 2003). In particular, substantially increased light availability may promote the growth of alien semi-aquatic grasses such as *Urochloa mutica* in the littoral zones of streams and rivers (Bunn *et al*. 1998). The relative change in macrophyte abundance in headwater streams following loss of riparian canopy cover is predicted to be greater than in lower catchment areas, given the naturally lower abundance of macrophytes in forested headwater streams (Vannote *et al*. 1980). Removal or degradation of riparian vegetation allows greater light penetration, higher water temperatures and hence increased rates of autotrophic production (Bunn *et al*. 1999b; Pusey and Arthington 2003). In upstream areas this is associated with a change in species composition from diatoms to filamentous algae and vascular macrophytes (Bunn *et al*. 1999b).

3.2 Aims

This report presents an assessment of the use of aquatic macrophytes as indicators of catchment land use and riparian disturbance in the wet tropics region of north Queensland. Macrophyte assemblages are described in terms of assemblage composition (species presence-absence and abundance) and assemblage metrics (univariate summary statistics of assemblage attributes). We also investigate the influence of hydraulic habitat on macrophyte assemblage patterns observed over catchment land-use and riparian disturbance gradients. For the purposes of this investigation macrophytes are defined as charophytes, mosses, liverworts, pteridophytes and non-woody angiosperms, found within the wetted channel perimeter and identifiable with the naked eye (e.g. Sculthorpe 1967; Jacobs and Wilson 1996). Taxonomy in this report follows Henderson (2002) except where indicated in the text.

3.3 Materials and Methods

Study Sites

Macrophytes were surveyed at 34 sites within the Mulgrave and Russell River catchments of the wet tropics region (Chapter 2). The rationale for site selection is detailed in Chapter 2. Nine sites were surveyed in each of the Little Mulgrave River and Babinda Creek and eight sites were surveyed in each of Behana and Woopen Creeks (Appendix 3.1). The Little Mulgrave River and Behana Creek had generally intact or minimally disturbed riparian zones whereas Woopen and Babinda Creeks had typically highly disturbed riparian zones (Appendix 3.1). Anthropogenic land use within the sub-catchments surveyed included sugar cane farming (predominantly Babinda and Behana Creeks), other crops such as bananas (Woopen Creek) and grazing (Woopen and Babinda Creeks). Behana Creek was the only stream with a flow regime modified by a dam. Each site was 100 m long and included a variety of hydraulic habitats (riffles, runs, pools) as macrophyte species distribution and abundance is often correlated with stream hydraulics (Chambers *et al*. 1991; Englund 1991; Biggs 1996; French and Chambers 1996).

Macrophyte Surveys

Observations of macrophyte assemblage structure were made on 10 equally spaced transects per site. Three 1 m^2 quadrats were placed on each transect. Quadrat size was chosen to maximise the chances of encountering macrophytes in the study systems (Downing and Anderson 1985) and to delineate an appropriate (representative) sampling unit for measurement of hydraulic parameters (see below). As previous experience in the region suggested that most macrophyte growth would be in the stream margins (B. Pusey pers. comm.), the outer quadrats on each transect were placed adjacent to the stream margins and the third quadrat placed at the centre of the transect. The cover (proportion of substratum coverage) of each macrophyte species in each quadrat was recorded using a modified Braun-Blanquet cover scale (Table 3.2). Use of a categorical cover scale reduces operator error and is more suited to rapid stream bioassessments than direct assessment methods such as biomass determination (Haslam 1982).

As a comparison of methods and to ensure that all taxa present at each site were recorded, macrophyte cover was also recorded in a 1-m wide belt transect (at each linear transect) using the Braun-Blanquet cover scale. Macrophytes not present in quadrats or belt transects but observed within the site boundary were recorded as incidental species and cover estimated for the entire site area surveyed using the Braun-Blanquet scale. Macrophytes were identified to the lowest taxonomic level possible in the field and where practical, specimens were sent to the Queensland Herbarium for confirmation of identification.

Table 3.2. Modified Braun-Blanquet cover scale to estimate macrophyte cover in quadrats and belt transects. Based on Braun-Blanquet table presented in Küchler (1967).

Cover Class	Degree of Coverage
7	76-100% of the area
6	51-75% of the area
5	25-50% of the area
4	11-25% of the area
3	6-10% of the area
2	1-5% of the area
1	less than 1% of the area

Catchment characteristics, land use and riparian condition

Catchment characteristics (catchment area upstream of each site, distance of each site to the river mouth, elevation) were determined from GIS (Chapter 2). Macrophyte assemblage structure has been shown to vary with catchment area, distance to mouth and elevation (Suren and Ormerod 1999; Mackay *et al*. 2003). Land use was described in terms of seven broad categories: conservation (including State Forest and National Parks), sugar cane, other cropping, grazing, residential/rural-residential, industrial and storage. Land-use types within each category are summarized in Table 3.3. Sugar cane was separated from other crops as sugar was the dominant crop grown in the region (Russell *et al.* 1996).

Riparian condition was assessed at 38 sites using the assessment protocol of Werren and Arthington (2002) (see Appendices 3.1 and 3.2). Riparian condition was assessed within the same 100 m site used for the macrophyte survey. For those locations where a macrophyte survey had not been conducted (see Appendix 3.1) a representative 100 m site was chosen for the riparian condition assessment. Riparian condition was described in terms of five key components of riparian vegetation structure: the width of the riparian zone, linear continuity, canopy vigour/crown health, the proportion of native and alien species and the extent of indigenous species regeneration (Appendix 3.2). Each component was scored from 1 (poor) to 5 (very good) for each stream bank. The site score was determined as the sum of the scores for each stream bank at each site. The maximum score possible for an individual stream bank was 25, and for an entire site 50. The lowest score possible for a site was 10. Whilst the rapid assessment procedure employed here did not allow for a full species inventory, common species were recorded and samples taken where features (flowers and fruits) aiding identification were available and accessible. Groundcover samples were collected for herbarium verification where invasive and native species could not be readily identified in the field (particularly species of the families Cyperaceae and Poaceae).

Table 3.3. Land-use categories.

Land-use Type	Acronym	Categories Included in Land-use Type
Conservation Areas	CONSERV	1.1.3 National Park
		1.1.4 Natural feature protection
		1.3.0 Conservation, minimal use
		1.3.3 Conservation: remnant native cover
Sugar	SUGAR	3.3.5 Sugar
Other Cropping-Horticulture	OTH_CROP	4.3.0 Irrigated cropping
		4.4.1 Irrigated tree fruits
		5.1.0 Intensive horticulture
Plantation	PLANTAT	3.1.0 Plantation forestry
Grazing	GRAZE	2.1.0 Grazing natural vegetation
		3.2.0 Grazing modified pastures
Residential-Rural Residential	RESID	5.4.0 Residential
		5.4.2 Rural residential
		5.5.3 Recreation and culture
Industrial and Commercial	INDUST	5.3.0 Manufacturing and industrial
		5.5.0 Services
		5.5.1 Commercial services
		5.5.2 Public services
		5.8.0 Mining
		5.9.0 Intensive uses: waste treatment and disposal
Reservoirs	STORAGE	6.2.0 Reservoirs and dams

Physico-chemical parameters

Physico-chemical data were recorded concurrently with macrophyte sampling. Resource availability was characterised in terms of light and nutrient (nitrogen, phosphorus) availability. Riparian canopy cover was measured as a surrogate for light availability. The riparian canopy cover above each quadrat was estimated using a spherical densiometer (Lemmon 1956). Nutrient concentrations were determined by methods outlined in Chapter 2. Dissolved oxygen, conductivity, pH and water temperature were measured *in situ* using Greenspan sensors. These readings were taken at approximately noon to standardise water temperature measurements. Three to five measurements were taken per site. Turbidity was recorded *in situ* with a TPS WP89 data logger and TPS 125192 turbidity probe. Three to five turbidity measurements were recorded per site.

Hydraulic parameters

The wetted width of each transect was measured to the nearest 0.1 m with a tape measure. Average water velocity within each quadrat was recorded at 0.6 times the stream depth

(Gordon *et al*. 1992) with a Swoffer model 2100 flow meter. The depth of each quadrat was recorded to the nearest centimetre with a staff. The substratum composition of each quadrat was visually estimated using a modified Wentworth Scale as the proportion of mud (<0.063 mm diameter), sand (0.063-2 mm), fine gravel (2-16 mm), gravel (16-64 mm), cobble (64-128 mm), rock (128-512 mm) or bedrock (>512 mm) present per quadrat (Gordon *et al*. 1992). The median particle size (d_{50}) was also determined at each site (see Chapter 2). Water slope was measured as the change in relative height of the water surface over the entire 100 m site length with a staff and dumpy level.

Depth and water velocity measurements were used to calculate Reynolds number and Froude number (Gordon *et al*. 1992). Reynolds Number (Re) is the ratio of inertial forces to viscous forces and describes whether flow is laminar (smooth) or turbulent (Gordon *et al*. 1992). It is calculated from the formula

$$\mathrm{Re} = \frac{VL}{v}$$

where V is velocity (ms^{-1}), L is length (m) and v is kinematic viscosity (m^2s^{-1}) (Gordon *et al*. 1992). Mean depth was used as the length measure (L) for calculating Re (Gordon *et al*. 1992). Values of Re less than 500 indicate laminar (smooth) flow and values greater than 2000 indicate turbulent (chaotic) flow (Gordon *et al*. 1992). Reynolds numbers between 500-2000 indicate a transitional zone where flow may be laminar or partly turbulent. Froude Number (Fr), is a useful measure of bulk flow characteristics (Gordon *et al*. 1992). Froude Number was calculated from the formula

$$Fr = \frac{V}{\sqrt{gD}}$$

where V is mean velocity (ms^{-1}), g is acceleration due to gravity (ms^{-2}) and D is hydraulic depth (m). Froude numbers less than 1 indicate slow or tranquil (subcritical) flow, Froude Numbers equal to 1 indicate critical flow and Froude Numbers greater than 1 indicate supercritical (fast or rapid) flows (Gordon *et al*. 1992).

Environmental parameters and acronyms used in this report are summarized in Table 3.4.

Statistical Analysis

A dual approach was used to assess the efficacy of aquatic macrophytes for use as indicators of land-use and riparian disturbance. Firstly, classification and ordination were used to examine spatial patterns in macrophyte assemblage structure within the study area. Secondly, univariate metrics were calculated from species composition and morphological data (see Table 3.1 for justification for selection of metrics) and autoregression used to examine relationships between metrics, land use and water quality. Relationships between macrophyte assemblage metrics and environmental variables were explored initially using Spearmans non-parametric correlation coefficients. Highly intercorrelated variables were excluded from further analyses to reduce collinearity (Tabachnik and Fidell 1989; Graham 2003).

Classification and Ordination

Classification and ordination were used in two ways. Firstly, sites were classified and ordinated using environmental data. The goal of this approach was to identify sites with similar land-use and water quality characteristics and determine whether unique macrophyte assemblages were associated with these habitat types. Prior to analysis environmental data were range standardised using the formula

$$\frac{D_{ij} - D_{\min}}{D_{range}}$$

where D_{ij} is the value of the *i*th row and *j*th column of the data matrix and D_{min} and D_{range} are the minimum value and range respectively (Belbin 1995). The Euclidean dissimilarity measure was used to calculate an association matrix of dissimilarities between sites. The dissimilarity matrix was used to generate an agglomerative hierarchical classification (Unweighted Pair-Group Method Using Arithmetic Averages, UPGMA) with β = -1 (Belbin 1995). An appropriate number of sample groups was determined by inspection of the dendrogram structure and use of the Group Definition (GDEF) function in PATN (Belbin 1995). Kruskal-Wallis tests were used to compare macrophyte assemblage attributes and environmental variables between site groups identified by UPGMA (Zar 1996).

Site groups identified by UPGMA classification were confirmed by ordination of the association matrix using Semi-Strong-Hybrid Multidimensional Scaling (SSHMDS; Belbin 1995). Where possible, ordination stress was held below 0.15 (Belbin 1995) by manipulating the number of dimensions and changing the cut levels and regression techniques used. The ordination was rotated (Varimax rotation) to simplify interpretation. Principal Axis Correlation (PCC) was used to correlate environmental variables with the ordination space. This procedure uses multiple regression to fit attributes to an ordination space as vectors of best fit (Belbin 1995). The significance of correlation coefficients produced by Principal Axis Correlation was tested using a Monte-Carlo procedure (Monte-Carlo Attributes and Ordination procedure in PATN) and 1000 randomisations.

The utility of aquatic macrophyte taxa to discriminate between UPGMA-defined site groups was examined using measures of constancy and fidelity (Belbin 1995). Constancy is the proportion of sites within any group in which a taxon occurs. Fidelity is the capacity of a taxon to predict a site group. A useful bioindicator would therefore occur at a relatively high frequency within a particular site group (high constancy) and would not occur in other site groups (i.e. high fidelity).

Secondly, classification and ordination were used to examine spatial variations in macrophyte assemblage structure across the study area in relation to environmental gradients – that is, sites were ordinated using species presence-absence and cover data. The goal of this approach was to identify unique macrophyte assemblages and the environmental attributes of the sites in which they occurred (i.e. the reverse of the procedure described above). A similar procedure for analysis as described above was used except that the Bray-Curtis dissimilarity measure was used instead of the Euclidean dissimilarity measure, as the Bray-Curtis measure is a more robust association measure for biotic data (Faith *et al*. 1987).

Metrics

Macrophyte assemblage composition data were used to calculate univariate assemblage metrics (Table 3.5). These metrics described key attributes of macrophyte assemblage structure, were predicted to vary over the resource gradients present within the study area (see Table 3.1) and were also considered to be easily implemented by and/or described to non-specialists. Total macrophyte cover per site was determined as the average of the ten transect cover estimates recorded at each site. Average cover values were determined from raw (percentage) cover estimates, and the average value converted to an equivalent Braun-Blanquet score (see Table 3.2). Species richness was calculated as the total number of taxa recorded per site. The growth form of each species recorded was classified as submerged, emergent or floating (based on the position of leaves relative to the water surface), and the percentage of each growth form present was calculated as a proportion of the total number of species present at each site. Finally, the percentage of alien taxa was calculated as a proportion of the total species present at each site. Alien taxa were determined from Henderson (2002).

Table 3.4. Summary of environmental parameters. See text for definitions of individual parameters.

Parameter	Unit	Acronym
Catchment and Land-use		
Catchment area	km^2	CATAREA
Distance to mouth	km	DISTM
Elevation	m.a.s.l.	ELEV
Conservation Areas	%	CONSERV
Sugar Cane	%	SUGAR
Other Cropping-Horticulture	%	OTH_CROP
Grazing	%	GRAZE
Plantation	%	PLANTAT
Residential-Rural Residential	%	RESID
Industrial and Commercial	%	INDUST
Reservoir	%	STORAGE
Riparian Canopy Cover	%	RIPCOV
Riparian Condition Total Score		RIPSCORE
Water Quality		
Dissolved Oxygen	ppm	DO
Conductivity	$\mu S\ cm^{-1}$	COND
pH	pH units	PH
Water Temperature	^oC	TEMP
Turbidity	NTU	TURB
Ammonia	μgL^{-1}	NH3
Oxides of Nitrogen	μgL^{-1}	NOX
Total Nitrogen	μgL^{-1}	TN
Total Phosphorus	μgL^{-1}	TP
Filterable Reactive Phosphorus	μgL^{-1}	FRP
Hydraulic Parameters		
Water Slope	(%)	SLOPE
Width	m	WIDTH
Depth	m	DEPTH
Water Velocity	ms^{-1}	VELOC
Median Particle Size	mm	D50
Substrate Composition	% of quadrat	MUD, SAND, FINEGR, GRAV, COBBLE, ROCK, BEDROCK
Froude Number		FROUDE
Reynolds Number		REYNOLD

Table 3.5. Macrophyte assemblage metrics and their definition.

Metric	Acronym	Definition
Total cover	COVER	Total macrophyte cover expressed as Braun-Blanquet cover score
Species richness	SPECRICH	Total number of individual taxa per site
% Submerged taxa	SUBMERG	Percentage of taxa present with submerged growth form
% Emergent taxa	EMERG	Percentage of taxa present with emergent growth form
% Native taxa	NATIVE	Percentage of taxa present that are native
% Alien taxa	ALIEN	Percentage of taxa present that are alien
% Poaceae	POACEAE	Percentage of taxa present that are grasses

Autoregression models

Relationships between assemblage metrics and land use, water quality and riparian condition were investigated using autoregressive modelling (Lichstein *et al*. 2002). Autoregressive models differ from linear regression models in having an additional term that accounts for autocorrelation. Autocorrelation is the lack of independence between observations (Legendre 1993). For example, sites located close together on the same stream could be expected to be more closely related than sites on adjacent streams. Autocorrelation results in inflated Type I error and violates the assumption of independence of observations for classical methods of statistical analysis (Legendre and Trousellier 1988). Thus erroneous conclusions about species-habitat relationships may result from autocorrelation (Betts *et al*. 2006). Spearman rank correlation coefficients were used first to investigate relationships between predictor variables (i.e., land use and water quality) and macrophyte assemblage metrics. Hierarchical partitioning (MacNally 1996) was then used to determine which of the variables found to be significantly correlated with individual assemblage metrics explained significant independent variation in these metrics. Variables identified by the hierarchical partitioning procedure as explaining significant independent variation in macrophyte assemblage metrics were then used as predictor variables in autoregression models.

Simultaneous autoregressive (SAR) models were fit to assemblage metrics with predictor variables standardised to zero mean and unit variance. SAR model fits were assessed using Nagelkerke's R^2 (Lichstein *et al*. 2002), Akaike's Information Criterion and the Wald statistic (Quinn and Keough 2002). Nagelkerke's R^2 was calculated from the formula

$$1-e^{\left(\left(\frac{-2}{n}\right)(LLfull-LLnull)\right)}$$

where *LLfull* is the log-likelihood for the full model and *LLnull* is the log-likelihood of the null model (Lichstein *et al*. 2002). Akaike's Information Criterion (AIC) adjusts the deviance for a given model based on the number of predictor variables included in the model. The AIC for the SAR model was compared with the AIC for an equivalent linear model. Lower values for AIC indicate better model fits (Quinn and Keough 2002).

To account for potential variation in assemblage metrics explained by hydraulic parameters, ordinary least squares (OLS) regression models were fitted to the residuals of the individual SAR models. Hierarchical partitioning was used to select hydraulic parameters that explained significant independent variation in the residuals of each SAR model. Variables identified as significant were included as predictor variables in OLS models.

Spatial regression requires the delineation of neighbours, often on the basis of distance between sampling points. Preliminary analysis using different neighbour definitions showed that R^2 and model coefficients for SAR models were sensitive to the distance used to define neighbours (although the significance of individual model parameters changed little). Changes were not consistent between metrics, suggesting that different spatial patterns were associated with each metric. Two sets of SAR models were therefore fitted to data for each metric. For the first set of models neighbour distance was set as the maximum distance between any pair of sites (approximately 1.5 km). This criterion emphasised spatial patterns acting at relatively small spatial scales. For the second set of models neighbour distance was set at the maximum distance between any pair of sets (approximately 40 km). This criterion emphasised spatial patterns acting at broader spatial scales and essentially identified each site as having 33 neighbours.

Hierarchical partitioning, SAR and OLS regression models were fit using packages available in *R* (Ihaka and Gentleman 1996). SAR models were fit using the *spdep* package version 0.3-22 (Bivand 2006), OLS models were fit using the *Design* package version 2.0-12 (Harrell 2005) and hierarchical partitioning carried out in the *hier.part* package version 1.0-1 (Walsh and MacNally 2005).

3.4 Results

Assemblage metrics were significantly correlated with riparian condition and riparian canopy cover (Appendix 3.3). NATIVE and SUB were positively correlated with riparian condition, possibly because of the presence of submerged bryophytes in shaded headwater reaches. The remaining metrics were negatively correlated with riparian condition and canopy cover. Surprisingly, macrophyte metrics were not well correlated with catchment land-use descriptors. Most metrics were significantly correlated with CONSERV, GRAZE and OTH_CROPS, but only COVER was significantly correlated with SUGAR (the dominant anthropogenic land use in the study area). SPECRICH was the only metric significantly correlated with RESID, INDUST or STORAGE. COVER was positively correlated with all agricultural land-use descriptors (i.e. SUGAR, OTH_CROP, GRAZE, PLANTAT).

With the exception of COVER and ALIEN, macrophyte metrics were poorly correlated with water quality parameters. COVER was positively correlated with TN and NO_x but negatively correlated with TP (Appendix 3.3). ALIEN was positively correlated with TEMP, TN and NO_x.

Land-use measures (particularly RIPCOV and RIPSCORE) were correlated with several water quality parameters. RIPCOV and RIPSCORE were significantly negatively correlated with agricultural land uses (i.e. GRAZE, PLANTAT, SUGAR, OTH_CROP) but were not correlated with RESID, INDUST and STORAGE. RIPSCORE and RIPCOV were also positively correlated with CONSERV. CONSERV was positively correlated with FRP and strongly negatively correlated with TN and NO_x. SUGAR, GRAZE and OTH_CROP were positively correlated with TN and NO_x (Appendix 3.3).

Riparian condition

Riparian condition scores were highest for sites in the Little Mulgrave River and Behana Creek but headwater sites in all sub-catchments had good riparian condition scores (Figure 3.2). Sites in Little Mulgrave River and Behana Creek had high proportions of native species and good linear continuity of riparian vegetation (Appendix 3.1). Babinda and Woopen Creeks had generally poor riparian condition and often both stream banks were affected to a similar degree (Figure 3.1). Nonetheless, it is evident that even for the Little Mulgrave River and Behana Creek, localised riparian degradation has occurred, although often limited to a single stream bank (Figure 3.1).

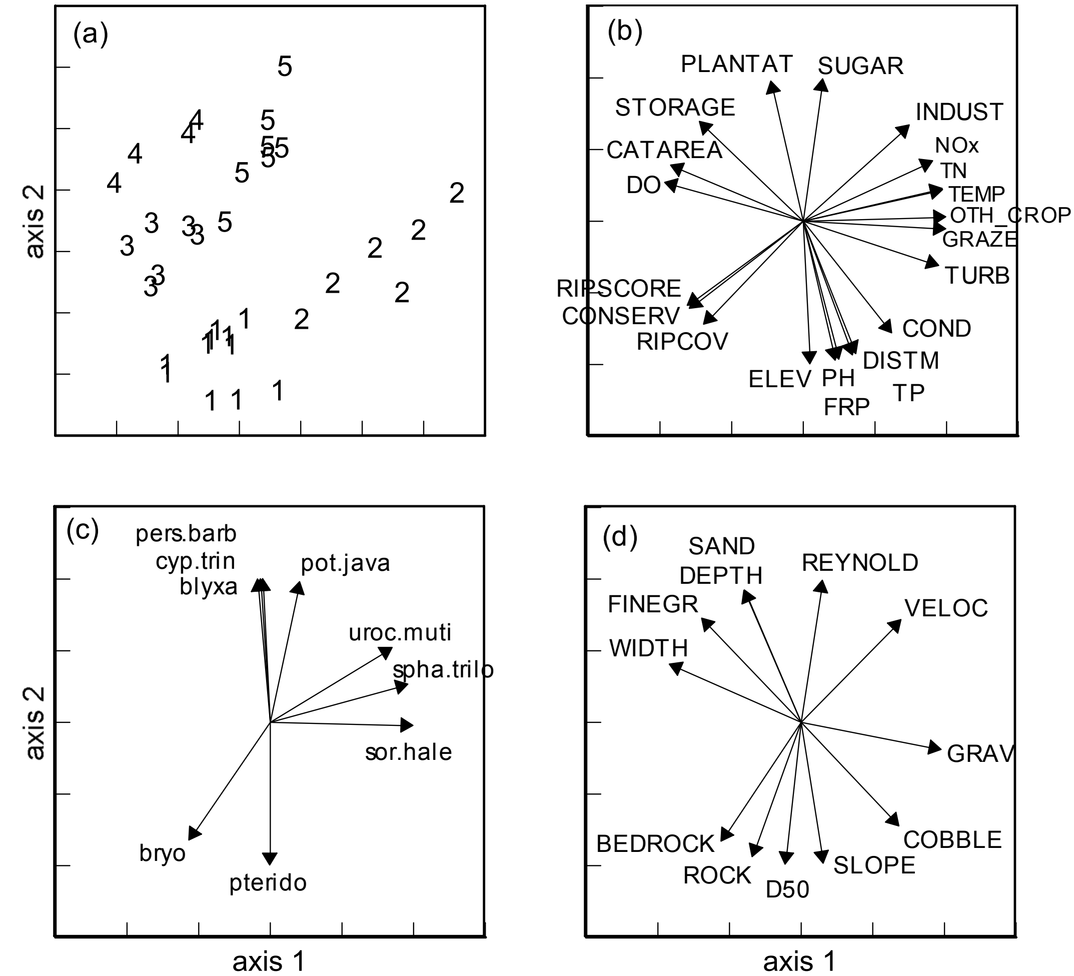

Figure 3.1. Ordination of sites based on land-use and water quality parameters. Stress 0.141, interval regression, 2 dimensions. (a) Location of sites in ordination space. (b) Directions of correlation of significant environmental attributes (P<0.05) with the ordination. (c) Directions of correlation of significant macrophyte taxa (P<0.05) with the ordination. Species acronyms: pers.barb *Persicaria barbata*; cyp.trin *Cyprinus trinervis*; blyxa *Blyxa* spp.; pot.java *Potamogeton javanicus*; uroc.mutic *Urochloa mutica*; spha.trilo *Sphagneticola trilobata*; sor.hale *Sorghum halepense*; pterido Pteridophyta; bryo Bryophyta. (d) Directions of correlation of significant hydraulic parameters (P<0.05) with the ordination. See Table 3.4 for variable acronyms.

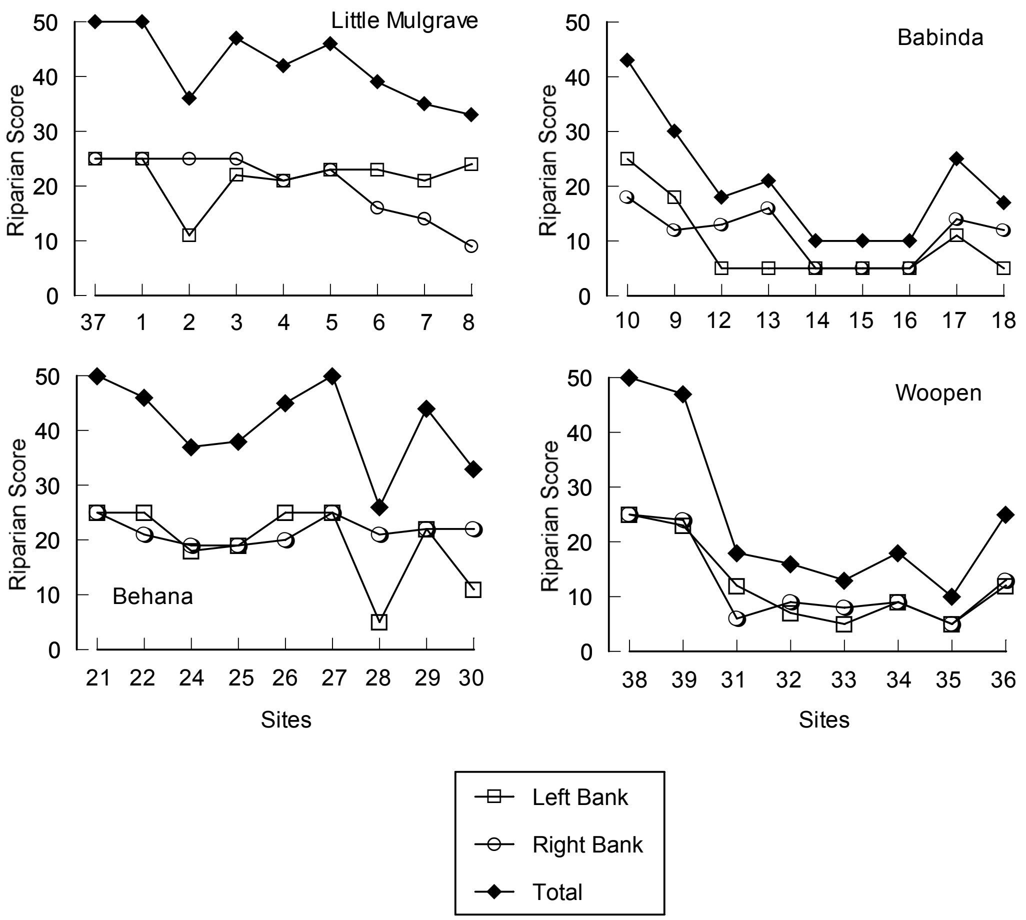

Figure 3.2. Riparian condition scores recorded for individual sites in the four sub-catchments surveyed. Sites on each *x*-axis are ordered from highest to lowest elevation.

A list of riparian species verified by the Queensland Herbarium is presented in Appendix 3.4. Common riparian canopy species include the watergum (*Tristaniopsis exiliflora*), brown laurel (*Cryptocarya triplinervis*), quandong (*Elaeocarpus* spp.), weeping bottlebrush (*Callistemon viminalis*), lilly pilly (*Syzygium* spp.) and golden penda (*Xanthostemon chrysanthus*). Leichhardt Tree (*Nauclea orientalis*) and bleeding hearts (*Homalanthus novo-guineensis*) were common in disturbed or open riparian zones. Invasive species such as guava (*Psidium guava*) were common in disturbed areas where native canopy species were sparse. Vine species included the invasive laurel vines (*Thunbergia* spp.*)*, mile-a-minute (*Mikania micrantha*) and glycine (*Neonotonia wightii*).

Monocotyledonous ground cover was represented by a diversity of taxa, including a number of invasive alien species (Appendix 3.4) such as para grass (*Urochloa mutica*) and guinea grass (*Megathyrsus maximus*). Grasses within the riparian zone were abundant and relatively diverse with 19 species confirmed from herbarium samples, including one species (*Centotheca philippinensis*) recorded as rare under the Queensland Nature Conservation (Wildlife) Regulation (1994) and vulnerable under the Endangered Species Protection Act (1992). Cyperaceae were relatively diverse (14 species) and included species with a submerged habit (e.g. *Cyperus trinervis*) and emergent habit (e.g. *C. odoratus*). A number of introduced Cyperaceae were also recorded. Navua sedge (*C. aromaticus*) was commonly recorded and, less frequently, umbrella sedge (*C. involucratus*).

Dicotyledonous ground cover included relatively few taxa. Species of the families Asteraceae, Polygonaceae and Onagraceae were the most frequently recorded dicotyledons. Singapore daisy (*Sphagneticola trilobata,* Asteraceae), a Class 3 declared weed species in Queensland, was very common. Other introduced species commonly recorded were knob weed (*Hyptis capitata*) and snake weed (*Stachytarpheta cayennensis*). Other notable recordings included giant bramble (*Rubus alceifolius*) and *Hygrophila* sp., although in the case of the latter, identification could not be confirmed because of the lack of fertile material.

Aquatic macrophytes

Forty-four macrophyte taxa were recorded from the study area (Table 3.6). The number of taxa present is likely to be higher than this as individual fern species could not be positively identified because of the lack of fertile material. However, there were probably at least four species of ferns present, based on frond morphology. A number of additional taxa could not be positively identified due to the lack of fertile material. The presence of *Vallisneria nana* R.Br. could not be confirmed owing to the lack of fertile material, however the occurrence of the morphologically similar *Blyxa* sp. (Hydrocharitaceae) was confirmed by fertile material. *V. nana* has been previously recorded from the Mulgrave River (Russell *et al*. 1996). *Hygrophila* sp. could not be identified to species and this has implications for the calculation of some metrics, particularly SPECRICH, NATIVE and ALIEN. Three species of *Hygrophila* are known from Queensland (Henderson 2002): *H. angustifolia* (native), *H. costata* (alien) and *H. triflora* (alien). Similarly, some grasses (Poaceae) and sedges (Cyperaceae) could not be conclusively identified because of the lack of fertile material, and these could have included native and/or alien taxa.

Approximately one third of the aquatic macrophyte taxa identified were Poaceae or Cyperaceae. Thus emergent taxa were the dominant morphological group, representing approximately 77% of the taxa recorded from the study area. Submerged growth forms were dominated by bryophytes, *Cladopus* (=*Torrenticola*) *queenslandicus* (Domin) C.D.K. Cook (Podestemaceae) and *Blyxa* sp. (Hydrocharitaceae). *C. queenslandicus* is a declared Rare species under the Queensland Conservation (Wildlife) Regulation (1994). *Cyperus trinervis* also occurred occasionally as a submerged form. Floating taxa and charophytes were not recorded.

In terms of frequency of occurrence the five most dominant taxa recorded from the study area were para grass [*Urochloa mutica* (Forssk.) T.O. Nguyen], Singapore daisy [*Sphagneticola trilobata* (L.) Pruski], *Persicaria barbata* (L.) H. Hara, mosses (Bryophyta) and *Cyperus trinervis* R. Br. Individually these taxa occurred at over 30% of the sites surveyed. *U. mutica* and *S. trilobata* are alien species with widespread distributions in Queensland. *S. trilobata* is the only declared pest plant (Class 3, Land Protection 2006) recorded from the study area during this survey.

Multivariate patterns in land use and water quality

Five distinct site groups were identified from classification and ordination of land-use and water quality data (Figure 3.2a; Appendix 3.5). Groups 1 and 2 were separated from groups 3-5 in ordination space by differences in land use, conductivity and phosphorus concentrations (Figure 3.2a,b). Furthermore, groups 1, 3 and 4 were separated from groups 2 and 5 along a gradient of land-use and riparian condition, with groups 1, 3 and 4 representing better riparian condition than groups 2 and 5.

Table 3.6. Frequency of occurrence (% of sites) of aquatic macrophyte taxa within the study area. Alien taxa indicated with an asterisk (*). Growth form code: EM emergent; SUB submerged.

Family	Taxon	Growth Form	Frequency of Occur.
Acanthaceae	*Hygrophila angustifolia* R.Br.	EM	8.8
Alismataceae	*Sagittaria* sp.?	EM	2.9
Araceae	*Colocasia esculenta* (L.) Schott*	EM	2.9
Asteraceae	*Ageratum conyzoides* L. subsp *conyzoides*	EM	5.9
	Sphagneticola (*Wedelia*) *trilobata* (L.) Pruski*	EM	55.9
	Unidentified Asteraceae	EM	2.9
Apiaceae	*Hydrocotyle* sp. 1	EM	2.9
	Hydrocotyle sp. 2	EM	2.9
Bryophyta		SUB	38.2
Caryophyllaceae	*Drymaria cordata* (L.) Willd. ex Roem & Schult.	EM	5.9
Commelinaceae	*Commelina* spp.	EM	17.6
Cyperaceae	*Cyperus aquatilis* R.Br.	EM	8.8
	Cyperus aromaticus (Ridl.) Mattf. & Kuek.*	EM	14.7
	Cyperus odoratus L.	EM	5.9
	Cyperus involucratus Rottb.*	EM	11.8
	Cyperus polystachyos Rottb.	EM	2.9
	Cyperus sphacelatus Rottb.	EM	2.9
	Cyperus trinervis R.Br.	EM/SUB	38.2
	Schoenoplectus mucronatus (L.) Palla ex J.Kearn.	EM	5.9
	Unidentified Cyperaceae	EM	23.5
Elatinaceae	*Elatine gratiolides* A.Cunn.	SUB	14.7
Haloragaceae	*Myriophyllum* sp.	SUB	11.8
Hepatophyta		SUB	5.9
Hydrocharitaceae	*Blyxa* sp.	SUB	29.4
	Hydrilla verticillata (L.f.) Royle	SUB	14.7
	Vallisneria nana R. Br.	SUB	5.9
	Unidentified Hydrocharitaceae	SUB	11.8
Lomandraceae	*Lomandra* sp.	EM	2.9
Malvaceae	Unidentified Malvaceae	EM	2.9
Poaceae	*Arundo donax* L. var. *donax**	EM	5.9
	Axonopus fissifolius (Raddi) Kuhlm.*	EM	2.9
	Chrysopogon filipes (Benth.) Reeder	EM	2.9
	Cyrtococcum oxyphyllum (Hochst.) ex Steud.) Stapf	EM	14.7
	Megathyrsus maximus (Jacq.) B.K.Simon & S.W.L. Jacobs	EM	11.8
	Pennisetum pupureum Schumach.*	EM	11.8
	Sacciolepis indica (L.) Chase	EM	2.9
	Sorghum halepense (L.) Pers.*	EM	5.9
	Urochloa mutica (Forssk.) T.O. Nguyen *	EM	58.8
	Unidentified Poaceae	EM	11.8
Philydraceae	*Philydrum lanuginosum* Banks & Sol. Ex Gaertn.	EM	2.9
Podestemaceae	*Cladopus queenslandicus* (Domin) C.D.K.Cook	SUB	14.7
Polygonaceae	*Persicaria barbata* (L.) H.Hara	EM	50.0
	Persicaria lapathifolia (L.) Gray*	EM	8.8
	Persicaria strigosa (R.Br.) H.Gross	EM	2.9
Potamogetonaceae	*Potamogeton javanicus* Hassk.	SUB	8.8
	Potamogeton sp.	SUB	2.9
Pteridophyta		EM	23.5
UNKNOWN			14.7

Group 1 consisted of 11 sites in the Little Mulgrave River and upper Woopen Creek (Figure 3.2a). These sites were located at higher elevations and had a large proportion of the upstream catchment area as conservation areas (i.e., National Parks or State Forests) (Figure 3.2b; Table 3.7). Group 1 sites therefore had high riparian canopy cover and high riparian condition scores (Table 3.7). NO_x concentrations were relatively low but FRP and TP concentrations were relatively high when compared with other site groups (Table 3.7). Group 1 sites were dominated by mosses and ferns (Figure 3.2c). Group 2 included six sites from lower Woopen Creek characterised by poor riparian condition and high proportions of grazing and cropping in the upstream catchment area (Figure 3.2a,b). TN and NO_x concentrations were relatively high and DO was relatively low. Mean water temperatures were approximately 1.5^{o}C higher than at the remaining site groups. Macrophytes associated with this site group included the alien species *U. mutica*, *S. trilobata* and *S. halepense* (Figure 3.2c).

Group 3 included six sites from upper Babinda Creek and upper Behana Creek. These sites had high scores for riparian condition, relatively low conductivity and very low NO_x concentrations (Figure 3.2a,b; Table 3.7). The dominant anthropogenic land use in this group was the cultivation of sugar cane but it constituted a very small proportion of the total catchment area (Table 3.7). These sites were dominated by the emergent species *Persicaria barbata* and *Cyperus trinervis* and the submerged *Blyxa* sp. and *Potamogeton javanicus* (Figure 3.2c). Group 4 was similar to group 3 but included sites in lower Behana Creek (Appendix 3.5; Figure 3.2a). Group 4 differed from group 3 in having a greater proportion of the land use consisting of sugar cane production and also higher concentrations of nitrogenous compounds.

Groups 4 and 5 included low elevation sites with a relatively high proportion (>5%) of the catchment land use being sugar cane cultivation (Table 3.7). Group 4 included sites in lower Behana Creek. Despite the relatively high proportion of sugar cane farming in this group riparian condition was high (mean score 37/50). Group 4 was the only group that included sites affected by flow regulation. Group 5 included seven sites in lower Babinda Creek. These sites had poor riparian condition (mean score 16), low pH (mean 5.76), high TN and NO_x concentrations but moderate TP and FRP concentrations (Table 3.7).

Constancy values for taxa significantly correlated with the ordination (Table 3.7) show that no single taxon had high fidelity for a single site group; most taxa occurred at relatively high frequencies in two or more site groups. However, bryophytes had high fidelity in that they were good indicators of site groups representing relatively pristine sites (groups 1 and 3). Bryophytes occurred in 82% of group 1 sites and 50% of group 3 sites (these groups had higher riparian condition scores and relatively high proportion of the catchment area as CONSERV). *P. barbata* had moderate fidelity in that it was indicative of sites with moderate to poor riparian condition (groups 4 and 5). However, para grass and Singapore daisy, both alien taxa, occurred at relatively high frequencies within three or more site groups (Table 3.7).

Table 3.7. Environmental data (mean ± standard error) for site groups identified by UPGMA classification of sites based on catchment land-use and water quality data. Only parameters identified as being significantly different are shown (Kruskal-Wallis non-parametric one-way ANOVA and Bonferroni adjusted significance levels). See Tables 3.2 and 3.3 for definition of parameters. Constancy values shown in brackets (where an individual taxon occurred in at least 75% of sites in any group).

Environmental Parameters	**Group 1 (*n*=11)**	**Group 2 (*n*=6)**	**Group 3 (*n*=6)**	**Group 4 (*n*=4)**	**Group 5 (*n*=7)**
CATAREA (km^2)	73 ± 12	14 ± 3	56 ± 6	94 ± 3	68 ± 4
DMOUTH (km)	48 ± 1	44 ± 1	31 ± 2	25 ± 1	28 ± 2
ELEV (m.a.s.l)	50 ± 8	34 ± 5	26 ± 6	0 ± 0	9 ± 3
RIPCOV (%)	82 ± 5	17 ± 5	61 ± 10	59 ± 18	11 ± 4
RIPSCORE	43 ± 2	17 ± 2	41 ± 3	37 ± 5	16 ± 2
COND ($\mu S\ cm^{-1}$)	54.40 ± 1.40	53.53 ± 0.61	24.87 ± 1.51	30.42 ± 1.02	28.71 ± 0.83
PH	7.03 ± 0.05	6.68 ± 0.08	6.20 ± 0.08	5.82 ± 0.05	5.76 ± 0.09
TN (μgL^{-1})	149.4 ± 14.4	220.9 ± 17.5	97.2 ± 16.5	162.8 ± 18.5	198.7 ± 8.3
NH3 (μgL^{-1})	4.6 ± 0.4	3.7 ± 0.3	3.3 ± 0.6	7.5 ± 0.5	3.2 ± 0.3
NO_x (μgL^{-1})	49.8 ± 3.9	128.9 ± 13.2	21.5 ± 4.2	78.8 ± 20.4	102.9 ± 15.1
TP (μgL^{-1})	26.8 ± 1.6	21.6 ± 1.7	12.2 ± 1.1	14.8 ± 0.8	16.4 ± 2.0
FRP (μgL^{-1})	12.6 ± 0.6	8.1 ± 1.2	3.3 ± 0.6	3.0 ± 0.4	3.8 ± 0.5
CONSERV (%)	97.8 ± 0.7	84.2 ± 3.8	98.5 ± 0.7	88.0 ± 1.9	88.9 ± 2.3
GRAZE (%)	0.7 ± 0.7	6.0 ± 1.2	0.2 ± 0.1	0 ± 0	1.1 ± 0.1
PLANTAT (%)	0 ± 0	0 ± 0	0.09 ± 0.06	0 ± 0	0.23 ± 0.01
SUGAR (%)	0.9 ± 0.3	3.9 ± 1.6	1.2 ± 0.8	11.7 ± 1.9	7.8 ± 1.8
OTH_CROP (%)	0.10 ± 0.03	5.76 ± 1.29	0.03 ± 0.03	0 ± 0	0.71 ± 0.03
RESID (%)	0.44 ± 0.14	0 ± 0	0.02 ± 0.01	0.22 ± 0.03	0.46 ± 0.22
STORAGE (%)	0 ± 0	0 ± 0	0 ± 0	0.02 ± 0.002	0 ± 0
Assemblage Attributes					
COVER	1.9 ± 0.2	4.0 ± 0.3	2.8 ± 0.4	2.3 ± 0.6	3.6 ± 0.4
POACEAE	5.1 ± 2.3	42.6 ± 8.2	23.7 ± 8.2	16.0 ± 5.6	22.5 ± 3.5
Blyxa sp.	0 ± 0	0.3 ± 0.3 (17)	0.2 ± 0.2 (17)	1.3 ± 0.3 (100)	0.6 ± 0.2 (57)
Bryophyta	1.2 ± 0.2 (82)	0 ± 0	0.8 ± 0.4 (50)	0 ± 0	0.1 ± 0.1 (14)
Persicaria barbata	0.3 ± 0.2 (18)	0.5 ± 0.2 (50)	0.3 ± 0.3 (17)	1.3 ± 0.3 (100)	1.3 ± 0.2 (100)
Sphagneticola trilobata	0.5 ± 0.2 (45)	1.8 ± 0.4 (83)	0 ± 0	0.5 ± 0.3 (50)	1.4 ± 0.3 (100)
Urochloa mutica	0.2 ± 0.1 (18)	3.5 ± 0.3 (100)	0.8 ± 0.4 (50)	1.0 ± 0.7 (50)	2.7 ± 0.3 (100)

Multivariate patterns in macrophyte assemblage structure

Four groups were identified by classification of sites based on macrophyte presence-absence data (Figure 3.3a; Appendix 3.6). Group 1 was characterised by the presence of mosses and *Cladopus* (=*Torrenticola*) *queenslandicus* (Figure 3.3a, b). These sites included site 1 in the upper Little Mulgrave River, sites 38 and 39 in upper Woopen Creek and sites 21 and 22 in upper Behana Creek. Collectively these sites had high scores for riparian condition and a high proportion of conservation land use (Table 3.8; Figure 3.3c,d). Water quality was characterised by low nitrogen (as indicated by TN and NO_x) but moderate phosphorus (TP and FRP) concentrations. Substrates were also very coarse, dominated by rock and bedrock. This assemblage occurred in areas of relatively low water velocity.

Group 2 consisted of eight sites from the Little Mulgrave River (Appendix 3.6). These sites were characterised by *Cyperus involucratus* (alien) and *C. aquatilis* (native), *Hydrocotyle* sp., submerged vascular macrophytes (primarily *H. verticillata*) and ferns. Riparian condition was moderate. These sites also had relatively high areas of conservation land use and low proportions of sugar cane farming. Water quality was characterised by high concentrations of TP and FRP and also high conductivity and pH (Table 3.8). Water velocities were higher than for group 1 sites. Substrates were characterised by a higher proportion of cobbles than group 1 sites.

Groups 3 and 4 were characterised by the presence of the alien species *U. mutica* and *S. trilobata* and the native species *Blyxa* sp., *C. trinervis* and *P. barbata*. These groups included sites with relatively lower riparian condition scores and lower areas of conservation land use compared with groups 1 and 2 (Table 3.8). Group 3 consisted of sites in Behana and Babinda Creeks whereas group 4 consisted of sites in Behana, Babinda and Woopen Creeks (Appendix 3.6). These sites were associated with sugar cane farming in low elevation areas. *Blyxa* sp. and *P. barbata* were associated with sandy substrata and moderate water velocities. *U. mutica* and *S. trilobata* were associated with high water velocities but this is probably due to their occurrence in marginal areas of fast flowing sites, rather than direct utilisation of fast flowing habitats. *U. mutica* and *S. trilobata* were also associated with high concentrations of TN and NO_x.

The grass *Cyrtococcum oxyphyllum* and Pteridophyta (ferns) had high fidelity as group indicators (Table 3.8). Each of these taxa had a very high frequency of occurrence in sites within a single site group. *Bryophyta* and *U. mutica* were good indicators of sites that were relatively pristine (groups 1, 2) and disturbed (groups 3, 4) respectively (Table 3.8).

Ordination of Braun-Blanquet cover data produced similar patterns to those found in the presence-absence ordination although some changes in group membership occurred (compare Figures 3.3 and 3.4 and Appendices 3.6 and 3.7). As with the macrophyte presence-absence ordination, groups 1 and 2 represented relatively pristine sites and groups 3 and 4 represented relatively disturbed sites. It is therefore evident that at the assemblage scale presence-absence data provides as much information as Braun-Blanquet cover data. Group 1 was still characterised by bryophytes and *C. queenslandicus*, but unlike the presence-absence classification, group 1 included two additional sites from the Little Mulgrave River (sites 6 and 37). These sites had been classified with group 2 in the presence-absence classification (Appendix 3.6). Group 2 consisted of six sites from the Little Mulgrave River that had relatively high cover values for *Cyperus involucratus* (alien), *C. aquatilis* (native), submerged vascular macrophytes (*Myriophyllum* sp. and *H. verticillata*) and ferns (Figure 3.4a,b). Groups 3 and 4 were characterised by the presence of the alien species *U. mutica* and *S. trilobata* and the native species *Blyxa* sp., *C. trinervis* and *P. barbata* and like the presence-absence ordination, included sites from Babinda, Behana and Woopen Creeks.

Table 3.8. Attributes of groups identified by UPGMA classification of macrophyte presence-absence data. Values for environmental data are the mean ± standard error. Only parameters identified as being significantly different are shown (Kruskal-Wallis non-parametric one-way ANOVA and Bonferroni adjusted significance levels). See Tables 3.2 and 3.3 for definition of parameters. Constancy values shown in brackets (where an individual taxon occurred in at least 75% of sites in any group).

Taxon	**Group 1 (*n*=5)**	**Group 2 (*n*=8)**	**Group 3 (*n*=4)**	**Group 4 (*n*=17)**
Bryophyta	100	75	0	12
Cyrtococcum oxyphyllum	0	0	100	6
Persicaria barbata	0	25	50	76
Pteridophyta	0	88	0	6
Sphagneticola trilobata	0	63	0	82
Urochloa mutica	0	25	50	94
Land-use and Water Quality Parameters				
CATAREA (km^2)	47.8 ± 13.6	91.2 ± 5.7	54.5 ± 16.6	55.0 ± 7.2
DMOUTH (km)	39.9 ± 4.9	47.9 ± 0.9	32.5 ± 4.6	32.6 ± 2.0
ELEV (m)	56.5 ± 13.1	39.6 ± 7.3	22.0 ± 11.2	15.9 ± 3.4
RIPCOV (%)	92 ± 2	77 ± 5	42 ± 10	26 ± 6
RIPSCORE	49 ± 1	41 ± 2	30 ± 4	23 ± 3
ROCK (%)	34 ± 6	25 ± 2	7 ± 3	5 ± 1
BEDROCK (%)	14 ± 5	2 ± 1	0 ± 0	1 ± 1
COND ($\mu S\ cm^{-1}$)	38.55 ± 6.42	55.31 ± 1.49	32.79 ± 4.47	35.86 ± 2.84
PH	6.65 ± 0.19	7.03 ± 0.06	6.30 ± 0.23	6.07 ± 0.10
FRP (μgL^{-1})	8.38 ± 1.74	12.81 ± 0.70	5.50 ± 2.19	4.41 ± 0.53
CONSERV (%)	99.96 ± 0.04	97.93 ± 0.53	93.80 ± 2.72	87.94 ± 1.78
GRAZE (%)	0 ± 0	0 ± 0	0.90 ± 0.64	2.43 ± 0.75
SUGAR (%)	0.04 ± 0.04	1.29 ± 0.35	4.81 ± 3.07	6.63 ± 1.11
Hydraulic Parameters				
REYNOLD	49276 ± 10105	76574 ± 10910	103478 ± 21011	152223 ± 23366
D50 (mm)	111 ± 13	112 ± 7	58 ± 22	49 ± 8
MUD (%)	0 ± 0	0.3 ± 0.2	3 ± 1	3 ± 1
ROCK (%)	29 ± 6	25 ± 2	7 ± 4	5 ± 2
BEDROCK (%)	11 ± 5	2 ± 1	0 ± 0	1 ± 1

Habitat characteristics associated with each of the four site groups identified from the classification of Braun-Blanquet cover data were similar to those identified by the classification of presence-absence data, although small differences in the loadings of individual environmental parameters on ordination axes were evident (compare Figures 3.3 and 3.4). Constancy values for individual taxa were also similar for each analysis (compare Tables 3.8 and 3.9).

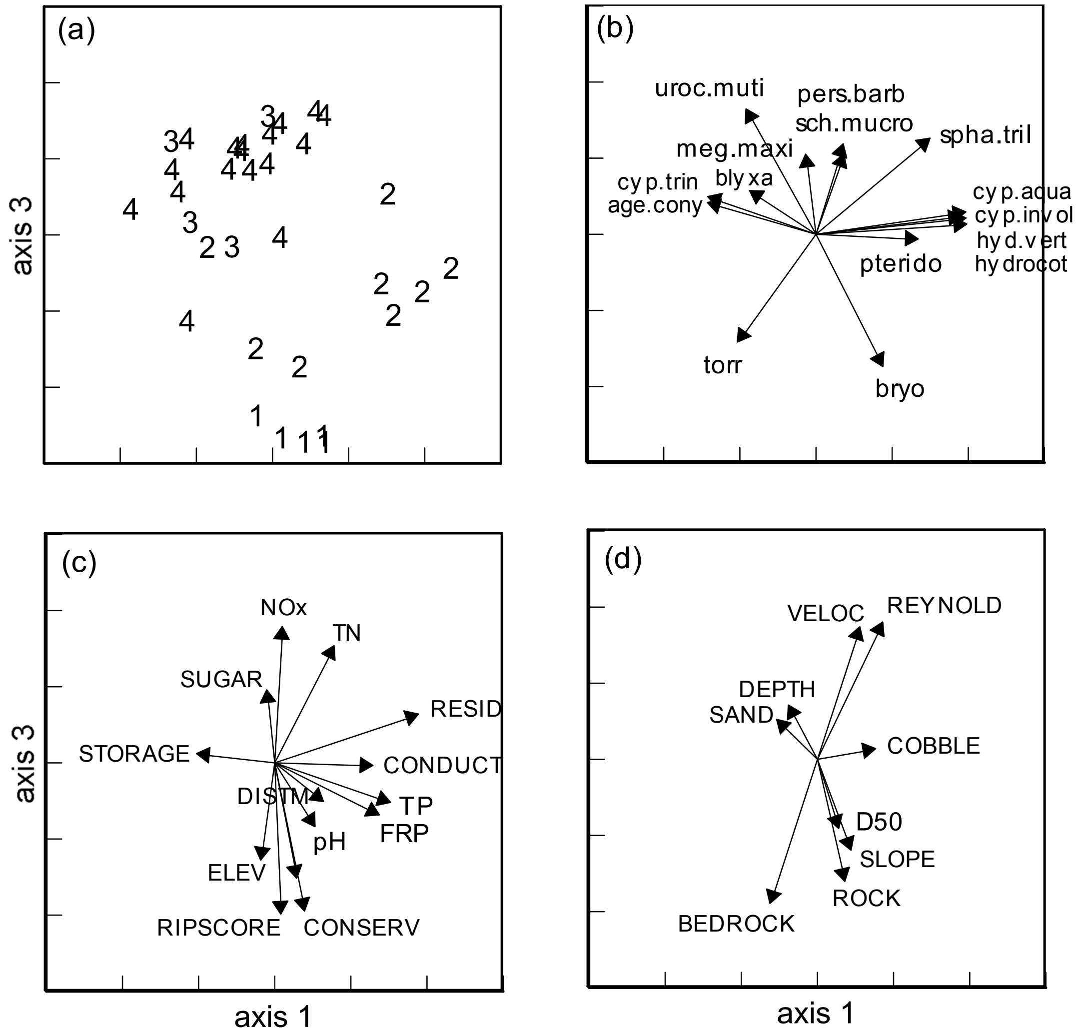

Figure 3.3. Ordination of sites based on species presence-absence data. Interval regression, stress 0.150, three dimensions. (a) location of sites in 2 dimensional ordination space. (b) Directions of significant correlations for macrophyte taxa (P<0.05) with the ordination. (c) Directions of significant correlations for land-use and water quality parameters (P<0.05) with the ordination. (d) Directions of significant correlations for hydraulic parameters (P<0.05) with the ordination. Species acronyms: pers.barb *Persicaria barbata*; cyp.trin *Cyprinus trinervis*; blyxa *Blyxa* spp.; pot.java *Potamogeton javanicus*; uroc.mutic *Urochloa mutica*; spha.trilo *Sphagneticola trilobata*; pterido Pteridophyta; bryo Bryophyta.

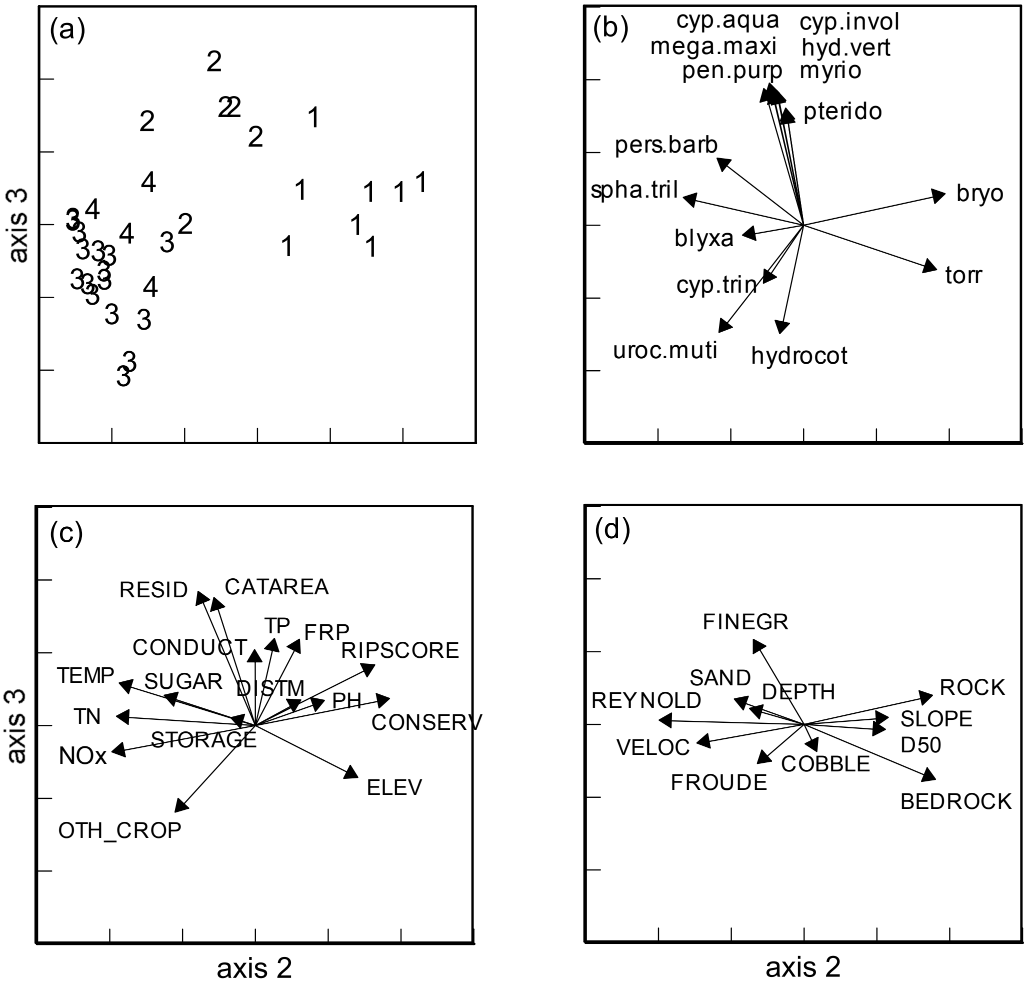

Figure 3.4. Ordination of sites based on species Braun-Blanquet cover scores. Interval regression, stress 0.142, three dimensions. (a) location of sites in 2 dimension ordination space. (b) Directions of correlation of significant macrophyte taxa (P<0.05) with the ordination. (c) Directions of correlation of significant land-use and water quality attributes (P<0.05) with the ordination. (d) Directions of correlation of significant hydraulic attributes (P<0.05) with the ordination.

Table 3.9. Attributes of groups identified by UPGMA classification of macrophyte Braun-Blanquet cover data. Only parameters identified as being significantly different are shown (Kruskal-Wallis non-parametric one-way ANOVA and Bonferroni adjusted significance levels). See Tables 3.3 and 3.4 for definition of parameters.

Taxon	**Group 1 (*n* = 8)**	**Group 2 (*n* = 6)**	**Group 3 (*n* = 16)**	**Group 4 (*n* = 4)**
Blyxa sp.	0 ± 0	0 ± 0	0.5 ± 0.15	1.0 ± 0.35
Bryophyta	1.75 ± 0.15 (100)	0.67 ± 0.19 (67)	0.06 ± 0.06 (6)	0 ± 0 (0)
Cladopus queenslandicus	1.13 ± 0.33	0 ± 0	0 ± 0	0 ± 0
Cyperus aquatilis	0 ± 0	0.5 ± 0.20	0 ± 0	0 ± 0
Cyperus involucratus	0 ± 0	0.8 ± 0.28	0 ± 0	0 ± 0
Persicaria barbata	0.12 ± 0.17 (13)	0.33 ± 0.30 (17)	0.81 ± 0.16 (69)	1.5 ± 0.25 (100)
Pteridophyta	0.25 ± 0.15 (25)	1.00 ± 0.24 (83)	0.06 ± 0.06 (6)	0 ± 0 (0)
Sphagneticola trilobata	0 ± 0 (0)	0.83 ± 0.15 (83)	1.38 ± 0.23 (81)	0.25 ± 0.31 (25)
Urochloa mutica	0.1 ± 0.17 (13)	0.3 ± 0.19 (33)	2.94 ± 0.21 (100)	0.25 ± 0.22 (25)
Land-use and Water Quality Parameters				
CATAREA (km^2)	50.11 ± 11.53	93.32 ± 5.89	47.79 ± 7.27	87.70 ± 5.54
DMOUTH (km)	42.7 ± 3.1	47.4 ± 0.8	34.3 ± 2.1	24.8 ± 0.6
ELEV (m)	55.5 ± 8.9	34.7 ± 5.2	19.8 ± 3.8	0.5 ± 0.4
RIPCOV (%)	85 ± 5	75 ± 7	16 ± 3	73 ± 6
RIPSCORE	47 ± 1	40 ± 2	19 ± 2	40 ± 2
PH	6.75 ± 0.13	7.04 ± 0.08	6.15 ± 0.12	5.89 ± 0.04
TN (μgL^{-1})	114.8 ± 15.3	154.7 ± 22.5	194.3 ± 11.1	159.5 ± 16.7
FRP (μgL^{-1})	9.00 ± 1.44	13.17 ± 0.84	5.19 ± 0.73	3.00 ± 0.35
CONSERV (%)	98.5 ± 0.9	97.8 ± 0.6	88.2 ± 1.9	90.0 ± 2.3
GRAZE (%)	1.1 ± 0.9	0 ± 0	2.8 ± 0.7	0 ± 0
SUGAR (%)	0.3 ± 0.2	1.4 ± 0.4	5.8 ± 1.2	9.8 ± 2.3
STORAGE (%)	0 ± 0	0 ± 0	0.0013 ± 0.0003	0.018 ± 0.005
Hydraulic Parameters				
SLOPE (%)	0.858 ± 0.145	0.820 ± 0.177	0.321 ± 0.056	0.048 ± 0.005
VELOC (ms^{-1})	0.18 ± 0.02	0.24 ± 0.03	0.31 ± 0.03	0.19 ± 0.01
REYNOLD	62470 ± 9740	79570 ± 13382	159400 ± 23410	83470 ± 11510
MUD (%)	0.16 ± 0.15	0.23 ± 0.13	2.9 ± 1.1	4.9 ± 0.55
ROCK (%)	29 ± 4	22 ± 1	6 ± 1	2 ± 1
BEDROCK (%)	10 ± 3	1 ± 0.4	1 ± 0.27	0 ± 0

Regression models

Ordinary least squares (OLS) regression models explained between 23.9% and 51.1% of the variation in macrophyte assemblage metrics (Appendix 3.8). However, SAR models generally explained between 6-10% additional variation in assemblage metrics (Table 3.10). Only three SAR models (ALIEN, EMERG and POACEAE) explained less variation than the equivalent OLS model. Only three SAR models (ALIEN, SUBMERG and EMERG) explained less than 40% of the variation in assemblage metrics.

Riparian score was a significant predictor for all but two SAR models (NATIVE and SUBMERG). These metrics demonstrated positive relationships with riparian condition i.e. metric scores increased with riparian condition. In contrast, the remaining metrics displayed significant negative relationships with riparian condition (Table 3.10). The magnitude of the regression coefficients suggests that riparian condition had the greatest influence on COVER and POACEAE and relatively minor influence on SPECRICH (Table 3.10). Water quality parameters were only significant for two models, and catchment land-use measures (mostly OTH_CROP) were significant predictors for three macrophyte metrics. For three models (NATIVE, ALIEN and EMERG) hydraulic parameters explained at least 10% of the variation in the SAR model residuals (Table 3.10).

SAR models were sensitive to the criterion chosen for defining neighbours. While the significance of individual coefficients varied little between the two neighbour definitions used for the SAR models in this study, the detection of autocorrelation (i.e. significance of rho) varied between lags (distances) for individual metrics in some cases (see COVER, SPECRICH, ALIEN in Table 3.10).

Autoregression models for 'Edge' habitats

The regression models presented in Table 3.10 were based on site-scale estimates of macrophyte cover. These estimates grouped two principal habitat types: 'edge' habitats in the stream margins, which tended to be characterised by emergent vegetation; and in-stream habitats that were generally devoid of macrophytes or characterised by vascular or non-vascular submerged taxa. Field observations indicated that a feature of sites located adjacent to agricultural areas was extensive macrophyte growth in edge habitats and little or no growth within in-stream habitats. The inclusion of in-stream quadrats may have masked relationships between emergent vegetation, land use and water quality. To investigate these relationships SAR models were rerun using metrics and habitat data calculated from edge quadrats only (i.e. the outer quadrats on each transect closest to the stream banks). These models are presented in Table 3.11. The SAR model for macrophyte cover in edge quadrats (EDGE_COVER) explained approximately 20% more variation than the COVER model (compare Tables 3.10 and 3.11). The SAR model for species richness of edge quadrats (EDGE_RICH) explained slightly more variation (5%) than the SAR model based on all quadrats. However, the POACEAE SAR model fit to edge quadrat data explained less variation (approximately 6%) than the equivalent SAR model fit to data for all quadrats (compare Tables 3.10 and 3.11).

Table 3.10. Parameters for SAR models fit to macrophyte assemblage metrics using two different neighbour definitions. Shading indicates the preferred SAR model based on R^2 and AIC. Also shown is the variation explained in the model residuals by hydraulic parameters (OLS regression of SAR model residuals). Significance: *0.05<P<0.025; **0.025<P<0.01; *** 0.01<P<0.001; ****P<0.001.

	COVER		SPECRICH		NATIVE		ALIEN	
Parameters	**Lag 1**	**Lag 2**	**Lag 1**	**Lag 2**	**Lag 1**	**Lag 2**	**Lag 1**	**Lag 2**
CATAREA			0.143****	0.129***				
RIPCOND	-0.934****	-0.650****	-0.145****	-0.124***	0.060	0.058	-0.348****	-0.378****
NOX	-0.115	-0.179						
FRP								
CONSERV	-0.023	-0.227						
OTH_CROP	0.384	0.096			-0.095***	-0.095***		
Intercept	3.643****	18.473**	0.647****	6.255****	1.832****	3.548	0.858****	5.895
Rho	-0.100**	-0.168	0.041	-0.227**	-0.022	-0.032	0.093*	-0.116
Wald (Rho)	7.168****	4.535*	1.778	8.875***	2.355	0.786	6.021**	2.045
Nagelkerke R^2	0.599	0.549	0.460	0.539	0.429	0.416	0.338	0.402
AIC	87.451	89.661	-10.368	-14.063	-20.333	-18.647	52.905	55.749
AIC (lm)	91.254	91.254	-10.611	-10.611	-20.051	-20.051	55.707	55.707
LM test for residual autocorrelation	2.422	1.035	0.281	0.704	0.306	2.198	0.366	2.783
Residual Variation Explained by Hydraulics								
Width					0.008**	0.006**		
Gravel							0.081	0.094
Bedrock							-0.168*	-0.187**
Intercept								
Adjusted R^2					0.145	0.144	0.138	

Table 3.10. (Continued). Significance: *0.05<P<0.025; **0.025<P<0.01; ***0.01<P<0.001; ****P<0.001.

	SUBMERG		EMERG		POACEAE	
Parameters	**Lag 1**	**Lag 2**	**Lag 1**	**Lag 2**	**Lag 1**	**Lag 2**
CATAREA						
RIPCOND	0.176	0.139	-0.269***	-0.254***	-0.415****	-0.362****
NO_X						
FRP					-0.170*	-0.147*
CONSERV						
OTH_CROP	-0.277***	-0.236****				
Intercept	1.296****	9.052*	1.189****	13.581***	0.992****	5.997*
Rho	-0.018	-0.192	0.091**	-0.220*	0.001	-0.153
Wald	0.144	3.591	7.471***	7.516***	0.000	2.699
Nagelkerke R^2	0.348	0.386	0.232	0.274	0.476	0.543
AIC	58.656	55.719	54.642	55.746	52.601	49.994
AIC (lm)	56.782	56.782	58.754	58.754	50.602	50.602
LM test for residual autocorrelation	0.261	1.663	5.654**	0.810	0.051	2.008
Residual Variation Explained by Hydraulics						
Gravel						
Bedrock				-0.150*		
Intercept						
Adjusted R^2				0.114		

Table 3.11. Parameters for SAR models fit to macrophyte assemblage metrics calculated from edge quadrats only and using two different neighbour definitions. Shading indicates the preferred SAR model based on R^2 and AIC. Significance: *0.05<P<0.025; **0.025<P<0.01; ***0.01<P<0.001; ****P<0.001.

	EDGE_COVER		EDGE_RICH		EDGE_ALIEN		EDGE_POACEAE	
Parameters	**Lag 1**	**Lag 2**	**Lag 1**	**Lag 2**	**Lag 1**	**Lag 2**	**Lag 1**	**Lag 2**
CATAREA			0.138****	0.113****				
ELEV	-0.046	-0.076						
RIPSCORE	-0.729****	-0.513***	-0.130****	-0.101****	-0.360****	-0.395****	-0.472****	-0.417****
PH	-0.312	-0.025						
TEMP	0.103	0.115						
TN	0.090	0.124						
NO_X	0.048	-0.057						
TP	-0.098	-0.189						
GRAZING	0.239	0.184						
PLANTAT	0.263	0.227						
SUGAR	-0.197	0.026						
OTH_CROP	0.399	0.157						
Intercept	3.916****	21.801***	0.633****	6.248****	0.949****	5.925	0.989****	6.391*
Rho	-0.081**	-0.178*	0.008	-0.262***	0.094**	-0.103	0.012	-0.159
AIC	84.519	85.778	-12.537	-20.417	51.262	54.985	56.400	53.795
AIC (lm)	88.732	88.732	-14.476	-14.476	54.723	54.723	54.452	54.452
R^2	0.793	0.805	0.436	0.518	0.363	0.423	0.408	0.486
Wald	6.730***	6.502**	0.057	13.644****	6.973***	1.867	0.064	2.837
LM test for residual autocorrelation	0.067	0.808	0.037	0.657	2.627	3.085	0.298	1.947

3.5 Discussion

Aquatic macrophyte assemblages of Australian lotic ecosystems have not been well described (Mackay *et al*. 2003). Consequently, the responses of aquatic macrophytes to anthropogenic disturbance of river catchments are not well known except in terms of gross assemblage changes such as infestation by alien species. This study has used macrophyte assemblage composition and simple assemblage metrics to investigate whether aquatic macrophyte assemblages of the wet tropics region of north Queensland could be used as reliable indicators of catchment land use, riparian condition and water quality. The results of this investigation have shown that macrophyte assemblage structure and metric scores were strongly associated with riparian condition but that relationships with land use and water quality were less clear.

Macrophyte assemblages as indicators of land use and riparian disturbance

We used two approaches to determine whether macrophyte assemblage structure could be used as an indicator of catchment land use and water quality. The first approach examined spatial patterns in catchment land use and water quality and related macrophyte assemblage to these patterns. The second approach determined spatial patterns in macrophyte assemblage structure within the study area and related catchment land-use and water quality data to these patterns. Both approaches produced broadly similar relationships between macrophyte assemblage structure and catchment land use and water quality. The results of both approaches are considered collectively in the following section.

Reliable bioindicators have predictable relationships with measures of environmental disturbance and have narrow environmental tolerances (Cranston *et al*. 1995) and should therefore occur in a discrete habitat type. The most reliable macrophyte indicator association found for the wet tropics region was the bryophyte-*Cladopus queenslandicus* assemblage that occurred in headwater sites. Bryophytes are commonly associated with headwater (high energy) habitats that are highly shaded and characterised by coarse substrata (e.g. Grasmück *et al*. 1995; Biggs 1996). *C. queenslandicus*, although a vascular plant, has a similar morphology to bryophytes and, like them, attaches to coarse substrata in flowing waters (Aston 1977; Dawson 1988). The bryophyte-*C. queenslandicus* assemblage occurred in the headwater reaches of all sub-catchments surveyed (mostly above 50 m AHD), suggesting that this assemblage type is ubiquitous in headwater streams of the region. However, disturbed headwater sites were not sampled so the response of this macrophyte assemblage to loss of riparian vegetation could not be established. The relatively low abundances of submerged vascular macrophytes in the study area suggests that competition with vascular macrophytes for space would not limit the growth of bryophytes in lower catchment areas.

The macrophyte assemblages of sites located below approximately 50 m AHD were dominated mostly by emergent vascular species. The proportion of anthropogenic land uses (predominantly sugar cane, other cropping and grazing) in the upstream catchment areas of these sites was higher when compared with group 1 sites (located mostly above 50 m AHD). Emergent assemblages occurring in the Little Mulgrave River (sites 2-8, group 2 in Figures 3.3 and 3.4) were characterised by a variety of taxa but only ferns (Pteridophytes) appear to have any utility as bioindicators. Ferns were present in many of the sites in the Little Mulgrave River and a small proportion of group 1 sites (see Table 3.8 and 3.9). They were generally absent from groups 3 and 4, which represented relatively disturbed sites (greater proportion of anthropogenic land uses in the upstream catchment area) with lower riparian condition and riparian cover. Group 2 sites had moderate scores for riparian condition and the occurrence of ferns in these sites may indicate the presence of a suitable moist microclimate resulting from riparian shading.

Sites in Behana, Woopen and Babinda Creeks (groups 3 and 4) were characterised by a variety of native and alien taxa including *Persicaria barbata*, *Sphagneticola trilobata*, *Cyperus trinervis* and *Urochloa mutica*, with *Blyxa* sp. and *C. trinervis* occurring as submerged taxa. *P. barbata* was the dominant species of this assemblage type in lower Behana Creek whereas *U. mutica* and *S. trilobata* dominated in sites with poor riparian canopy cover and riparian scores. The high frequency of occurrence of *P. barbata* in groups 3 and 4 (69 and 100% respectively for the site classification based on cover data) initially suggests that this species may have utility as a bioindicator. However, it appears that the occurrence or cover of *P. barbata* does not itself indicate poor stream condition (see group attributes in Tables 3.8 and 3.9). Groups 3 and 4 did not differ appreciably in terms of water quality but riparian condition varied considerably between these groups. While groups 3 and 4 had a relatively high proportion of land use as sugar cane (>5%), sites in these groups still retained approximately 90% or greater of the upstream catchment area as conservation (National Park, State Forest etc.). The greatest differences between groups 3-4 and 1-2 appear to lie in substratum composition, with groups 3 and 4 having a low proportion of rock but higher proportions of mud, when compared with groups 1 and 2. The occurrence of *P. barbata* in groups 3 and 4 may therefore indicate suitable substrata for establishment, rather than specific land-use or riparian influences.

The alien species *U. mutica* and *S. trilobata* also appear to have limited applicability as bioindicators of catchment land-use and/or riparian disturbance. While both species clearly dominated sites with low riparian condition (see group 3 of Table 3.9) the occurrence of both species in sites with relatively good riparian condition (see groups 2 and 4 of Table 3.9) shows that both species can also occur in relatively undisturbed environments. Both species have widespread distributions within Queensland (Henderson 2002). The presence of *U. mutica* and *S. trilobata* in sites with varying riparian condition suggests that both species have relatively wide ecological tolerances. Williams and Baruch (2000) have emphasised the importance of ecophysiological data to understand the effects of invasive species (particularly invasive grasses) on native species (see also Richards *et al*. 2003). There are few ecophysiological data available for Singapore daisy but the physiology of para grass, by virtue of its use as a pasture grass, has been intensively investigated (Miller 1980; Saxena *et al*. 1996; Guenni *et al*. 2002; Guenni *et al*. 2004). African grasses such as para grass have been found to allocate a greater proportion of their biomass to assimilating surfaces such as leaves, which favours whole-plant carbon fixation and growth (Williams and Baruch 2000). Para grass may not necessarily have a higher nutrient requirement than Australian native taxa but may respond more rapidly to nitrogen enrichment and use available nutrients more efficiently than the native taxa (Willliams and Baruch 2000). Few ecophysiological data are available for Australian native macrophyte taxa against which the performance of alien taxa such as *U. mutica* can be assessed.

U. mutica is commonly associated with disturbed habitats, including disturbed riparian zones where light availability is high, and is not thought to grow as well in shaded habitats (e.g. Wong 1990). Bunn *et al*. (1998) showed that 90% shade (as shade cloth) reduced total biomass of para grass by 52% in three months when compared with an unshaded control. In the present study, the presence-absence classification showed that para grass occurred in 50% of sites in group 3 which had a mean riparian canopy cover of 42 % (Table 3.8). Similarly, the classification of cover data showed that para grass only occurred in 25% of sites in group 4 (mean riparian canopy cover 73%) but occurred in 100% of group 3 sites (mean riparian canopy cover 16%). However, it is difficult to determine the riparian canopy cover that would limit or prevent the growth of para grass in wet tropics streams. Despite suggestions that para grass is not shade tolerant it has been shown that the growth of para grass and other tropical pasture grass species in shaded environments can be as great or exceed growth in full sunlight when full sunlight environments are nitrogen limited (Wilson and Wild 1990). Shaded environments may support a better soil microclimate than open environments, retaining soil moisture and stimulating bacterial growth and soil mineralisation (Wilson and Wild 1990). Saxena *et al*. (1996) found that under a mixed tree stand (approximately 50% shade) the total net primary productivity of para grass was 15% higher than in open (unshaded) conditions. The relatively high occurrence of para grass in sites with good riparian condition may therefore reflect a suitable soil microclimate, including relatively high nitrogen availability. We have insufficient data to demonstrate the influence of these processes for our study sites.

There are few physiological data available for *S .trilobata. S. trilobata* is a Class 3 declared weed (Land Protection 2006) that occurs along stream and river margins in shaded and unshaded habitats. Thus the distribution of this species in wet tropics streams would not be limited necessarily by the retention of good riparian canopy cover, although extreme shading (as occurs in headwater sites) could be effective. There were no obvious patterns in the distribution of Singapore daisy in relation to water quality, land use or riparian condition. However, it should be noted that the spread of alien species within the wet tropics region could be facilitated by vehicular movement, the presence of bridges and roadways and other anthropogenic activities in addition to those associated directly with land-use changes (King and Buckney 2000; Goosem 2002; Wet Tropics Management Authority 2005).

Assemblage metrics as descriptors of land use and riparian condition

Seven metrics were initially suggested as suitable descriptors of macrophyte assemblage structure based on predicted changes in assemblage structure following land-use changes: COVER, SPECRICH, NATIVE, ALIEN, SUBMERG, EMERG and POACEAE. These metrics were also considered to be easily employed by non-specialists. SAR models for metrics derived from whole-of-site data generally explained insufficient variation to be used confidently as indicators of catchment land use or water quality (Table 3.10). The best SAR models were COVER (59.9% variation explained) and POACEAE (54.3% variation explained). The remaining SAR models explained between 27%-46% of the variation in individual metrics. Very few land-use or water quality parameters were significant predictors in the SAR models fit to whole-of-site data. In comparison, riparian condition was a significant predictor in all but two of the SAR models fit to whole-of-site data (see Table 3.10). Model coefficients indicate that riparian condition has a negative influence on macrophyte cover, species richness and the proportions of alien taxa, emergent taxa and Poaceae present at sites in the wet tropics. SAR models showed that the proportions of native and submerged taxa were positively associated with riparian condition (but not significantly). The proportion of land use under crops other than sugar cane was a significant (negative) predictor for SAR models for these metrics. However, the relatively low R^2 for these models suggests that these metrics would not be robust indicators of the impacts of other types of cropping on aquatic ecosystems.

Ordinary least squares (OLS) regression models fit to SAR model residuals (with hydraulic parameters as predictors) explained greater than 10% of the variation in SAR model residuals for three metrics (NATIVE, ALIEN, EMERG). The inclusion of hydraulic parameters in SAR models would not explain sufficient variation to warrant use of these metrics as bioindicators. Nonetheless, it is apparent from the OLS models that higher proportions of emergent and alien taxa are associated with relatively fine substrata, as indicated by the negative coefficient for bedrock (Table 3.10).

Metrics for Edge data

Several metrics were calculated based on data collected from edge quadrats only. Based on percentage of variation explained by SAR models, three (edge) metrics were found to have potential as indicators of catchment and riparian disturbance in the wet tropics region: macrophyte cover in edge quadrats (EDGE_COVER), species richness of edge quadrats

(EDGE_RICH) and proportion of taxa belonging to the family Poaceae (POACEAE, all quadrats). SAR models explained 79.3%, 51.8% and 54.3% of the variation in these metrics respectively. EDGE_COVER was clearly the best metric in terms of variation explained by the SAR model. This metric is probably the easiest to use as it requires little specialist knowledge of the flora present and cover estimates are based on broad cover categories, reducing potential for operator error in assessments. EDGE_RICH and POACEAE both require flowering material to confirm the identity of specimens, and this may be difficult if sampling is conducted outside of flowering times. Although some training may be required by non-specialists to properly utilise these metrics they are nonetheless relatively easy to use as specialist equipment is not required.

While EDGE_RICH and POACEAE are potentially useful metrics there are relatively large proportions of the variation unexplained by the SAR models for these metrics. Use of the metrics could be better justified by defining additional sources of potential variation. A similar argument could be applied to other edge and whole-of-site metrics, as they also explained relatively large amounts of variation (approximately or higher than 40%) but insufficient variation to be used confidently as assemblage metrics in a biomonitoring program. Additional sources of variation in the metrics include nutrient availability, substratum stability and disturbance frequency.

EDGE_RICH and POACEAE are essentially descriptors of emergent macrophyte assemblages within the study area. The primary source of nutrients (N and P) for emergent macrophytes would be stream sediments and/or the water column. The relative importance of either nutrient pool for emergent macrophytes may depend upon their respective nutrient concentrations (Carignan 1982; Best *et al*. 1996). Nutrient concentrations in surface waters, as used in this study, may not accurately describe nitrogen and phosphorus availability for emergent macrophytes if sediment nutrient pools are being used exclusively in preference to water column nutrient pools. SAR models for metrics associated with emergent macrophyte assemblages may therefore be improved by the inclusion of measures of interstitial nutrients. For example, King and Buckney (2000) found that differences in within-stream vegetation in urban streams of Sydney were associated with nutrient concentrations in stream sediments, and concluded that elevated sediment nutrient availability facilitated invasion by alien species. However, whilst interstitial nutrient concentrations may explain further variation in the measured metrics they are likely to be spatially heterogeneous reflecting factors such as local sediment sorption properties, exchange across the water-sediment boundary and, the presence of vegetation itself. Consequently, sampling interstitial nutrients is likely to present considerable challenges.

Hydrology and hydraulic habitat are potential additional sources of unexplained variation in SAR metric models. Hydrologic data were unavailable for this study but hydrological attributes such as flood frequency and time since last flood are known to be important correlates of stream macrophyte assemblage structure (Riis and Biggs 2003; Mackay and Marsh 2005). In particular, the frequency of flood events capable of mobilising stream substrata can limit the establishment of macrophytes and influence above-ground biomass (Riis and Biggs 2003). Frequent substratum mobilisation may preclude the establishment and growth of macrophytes (Riis and Biggs 2003) and substratum stability at base-flow conditions (i.e. the time of our survey) could also influence the establishment of submerged macrophytes in lower Babinda and Behana Creeks. These streams had sandy substrata that were mobilised at the water velocities (approximate maximum water velocities 0.40-0.70 ms^{-1}) recorded at the time of sampling (S. Mackay personal observation). Water velocities in edge habitats may not have limited the growth of emergent vegetation to the same extent as velocities in stream habitats. OLS models fit to SAR (edge) model residuals explained little additional variation, suggesting that the hydraulic parameters measured (depth, water velocity, substratum composition) had little effect on assemblage metrics at the edge scale at the time of sampling.

While EDGE_COVER, EDGE_RICH and POACEAE are potentially useful metrics for assessing riparian disturbance they do not appear to be useful metrics for assessing land-use or water quality impacts. Edge metrics (like whole-of-site metrics) were strongly related to riparian condition, suggesting that light limitation (and potentially temperature) were the main factors influencing assemblage metrics. The weak relationships between anthropogenic land-use, water quality and assemblage metrics may have been due to the ‘length’ of the catchment disturbance gradient and the time of sampling. For example, the percentage of conservation land uses (National Park, State Forest etc.) was at least 85% for all sites, even in relatively disturbed catchments such as Woopen and Babinda Creeks. Relatively good in-stream habitat and biotic integrity may occur in catchments with very high proportions of anthropogenic land uses (see Harding *et al*. 1999). Investigations of the effects of land use on water quality and biotic assemblage structure in streams have also reported negative impacts (i.e., reduced stream health) over relatively short disturbance gradients. Snyder *et al*. (2003) found that sites with poor Index of Biotic Integrity (IBI) scores had greater than 7% of urban land use in the upstream catchment. In this study SUBMERG and NATIVE were negatively correlated with OTH_CROP, and POACEAE was negatively correlated with FRP (Appendix 3.3). Harding *et al*. (1999) suggested that measures of agricultural intensity rather than percentage of differing land use may be a more useful indicator of the impacts within a river system.

Water quality was not strongly associated with metric scores (although assemblage composition was found to vary over water quality gradients). Variations in water quality throughout the study area were relatively small (see Table 3.7) so it is perhaps unsurprising that assemblage metrics were not strongly related to water quality. The region is characterised by a narrow coastal plain and therefore streams of the region are potentially receiving fewer agricultural runoff inputs under base-flow conditions than other eastern Queensland streams with larger catchment areas. The highest nutrient loads are transported by flood flows (Brodie and Mitchell 2005). Nonetheless, TN and TP concentrations exceeded Queensland EPA guidelines for upland (TN/TP) and lowland (TP only) streams (EPA 2006)(see Chapter 2). Surprisingly, the greatest deviation from TP reference guidelines was for the relatively pristine headwater sites and the Little Mulgrave River (see Table 3.7). Russell *et al*. (1996) hypothesized that the occurrence of extensive beds of *Hydrilla verticillata* and *Vallisneria nana* in the Mulgrave River was associated with sewage discharges. However, elevated TN and TP levels were not associated with excessive submerged macrophyte growth in the wet tropics study area (see regression coefficients in Table 3.10).

Conclusions and Recommendations

The key macrophyte assemblages identified in the wet tropics region have limited applicability as direct indicators of catchment land use (over the land-use gradient surveyed). While each assemblage type was associated with different land-use categories, all assemblage types were arrayed over a gradient of riparian canopy cover and riparian condition. Riparian canopy cover and condition scores were negatively correlated with the proportions of anthropogenic land uses (see Appendix 3.3). This suggests that riparian restoration itself would restore aquatic macrophyte assemblages in disturbed streams such as Babinda and Woopen Creeks to a ‘pre-disturbance’ state. Thus, in terms of the metrics presented here, riparian restoration alone would be expected to have significant benefits for aquatic macrophyte assemblages in the wet tropics region, independent of any land-use impacts. Consequently, it is evident that a riparian condition assessment would provide an adequate indication of the state of aquatic macrophyte assemblages in wet tropics, based on the range of metrics presented.

Hydraulic habitat was also an important determinant of macrophyte assemblage structure. Haury (1996) noted that 'water quality diagnosis with macrophytes cannot ignore the physical context'. For example, the bryophyte-*Cladopus queenslandicus* assemblage occurred on very coarse substrata such as rocks and bedrock. This is undoubtedly because of the growth habit of the taxa (attaching directly to stream substrata) and the high stability of substrata required for bryophyte establishment (Biggs 1996; Suren and Duncan 1999). This assemblage would therefore not be expected to occur in lower elevation sites such as lower Babinda and Behana Creeks, which had relatively fine and mobile substrata.

Three assemblage metrics (EDGE_COVER, EDGE_RICH and POACEAE) have promise as indicators of riparian disturbance in the wet tropics region. However, their utility as indicators of catchment land use *per se* appears limited. These metrics were therefore either poor descriptors of assemblage structure, or were not strongly influenced by catchment land use or water quality. SAR models for all metrics included riparian condition as a significant predictor, with water quality and land-use descriptors poor predictors. Riparian condition was negatively related to COVER, ALIEN and POACEAE, suggesting that maintenance or restoration of riparian zones would be the most successful approach to maintain aquatic macrophyte assemblages in wet tropics streams in a relatively intact state.

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Appendices

Appendix 3.1. Riparian condition scores for sites surveyed in the wet tropics region.

		% Cover				Riparian Condition Assessments												
						Left Bank					Right Bank					Totals		
Tributary	Site Number	Left Bank	Central	Right Bank	Mean site value	Width of riparian zone	Linear continuity	Canopy vigour	Natives vs exotics	Indigenous regeneration	Width of riparian zone	Linear continuity	Canopy vigour	Natives vs exotics	Indigenous regeneration	Left Bank score	Right Bank Score	Total Site Score
LM	1	98.4	93.8	98.7	**97.0**	5	5	5	5	5	5	5	5	5	5	25	25	**50**
LM	2	72.3	40.2	73.3	**61.9**	2	2	2	2	3	5	5	5	5	5	11	25	**36**
LM	3	94.9	89.1	98.0	**94.0**	4	4	4	5	5	5	5	5	5	5	22	25	**47**
LM	4	84.4	76.0	81.5	**80.6**	4	5	5	2	5	4	4	5	3	5	21	21	**42**
LM	5	83.0	86.8	97.5	**89.1**	4	5	5	4	5	3	5	5	5	5	23	23	**46**
LM	6	86.6	79.7	66.1	**77.5**	3	5	5	5	5	5	3	3	2	3	23	16	**39**
LM	7	92.6	82.3	71.7	**82.2**	2	5	5	5	4	4	3	3	2	2	21	14	**35**
LM	8	52.0	38.5	44.5	**45.0**	5	5	5	5	4	3	2	2	1	1	24	9	**33**
Babinda	9	6.8	10.3	58.0	**25.0**	4	5	5	1	3	2	3	3	2	2	18	12	**30**
Babinda	10	61.6	27.7	62.3	**50.5**	5	5	5	5	5	2	4	4	4	4	25	18	**43**
Babinda	11					5	5	5	5	5	5	5	5	5	5	25	25	**50**
Babinda	12	16.5	27.8	61.0	**35.1**	1	1	1	1	1	2	4	3	2	2	5	13	**18**
Babinda	13	0.5	0.6	7.2	**2.7**	1	1	1	1	1	4	4	3	2	3	5	16	**21**
Babinda	14	1.9	0.0	7.4	**3.1**	1	1	1	1	1	1	1	1	1	1	5	5	**10**
Babinda	15	17.2	2.2	20.1	**13.2**	1	1	1	1	1	1	1	1	1	1	5	5	**10**
Babinda	16	3.2	0.7	5.8	**3.2**	1	1	1	1	1	1	1	1	1	1	5	5	**10**
Babinda	17	9.1	2.4	13.6	**8.4**	2	2	3	2	2	2	4	4	2	2	11	14	**25**
Babinda	18	12.3	0.8	21.0	**11.4**	1	1	1	1	1	2	2	3	2	3	5	12	**17**
Babinda	19					1	1	1	1	1	1	1	1	1	1	5	5	**10**
Babinda	20																	
Behana	21	95.9	68.8	98.2	**87.7**	5	5	5	5	5	5	5	5	5	5	25	25	**50**
Behana	22	92.0	82.5	94.7	**89.7**	5	5	5	5	5	2	5	5	5	4	25	21	**46**
Behana	23																	
Behana	24	66.2	44.4	30.1	**46.9**	3	5	5	2	3	5	4	5	2	3	18	19	**37**
Behana	25	66.0	45.1	85.5	**65.5**	2	4	4	5	4	2	4	4	5	4	19	19	**38**
Behana	26	73.7	72.4	80.2	**75.4**	5	5	5	5	5	5	4	4	4	3	25	20	**45**
Behana	27					5	5	5	5	5	5	5	5	5	5	25	25	**50**
Behana	28	12.2	3.3	13.7	**9.7**	1	1	1	1	1	5	4	5	3	4	5	21	**26**
Behana	29	90.6	88.2	98.8	**92.5**	2	5	5	5	5	2	5	5	5	5	22	22	**44**
Behana	30	46.6	41.8	89.2	**59.2**	3	2	2	2	2	3	5	5	5	4	11	22	**33**
Woopen	31	9.7	15.5	26.2	**17.2**	3	3	3	1	2	2	1	1	1	1	12	6	**18**
Woopen	32	48.9	20.8	49.3	**39.7**	1	1	2	1	2	1	2	2	2	2	7	9	**16**
Woopen	33	10.0	2.5	14.4	**9.0**	3	2	2	1	1	3	2	2	1	1	9	9	**18**
Woopen	34	9.0	6.7	12.7	**9.5**	1	1	1	1	1	1	1	1	1	1	5	5	**10**
Woopen	35	2.0	1.3	5.4	**2.9**	4	2	3	1	2	3	4	3	1	2	12	13	**25**
Woopen	36	22.8	4.2	34.9	**20.6**	1	1	1	1	1	2	2	2	1	1	5	8	**13**
LM	37	92.8	84.3	89.9	**89.0**	5	5	5	5	5	5	5	5	5	5	25	25	**50**
Woopen	38	95.4	91.0	96.1	**94.2**	5	5	5	5	5	5	5	5	5	5	25	25	**50**
Woopen	39	95.7	93.3	94.3	**94.4**	3	5	5	5	5	4	5	5	5	5	23	24	**47**
Babinda	40																	

Appendix 3.2. Pro forma used for Assessing Riparian Condition. From Werren and Arthington (2002).

1. Width of Riparian Zone (as proportion of average stream width or low flow width)	**score**	**4. Proportion of Natives vs Exotics** (as % cover in community)	**score**
>3 x wetted width	5	native species account for 90-100% of cover	5
>2 x wetted width	4	native species account for 75-89% of cover	4
1-2 x wetted width	3	native species account for 60-74% of cover	3
<1 x wetted width	2	native species account for 35-59% of cover	2
forested verge absent/severely depleted	1	native species account for <35% of cover	1
2. Linear Continuity (% of naturally vegetated bank length -100m sample)	**score**	**5. Extent of Indigenous Regeneration** (stem size class variation/seedling abundance)	**score**
91-100% vegetated with expected riparian vegetation (e.g., native forest, tall shrubs, etc) without significant discontinuities	5	various stem size classes represented in community (where relevant) and/or canopy seedlings abundant	5
75-90% vegetated (see above) with significant discontinuities 1-2	4	variation in stem size classes evident (where relevant); canopy seedlings frequent	4
50-74% vegetated (see above) with significant discontinuities 3-4	3	little variation in stem size classes (where relevant); canopy seedlings occasional	3
25-50 % vegetated (see above)with significant discontinuities >5	2	stem size class distribution uniform (where relevant); canopy seedlings rare/not present	2
0-24% vegetated (see above) with significant discontinuities >5	1	few canopy stems present, or when so relatively uniform; canopy seedlings absent	1
3. Canopy Vigour/Crown Health and Structural Intactness	**score**	**OVERALL CONDITION SCORE**	
tree canopy appears intact; no/few standing dead spars	5	**The larger the score the better the condition of riparian vegetation. Maximum = 25**	
canopy slightly irregular and/or with some gaps; no/few dead spars	4	**Total Score (Σ) =**	
canopy +/- sparse or lacking vigour; dead spars may be evident; minor crown dieback	3	**Note: scores are summed and multiplied by 4 to give a relative score out of 100.**	
tree canopy sparse, individuals exhibit crown dieback; dead spars prevalent	2		
canopy very sparse/non-existent; shrubs &/or grasses prevalent (spars may occur)	1	**Final Score = Σ x 4 =**	

ADDITIONAL COMMENTS:

Bank Form:

In-stream Features:

Vegetation Structural type:
………………………..proxima l ………………………..distal
Other: (dominant/co-dominant species, weed species, etc)

Appendix 3.3. Spearman correlation coefficients for assemblage metrics and land-use and water quality parameters. Only significant correlations shown. Significance * P=0.05; **P=0.01.

ICH	NATIVE	ALIEN	POACEAE	EMERG	SUBMERG	CATAREA	DISTM	ELEV	RIPCOV	RIPSCORE	DO	COND	PH
	1												
	-0.861**	1											
	-0.618**	0.641**	1										
	-0.593**	0.718**	0.757**	1									
	0.692**	- 0.767**	-0.719**	-0.950**	1								
						1							
							1						
						-0.529**	0.879**	1					
	0.585**	- 0.678**	-0.732**	-0.589**	0.514**		0.435*	0.471**	1				
	0.523**	- 0.645**	-0.623**	-0.626**	0.544**		0.358*	0.428*	0.878**	1			
		-0.419*			0.372*						1		
							0.618**	0.348*			- 0.471**	1	
							0.901**	0.796**	0.548**	0.432*		0.703**	1
		0.442**							-0.349	-0.427*	- 0.624**	0.403*	
												0.447**	
	-0.377*	0.444**							-0.552*	-0.617**			
						0.504**			0.343*			0.408*	
		0.392*		0.357*	-0.346*				-0.526**	-0.648**			
							0.737**	0.534**	0.339*			0.708**	0.728**
			-0.466**				0.801**	0.635**	0.467**			0.781**	0.843**
		- 0.481**	-0.481**	-0.458**	0.397*		0.537**	0.631**	0.659**	0.703**			0.503**
	-0.476**	0.484**	0.500	0.352*		-0.636**			-0.632**	-0.683**			
							-0.362*		-0.520**	-0.448**	0.385*	- 0.577**	- 0.535**
						0.419*	- 0.730**	- 0.900**	-0.546**	-0.547**			- 0.680**
	-0.647**	0.697**	0.488**	0.481**	-0.479**				-0.717**	-0.823**	-0.403*		
						0.811**		- 0.539**					
							- 0.542**	- 0.560**					- 0.481**
						0.370*	-0.49**	-0.52**					-0.377*

Appendix 3.3. Continued.

	TEMP	TURB	TN	NH3	NOX	TP	FRP	CONSERV	GRAZE	PLANTAT	SUGAR	OTH_CROP	RESID	INDUST	STORAGE
COVER															
SPECRICH															
NATIVE															
ALIEN															
POACEAE															
EMERG															
SUBMERG															
CATAREA															
DISTM															
ELEV															
RIPCOV															
RIPSCORE															
DO															
COND															
PH															
TEMP	1														
TURB		1													
TN	0.404*		1												
NH3				1											
NOX	0.485**				1										
TP						1									
FRP						0.803**	1								
CONSERV	-0.560**	-0.409*	-0.714**		-0.815**		0.341*	1							
GRAZE	0.419*		0.598**	-0.441**	0.599**			-0.631**	1						
PLANTAT				-0.559**		-0.428*	-0.548**		0.340*	1					
SUGAR	0.395*		0.538**		0.618**	-0.366*	-0.467**	-0.824**			1				
OTH_CROP	0.572**		0.603**		0.635**			-0.583**	0.749**			1			
RESID											0.459**		1		
INDUST					0.423*						0.641**			1	
STORAGE				0.544**							0.471**	-0.402*		0.463**	1

Appendix 3.4. *Species recorded in aquatic plant (A) and ground cover (R) assessments. * Non-native species. † species recorded as rare under the Queensland Nature Conservation (Wildlife) Regulation (1994).* ***‡ Species listed as vulnerable under the Endangered Species Protection Act (1992).***

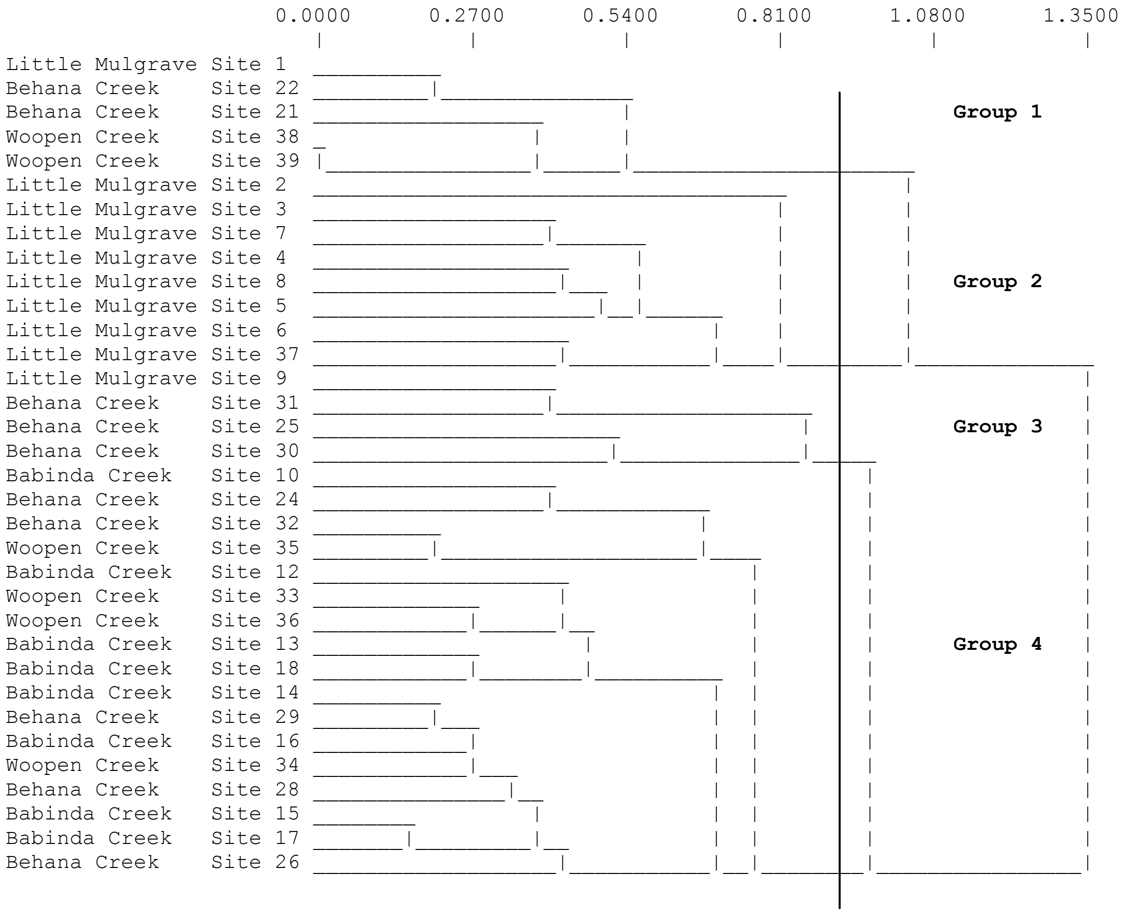
Family	Species	Common name	Survey
Alismataceae	*Saggitaria* sp.	Arrow head	A, R
Araceae	*Syngonium podophyllum* Schott*	Schott arrowhead plant	R
	Colocasia esculenta (L.) Schott*	Taro	A, R
Cyperaceae	*Cyperus aquatilis* R.Br.	Water nutgrass	A, R
	Cyperus aromaticus (Ridl.) Mattf. & Kuek.*	Nauva sedge	A, R
	Cyperus brevifolius (Rottb.) Hassk.*	Mullumbimby couch	R
	Cyperus haspan var haspan L.		R
	Cyperus involucratus Rottb.*	Umbrella sedge	A, R
	Cyperus odoratus L.	Flat sedge	R
	Cyperus pilosus Vahl	Hairy flat sedge	R
	Cyperus polystachyos Rottb.		A, R
	Cyperus sphacelatus Rottb.		A, R
	Cyperus trinervis R.Br.	Australian flatsedge	A, R
	Fimbristylis littoralis Gaudich		R
	Rhynchospora corymbosa (L.) Britton		R
	Schoenoplectus mucronatus (L.) Palla ex J.Kearn.	Club rush	A, R
	Scleria laevis Retz.		R
Hydrocharitaceae	*Blyxa* sp.		A
	Hydrilla verticillata (L.f.) Royle	Hydrilla	A
	Vallisneria nana R. Br.†	Ribbon weed	A
Lomandraceae	*Lomandra* sp.	Matrush	A, R
Onagraceae	*Commelina* spp.		A, R
Philydraceae	*Philydrum lanuginosum* Banks & Sol. Ex Gaertn.	Frogsmouth	A, R
Poaceae	*Arundo donax* L. var. *donax**	Giant reed	A, R
	Axonopus fissifolius (Raddi) Kuhlm.*	Narrow-leaf carpet grass	A, R
	Centotheca lappacea (L.) Desv.		R
	Centotheca philippinensis (Merr.) C.Monod†‡		R
	Chrysopogon fallax S.T.Blake	Golden beard grass	R
	Chrysopogon filipes (Benth.) Reeder		A, R
	Cyrtococcum oxyphyllum (Hochst.) ex Steud.) Stapf		A, R
	Digitaria setigera Roem. & Schult.*		R
	Echinochloa colona (L.) Link*	Awn less barnyard grass	R
	Megathyrsus maximus (Jacq.) B.K.Simon & S.W.L.Jacobs*	Guinea Grass	A, R
	Oplismenus burmannii (Retz.) P.Beauv.		R
	Oplismenus hirtellus subsp. *imbecillis* (R.Br.) U.Scholz		R
	Paspalum conjugatum P.J.Bergius*	Johnsons River Grass	R
	Paspalum scrobiculatum L.	Scrobic or ditch millet	R
	Pennisetum pupureum Schumach.*	Elephant grass	A, R
	Pennisetum setaceum (Forssk.) Chiov.*	Fountain grass	R

Family	Species	Common name	Survey
	Sacciolepis indica (L.) Chase		A, R
	Sorghum halepense (L.) Pers.*		A, R
	Urochloa mutica (Forssk.) T.Q.Nguyen*	Para grass	A,R
Potamogetonaceae	*Potamogeton javanicus* Hassk.	Java pondweed	A
Acanthaceae	*Hygrophila* sp.		A, R
Amaranthaceae	*Alternanthera ficoidea* (L.) P. Beauv.*		R
Apiaceae	*Hydrocotyle* sp.	Pennywort	A, R
Asteraceae	*Ageratum conyzoides* L. subsp *conyzoides*	Blue billy goat weed	A, R
	Crassocephalum crepidioides (Benth.) S.Moore		R
	Cyanthillium cinereum (L.) H.E. Robins.		R
	Eclipta prostrata (L.) L.	False daisy	R
	Sphagneticola (*Wedelia*) *trilobata* (L.) Pruski*	Singapore daisy	A, R
	Synedrella nodiflora Gaertn.*	Nodeweed	R
Caryophyllaceae	*Drymaria cordata* (L.) Willd. ex Roem & Schult.	Tropical chickweed	A, R
Clusiaceae	*Hypericum gramineum* G.Forst.	St. Johns wort	R
Elatinaceae	*Elatine gratiolides* A.Cunn.	Waterwort	A
Euphorbiaceae	*Chamaesyce hirta* (L.) Millsp.*	Hairy spurge	R
	Phyllanthus tenellus Roxb.*		R
Fabaceae	*Centrosema molle* Mart. ex Benth.*		R
	Neonotonia wightii (Wight & Arn.) J.A.Lackey*		R
Haloragaceae	*Myriophyllum* sp.	Watermilfoil	A
Lamiaceae	*Hyptis capitata* Jacq.*	Knob weed	R
Leguminosae	*Mimosa pudica* var. *unijuga* (Walp. & Duchass.) Griseb.*	Common sensitive plant	R
Malvaceae	*Sida rhombifolia* L.*		R
Onagraceae	*Ludwigia hyssopifolia* (G.Don) Exell		R
	Ludwigia octovalvis (Jacq.) P.H.Raven		R
Podestemaceae	*Torrenticola queenslandica* (Domin) Domin ex Steenis†		A
Polygonaceae	*Persicaria attenuata* (R.Br.) Sojak*		R
	Persicaria barbata (L.) H.Hara		A, R
	Persicaria lapathifolia (L.) Gray*		A, R
	Persicaria strigosa (R.Br.) H.Gross		A, R
Rosaceae	*Rubus alceifolius* Poir.*	Giant bramble	R
Rubiaceae	*Hedyotis auricularia* var. *melanesica* L.*		R
Verbenaceae	*Faradaya splendida* F.Muell.		R
	Stachytarpheta cayennensis (Rich.) J.Vahl*	Snakeweed	R

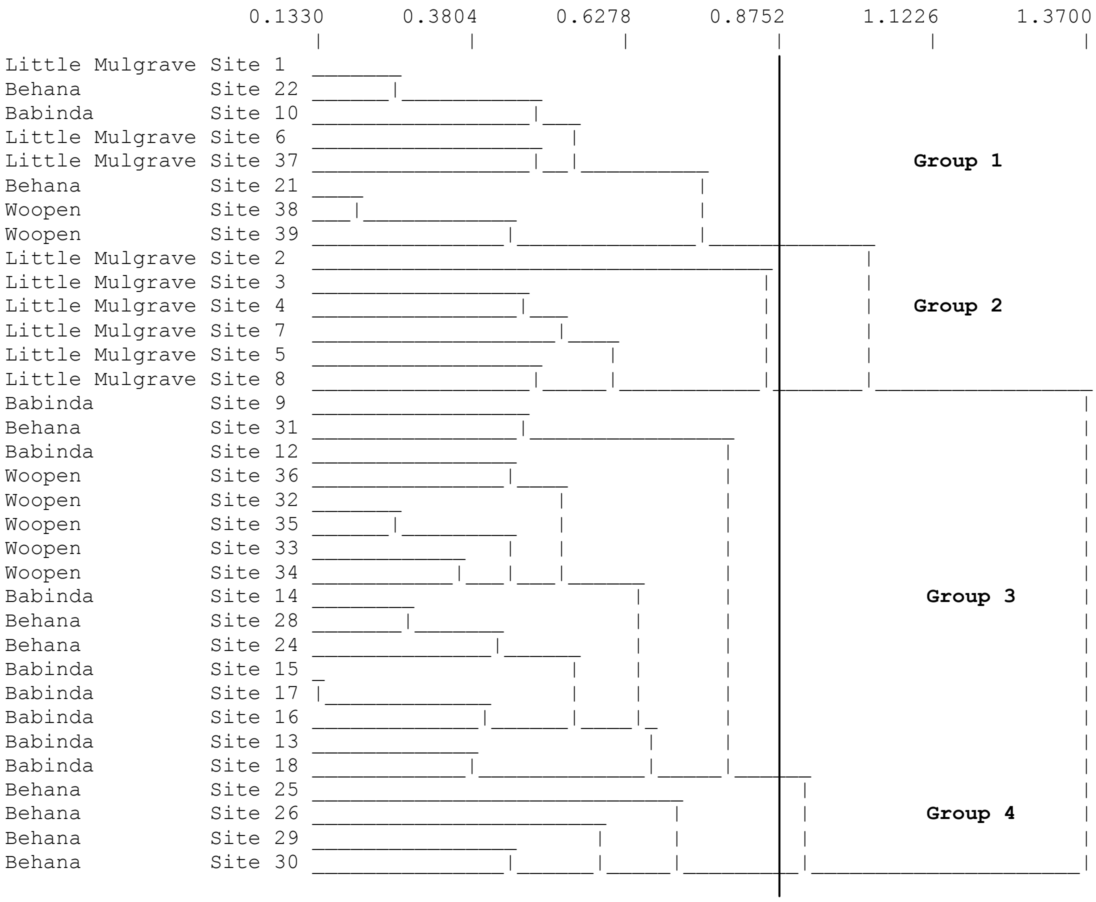
Appendix 3.5. UPGMA classification of sites based on land use and water quality.

```
                   0.0092      0.0350      0.0607      0.0865      0.1122      0.1380
                        |           |           |           |           |           |
Little Mulgrave Site 1  _
Little Mulgrave Site 37 |________________
Little Mulgrave Site 2  ________________|______
Woopen          Site 38 _________________     |
Woopen          Site 39 ________________|_____|_
Little Mulgrave Site 3  _______________________|________              Group 1
Little Mulgrave Site 4  _____                          |
Little Mulgrave Site 5  ____|                          |
Little Mulgrave Site 6  ___||_______                   |
Little Mulgrave Site 7  ______     |                   |
Little Mulgrave Site 8  _____|_____|___________________|___________________________
Woopen          Site 31 _____________                                             |
Woopen          Site 36 ____________|___________________                          |
Woopen          Site 32 _____________________          |              Group 2     |
Woopen          Site 33 __________          |          |                          |
Woopen          Site 34 _________|__________|______    |                          |
Woopen          Site 35 __________________________|____|__________________________|__
Babinda         Site 9  _______                                                     |
Babinda         Site 10 ______|______________________                               |
Behana          Site 21 ______                      |                               |
Behana          Site 22 _____|__________            |                 Group 3       |
Behana          Site 24 __________     |            |                               |
Behana          Site 25 _________|_____|____________|_____________                  |
Behana          Site 26 _____________                            |                  |
Behana          Site 29 ____________|______                      |    Group 4       |
Behana          Site 28 ________________  |                      |                  |
Behana          Site 30 _______________|__|______________________|__                |
Babinda         Site 12 _______________                            |                |
Babinda         Site 13 _________     |                            |                |
Babinda         Site 14 ________|_____|_____                       |  Group 5       |
Babinda         Site 15 ______             |                       |                |
Babinda         Site 16 _____|_____        |                       |                |
Babinda         Site 17 ___________|_______|_________              |                |
Babinda         Site 18 ____________________________|______________|________________|
```

Appendix 3.6. Dendrogram for site classifications based on presence-absence data.



Appendix 3.7. Dendrogram for site classification based on species Braun-Blanquet cover data.



Appendix 3.8. Results of ordinary least squares (OLS) regression models for macrophyte metrics. Significance: *0.05<P<0.025; **0.025<P<0.01; *** 0.01<P<0.001; ****P<0.001.

	Metrics						
Parameters	**COVER**	**SPECRICH**	**NATIVE**	**ALIEN**	**SUBMERG**	**EMERG**	**POACEAE**
CATAREA		0.155****					
RIPCOND	-0.737***	-0.157****	0.063	-0.418****	0.179	-0.318***	-0.416****
NOX	-0.186						
FRP							-0.171
CONSERV	-0.227						
OTH_CROPS	0.174		-0.092**		-0.259**		
Intercept	2.824****	0.739****	1.722****	1.220****	1.235****	1.649****	0.995****
Adjusted R^2	0.481	0.475	0.374	0.383	0.328	0.239	0.511

Chapter 4

Macroinvertebrates as Indicators of Ecosystem Health in Wet Tropics Streams

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4.1 Introduction

Macroinvertebrates are a significant and important component of all freshwater ecosystems, making up a substantial part of the biodiversity and performing many ecosystem functions (e.g., decomposition, nutrient cycling and energy transfer). They include aquatic adult and/or larval stages of insects and many other invertebrate phyla, including worms, molluscs, crustaceans, spiders and mites. They are crucial to the proper functioning of aquatic ecosystems, typically making the first linkages in aquatic food webs. They connect the primary producers and microbes to higher consumers, especially fish, amphibians, reptiles, birds and mammals. For example, macroinvertebrates have been shown to be the most important component of the diet of freshwater fish in Australia (Arthington 1992; Pusey *et al.* 1995; Kennard *et al.* 2001) and represent ~80% of the food of the platypus (Faragher *et al.* 1979). The adult stages of many aquatic insects emerge into the terrestrial environment to mate and complete their life cycles and in doing so become an important food source for terrestrial predators, particularly insectivorous birds, thereby linking terrestrial and aquatic food webs (e.g. Likens and Bormann 1974; Jackson and Fisher 1986; Gray 1993).

In forest streams, much of the productivity is sustained by the input of terrestrial organic matter, mostly leaves from the surrounding forest trees (Kaushik and Hynes 1971). Many of the invertebrates play a vital role in transforming this coarse particulate organic matter by scraping, gouging and shredding it, making it available to other invertebrates and facilitating microbial colonisation and decomposition (Pearson and Tobin 1989; Pearson *et al.* 1989; Nolen and Pearson 1993; Gessner *et al*. 1999; Pearson and Connolly 2000). In this way, aquatic invertebrates are analogous to the terrestrial detritivorous invertebrates that facilitate terrestrial detritus decomposition and nutrient cycling, maintaining soil fertility and terrestrial productivity.

4.1.1 Macroinvertebrates as biomonitors

Aquatic macroinvertebrates offer a time-integrated sample of environmental conditions over their lifetime (weeks to years) and consequently have been regularly used as indicators of water quality and ecosystem health (Rosenberg and Resh 1993). They are numerous and ubiquitous, occurring in nearly all water bodies and are easily sampled using cheap, readily available equipment, making them ideal for this purpose. The aquatic macroinvertebrates are typically diverse, with different species having specific requirements for biophysical conditions. As a consequence, their distributions follow natural gradients in environmental conditions and they have been shown to respond to changes in water quality and physical parameters associated with anthropogenic disturbance (Connolly and Pearson 2004). For example, they have been demonstrated to be sensitive to changes in water chemistry, including dissolved oxygen concentration (Connolly *et al.* 2004), pH (Rutt *et al.* 1990), salinity (Metzeling 1993) and to be vulnerable to toxic contaminants such as insecticides (Liess 1994; Shultz and Liess 1995). They have also been shown to respond to organic pollution (Pearson and Penridge 1987) and nutrient enrichment (Pearson and Connolly 2000). The clearing of riparian vegetation and increases in sedimentation have also been shown to be detrimental to macroinvertebrate assemblages (Ryan 1991; Quinn *et al.* 1992; Connolly and Pearson (in press)).

Consequently, macroinvertebrates are the most commonly used bio-indicator of water quality and aquatic ecosystem health throughout the world, being used as an integral component of most aquatic monitoring programs (Pinder *et al.* 1987, U.K; Plafkin *et al.* 1989, USA; Metcalfe 1989, Europe; Metcalfe-Smith 1994, Europe; Resh *et al.* 1995, USA; Wright 1995, UK; Hawkins *et al.* 2000, Australia; Simpson and Norris 2000, Australia).

4.1.2 Use of macroinvertebrate indices

The widespread use of macroinvertebrates as monitors of water quality and ecosystem health has brought about many ways of designing monitoring programs and analysing macroinvertebrate survey data, and a variety of protocols and indices have been developed (Wright *et al.* 1984; Rosenberg and Resh 1993; Resh *et al.* 1995; Reynoldson *et al.* 1995; Growns *et al.* 1997; Hawkins *et al.* 2000). Over the last two decades methodologies have mainly focused on standardising sampling techniques, often incorporating rapid assessment protocols (Plafkin *et al.* 1989; Barbour *et al.* 1999), and indices have been used as a convenient means to summarise information into single variables to compare sites and rank the degree of disturbance. In Australia, the ‘Australian River Assessment System’ (AusRivAS) (Simpson and Norris 2000) and the ‘Stream Invertebrate Grade Number – Average Level’ (SIGNAL) (Chessman 1995) have become the most widely used schemes. They have been most used in south-eastern Australia because, in northern Australia, limited knowledge of the macroinvertebrate fauna has limited their application.

AusRivAS is a derivative of the ‘River Invertebrate Prediction and Classification System’ (RIVPACS) from the United Kingdom (Wright *et al.* 1984). It was developed as part of the National River Health Program (NRHP) as a means to assess river health at a national level. The scheme is based on a set of statistical models that compare a list of macroinvertebrate families from a sample site (test site) with a database from a number of reference sites that are believed to be least affected by anthropogenic activity. It compares observed with expected values using a set of physico-chemical variables to predict the expected fauna that would occur at a site in the absence of any impacts. The predictors of macroinvertebrate assemblage structure are determined using discriminant function analysis of an extensive dataset of reference sites describing macroinvertebrate distributions in the region. The accuracy of the output depends heavily on the reliability of the reference site data and the ability to discriminate between site groups, so comparisons must be constrained to streams of similar types and from the same region as the reference streams.

SIGNAL is a grading system that scores a site by a single value determined by averaging tolerance/sensitivity scores allocated to each taxon (family) present, essentially weighting the assemblage data by the tolerance of the component taxon to disturbance. It does not depend on comparisons with reference sites, although it is used in this way as an additional feature of the AusRivAS system, which can output an observed-over-expected value for SIGNAL scores based on the predicted macroinvertebrate assemblage. SIGNAL was first developed in 1993 for use in the Hawkesbury-Nepean river system in New South Wales to investigate the impacts of sewage discharge and followed the Biological Monitoring Working Party (BMWP) system used in the United Kingdom (Armitage *et al.* 1983): indeed, the scores for each taxon in the first version of SIGNAL were virtually identical to the scores for the same taxon in the BMWP system. SIGNAL 2 was released in 2003 with many taxon scores modified to reflect Australian conditions, and additional taxa were included (Chessman 2003). SIGNAL was principally designed as an indicator of water quality with the precept that the macroinvertebrate community will shift from an assemblage made up of sensitive taxa in a pristine environment to an assemblage consisting mainly of tolerant taxa in an environment with degraded water quality, which will be reflected in the allocated scores. The accuracy of SIGNAL depends on the correct allocation of scores with the macroinvertebrate taxa responding as predicted.

4.1.3 Monitoring macroinvertebrates in Queensland

The National River Health Program (NRHP) was implemented in Queensland and both AusRivAS and SIGNAL are currently being used, but without well developed regional models, which restricts their effectiveness. For example, both these indices, using data from the NRHP, struggled to adequately assess the condition of reaches in the Burdekin River

(Grinter *et al.* 2000; Connolly 2006) and the Mary River (Connolly 2003) during a condition assessment undertaken for water resource planning. In fact, the use of these indices masked some effects and was misleading in many cases. The problem was not so much that the indices were themselves badly conceived but more that their application was ineffective. It was not so surprising that very little could be derived from the information provided in these assessments, given that very few small-scale samples were expected to describe complex patterns over large areas, using indices based on models derived in very different bioregions.

This example is not unusual and highlights the reality that monitoring programs are being poorly designed and little effort is being put into checking that the data being collected is sufficient to achieve the goals of the programs. Instead, many programs use data that are insufficient to detect the differences they are investigating. Unfortunately, poor results do not always lead to improvement of subsequent sampling designs, so these errors are frequently repeated.

The poor or inconsistent performance of monitoring programs in Queensland has prompted programs specifically aimed at developing and testing methods to be used to measure ecosystem health in Queensland streams. One example was the Design and Implementation of Baseline Monitoring – Stage 3 (DIBM3) (Smith and Storey 2001). The main aim of that program was to develop a cost-effective, coordinated Ecosystem Health Monitoring Program (EHMP) for south-east Queensland, which is currently being applied. Part of the DIBM3 program was designed to identify suitable indicators of ecosystem health for use in the EHMP. The macroinvertebrate indices proposed for inclusion in the EHMP were: PET (Plecoptera, Ephemeroptera and Trichoptera) richness, SIGNAL index and total richness (at the family level) (Marshall *et al.* 2001).

4.1.4 Data for the wet tropics

The wet tropics is a unique bioregion that supports the greatest biodiversity of any region in Australia. It is also a major agricultural area with cattle production and mixed cropping and horticulture on the Atherton Tablelands, and extensive sugar cane production in the lowlands. Extensive clearing of the lowlands has occurred and most streams are modified, with major loss of riparian vegetation, bank destabilisation, and contamination by agricultural chemicals. Agricultural activity has been implicated as a major threat to water quality and ecosystem health on the Great Barrier Reef (Baker et al 2003).

Information on the macroinvertebrate fauna in the wet tropics is extensive compared with other bioregions in Queensland (Pearson *et al.* 1986; Pearson 1994; 2005). Patterns of diversity have recently been reviewed (Connolly *et al.* (in press)), but this review noted that there is little data on the lowland streams in the wet tropics. Effort has concentrated on the uplands and rainforest streams, mainly because of the interest in the Wet Tropics World Heritage Area rainforests and through support from the Rainforest CRC. Queensland's Department of Natural Resources and Water (DNRW) have collected data on lowland streams but this has not been published, and no AusRivAS or other predictive models have been developed specifically for this bioregion.

4.2 Aims

The 'River Health Assessment Tools' task of the *Catchment to Reef* program was developed with similar intentions to the *Design and Implementation of Baseline Monitoring – Stage 3* (DIBM3) (Smith and Storey 2001), but with a focus on streams in the wet tropics bioregion. The goal was to determine the best methods to measure ecosystem health in wet tropics streams using a coordinated combination of ecosystem components as indicators, including fish, macroinvertebrates, aquatic and riparian vegetation and water quality. To achieve this required some baseline data collection in the form of a case study in which each ecosystem component was measured at multiple sites and environmental conditions and disturbances were measured in detail. We chose the Russell and Mulgrave catchments because they offered the opportunity of a paired catchment study where land-use practices differed substantially between two adjacent but very similar catchments. This report describes the results from the macroinvertebrate component of that case study.

For this component we aimed to describe the patterns of macroinvertebrate distributions in selected lowland streams of the wet tropics and to obtain baseline data that could be used to develop predictive models of macroinvertebrate composition. We aimed to associate the response of macroinvertebrate assemblages to the physico-chemical environment and anthropogenic disturbance in these streams, and to determine methodologies for surveying macroinvertebrates to monitor these impacts. How the macroinvertebrate assemblages were affected by variations in riparian vegetation and land-use practices was determined by contrasting sites that differed in a variety of bio-physical characteristics.

Our approach involved describing natural gradients and responses to anthropogenic disturbance and using sites of known condition as a benchmark to evaluate the ability of macroinvertebrate indices to discriminate differences between sites. We tested several macroinvertebrate indices, including species richness, family richness, SIGNAL 2 and PET (the number of species of the orders Plecoptera, Ephemeroptera and Trichoptera).

The specific objectives were to:
- describe in detail the distribution of macroinvertebrate species in four streams in the Mulgrave and Russell catchments;
- pay special attention to the longitudinal gradient in these streams because their physical characteristics change rapidly as they flow from the foothills and across the floodplain: it was expected that the macroinvertebrate assemblage would respond strongly to this physical gradient; therefore, to compare streams, this gradient needed to be accounted for in the sampling design and analysis;
- classify sites within these streams on the basis of their macroinvertebrate assemblages, and identify the key variables driving macroinvertebrate distributions;
- assess the influence of anthropogenic disturbances, such as riparian vegetation clearing, by accounting for natural variations and gradients;
- design river health monitoring protocols for wet tropics streams, using this case study, for further subsequent testing prior to inclusion in a monitoring manual.

4.3 Methods

4.3.1 Study Sites

The study area was treated as a paired-catchment comparison with two streams sampled in both the Russell and Mulgrave catchments. The study was restricted to sections of the stream situated on the floodplain, from the base of the range to the confluence with the Mulgrave or Russell rivers.

The Russell and Mulgrave catchments are similar in topography and climate but land use and the riparian condition within them differ markedly (see Chapter 2). However, in both catchments, as for most of the wet tropics, the floodplain has been extensively converted to agricultural use, dominated by sugar cane production.

The two streams sampled in the Mulgrave catchment were the Little Mulgrave River and Behana Creek; the two streams sampled in the Russell catchment were Woopen Creek and

Babinda Creek. All four streams rise in pristine rainforest within the ranges of the Wet Tropics World Heritage Area, then flow across a narrow floodplain. The Little Mulgrave River and Behana Creek had generally intact and continuous riparian vegetation, whereas in Woopen and Babinda Creeks the riparian vegetation was highly disturbed, dominated by invasive grasses and weeds such as Singapore daisy, with large trees occurring only in occasional patches (see Chapter 3). A detailed description of the study area and the hydrology, geomorphology and water quality of the study streams has been provided in Chapter 2 of this report. The data indicate that the physico-chemical conditions in Woopen and Babinda Creeks are degraded compared with the Little Mulgrave River and Behana Creek.

Forty sites were sampled for macroinvertebrates in the four streams. Strong naturally occurring gradients occur in physical and biological parameters along these streams and anthropogenic influences also increase with distance downstream (Chapter 2). Therefore, it was necessary to sample with adequate intensity to account for these gradients in subsequent analyses. Consequently, sample site locations were distributed at approximately 1 to 1.5. km intervals along each stream from the base of the foothills to the confluence with the Mulgrave or Russell rivers (see Chapter 2).

4.3.2 Macroinvertebrate sampling

Macroinvertebrates were collected by taking large aggregate samples from across the riffle being sampled. The aim was to collect a large number of individuals from the variety of microhabitats within the riffle. It was necessary to sample a large area of the riffle and to collect a large number of individuals per sample to maximise the likelihood of collecting all of the species pool at a site, and thereby to enable confident comparisons of species richness and assemblage composition between sites.

This method did not provide a quantitative measure of density (of species or individuals). Sampling effort was estimated from the time taken to collect the sample and by the amount of material collected. Repeatability of the species richness estimates given by this method is best guaranteed by sampling a relatively large area throughout the riffle, to overcome the inherent patchiness of macroinvertebrates and to include all microhabitats, and by collecting a large number of individuals in each sample. A large number of individuals are needed because the number of macroinvertebrate species in a sample is determined by the number of individuals collected, but the rate of accumulation of species varies depending on the frequency distribution in the assemblage and their spatial distribution in the habitat and, therefore, will not be consistent under different conditions. To properly compare species richness, sufficient numbers of individuals must be collected to ensure that most species are represented at each site. This method was appropriate for the types of statistical analyses being performed and was necessary for this baseline survey to detect the full extent of species distributions in the study area within the constraints of a one-off survey.

Further, because sediment size varied by orders of magnitude among the sample sites (Chapter 2), the efficiency of quadrat sampling, or other area-based methods, would differ between sites in a way that could bias sample estimates. For example, the efficiency of a quadrat sample would differ between sites with substratum consisting of boulders and cobbles and sites with substratum consisting of gravels and sands, because the surface area of each substratum would differ significantly, and because of difficulties using quadrats in coarse substrata.

Five replicate samples were collected in both riffle and edge habitats at each site, but only the results of one replicate taken in the riffle habitat at each site are presented here. The analysis used in this report did not require replication within sites. The analysis of sampling effort (number of individuals) showed that we had sampled adequate numbers of individuals to reliably represent the macroinvertebrate assemblages at the study sites with one replicate sample. If we had found that the number of individuals in a single sample was not adequate to reliably represent the assemblage (e.g., the rate of species accumulation suggesting a higher number of species may have occurred at a site) then additional samples would have been required. However, further analysis and development of predictive models may benefit from the inclusion of additional site replicates. Because edge habitats were very inconsistent between sites, comparisons were difficult and no results are presented here.

Each sample was collected using a triangular dip net with a 210 µm mesh. The flat base of the net was pressed into the substratum facing upstream into the flow. The substratum was disturbed upstream of the net by vigorously brushing the substratum with a bare hand, causing dislodged material to be washed into the net. During the collection of each sample the net was relocated at a number of randomly chosen positions throughout the riffle for a period of approximately 10 minutes, resulting in a large aggregate quantity of material collected over the full extent of the riffle. Sampling commenced at the downstream section of the riffle and progressively moved upstream in a wide zigzag pattern, crossing the full width of the riffle.

The material collected in the dip net was washed into a 1 L plastic container, fixed in 80% ethanol and returned to the laboratory for processing. In the laboratory, samples were washed through 1 mm and 210 µm sieves. Material collected on these sieves was sorted under a magnifying lamp and macroinvertebrates were separated and identified under a stereo dissector or high-power microscope. Remaining coarse particulate organic matter (CPOM) was dried and weighed. Macroinvertebrates of the orders Ephemeroptera and Trichoptera were identified to species and their identifications confirmed by a recognised taxonomist for each group. Other orders were identified to the highest taxonomic level possible or allocated to Operational Taxonomic Units (OTUs) where morphological differences were obvious but no taxonomic keys were available.

4.3.3 Environmental variables – Physico-chemical parameters

The physico-chemical environment at the study sites varied within streams and between streams both naturally, because of natural geomorphologic and other environmental gradients, and as a result of anthropogenic influences associated with land use, and changes in vegetation (see Chapters 2 and 3). Landscape variables were determined using available GIS data coverage for the study area (Chapter 2). Geomorphologic measurements were made at each site concurrently with macroinvertebrate sampling (Chapter 2). Other physico-chemical parameters were measured at each site when aquatic and riparian vegetation was surveyed, although these parameters were only measured at 34 sites (Chapter 3). Water quality parameters were measured at 28 study sites and at multiple sites along the length of the Little Mulgrave River, and Behana and Babinda Creeks by navigating these by canoe (Chapter 2). Water quality samples were processed in the Australian Centre for Tropical Freshwater Research analytical laboratory at James Cook University, Townsville.

The results of the surveys were combined into an Environmental Variables matrix for analysis with the macroinvertebrate assemblage data. Table 4.1 describes the environmental variables used in these analyses and provides codes used to label these variables in subsequent figures.

4.3.4 Statistical Analysis

Classification and *Ordination*

Classification and ordination were used to investigate the similarity of macroinvertebrate assemblages at study sites and to describe the spatial patterns of macroinvertebrate assemblages in the study area. All data were range standardised using the formula $D_{ij} - D_{min}/D_{range}$ prior to analyses. The Bray Curtis dissimilarity measure was used to calculate an association matrix of dissimilarities between sites (Faith *et al.* 1987).

An agglomerative hierarchical classification was generated by means of Flexible Pair-Group Method using arithmetic Averages (UPGMA) with $\beta = -1$. This was done for macroinvertebrate assemblage data for each stream separately and for data combined from all samples collected in all four streams.

Site groups identified by the UPGMA classification were confirmed by ordination of the association matrix using Semi-Strong-Hybrid Multidimensional Scaling (SSH MDS) (Belbin 1995). Principle Axis Correlation (PCC) was used to correlate environmental variables with ordination space determined by SSH MDS (Belbin 1995). The likely significance of correlation coefficients produced by PCC was tested using a Monte-Carlo Attributes and Ordination (MCAO) procedure in PATN (Belbin 1995) with 1000 random permutations.

Univariate relationships

The relationship between species richness and other metrics calculated from the macroinvertebrate assemblage data was analysed using linear regression analysis. Differences between groups were tested using analysis of covariance (ANCOVA), with mean sediment size (measured using the logarithmic Phi scale (Wentworth 1922)), used as a covariate. Type III sums of squares was used in ANCOVAs and significance value was set at $p \leq 0.05$.

Testing indices

Significant ANCOVA models for Behana and Babinda creek comparisons, using mean sediment sizes of riffles as a covariate, were used to test the performance of several macroinvertebrate assemblage indices, namely species richness, family richness, SIGNAL 2 (Chessman 2003 a, b) and PET (total number of species of Plecoptera, Ephemeroptera and Trichoptera). The ability of the various indices to detect differences between streams was judged by the significance value (F ratio and p value) of the ANCOVA model and by comparing the location of site values relative to the prediction intervals around the regression model for Behana Creek. The prediction intervals were calculated using the following formula, $y_0 \pm t(n - p - 1).s.\sqrt{1 + X_0^1(X^1X)^{-1}X_0}$ (using SigmaPlot 2002 for Windows Version 8), and are the confidence intervals for the population (i.e., the range where the data values (sites in our case) will fall 95% of the time for repeated measurements of the same population). An index was considered able to detect differences if the significance level of the ANCOVA was < 0.05 and sites from Babinda Creek fell outside of the prediction intervals for the Behana Creek regression.

No AusRivAS models were available for the wet tropics streams so the AusRivAS index could not be tested.

Testing sample size

The effect of sample size on the ability of the univariate assemblage indices to detect differences between streams of different condition was tested by generating random subsets of the sample assemblages for Behana and Babinda Creeks, where the number of individuals per sample was constrained to 50, 100, 250, 500 and 1000 randomly chosen individuals.

Table 4.1. Summary of environmental variables used in analyses.

Parameter	Unit	Abbreviation
Catchment and Land use		
Catchment area	km^2	CATAREA
Distance from uppermost site	km	DIST
Elevation	m.a.s.l.	ELEV
Conservation Areas	%	CONSERV
Sugar Cane	%	SUGAR
Other Horticulture-Cropping	%	OTH_CROPS
Grazing	%	GRAZE
Plantation	%	PLANTAT
Residential-Rural res	%	RESID
Industrial and Commercial	%	INDUST
Reservoir	%	STORAGE
Riparian canopy cover	%	RIPCOV
Riparian condition total score		RIPSCORE
Water Quality		
Dissolved Oxygen	Ppm	DO
Conductivity	$\mu S\ cm^{-1}$	COND
pH	pH units	PH
Water Temperature	^oC	TEMP
Turbidity	NTU	TURB
Ammonia	μgL^{-1}	NH3
Oxides of Nitrogen	μgL^{-1}	NOX
Total Nitrogen	μgL^{-1}	TN
Total Phosphorus	μgL^{-1}	TP
Filterable Reactive Phosphorus	μgL^{-1}	FRP
Hydraulic		
Water slope	(%)	SLOPE
Width	m	WIDTH
Depth	m	DEPTH
Water velocity	ms^{-1}	VELOC
Median particle size	mm	D50
Substrata composition	% of quadrat	MUD, SAND, FINEGR, GRAV, COBBLE, ROCK, BEDROCK
Froude number		FROUDE
Reynolds number		REYNOLD

Only the results for species richness are presented in this report. The effect of sample size was not tested for other indices at this stage. Species richness was found to be the most sensitive index (see below) so it was assumed that the level of sampling effort required to detect differences using other indices would be the same or greater.

The procedure used was equivalent to determining a 'rarefied' sample at each level of sampling effort. The specified numbers of individuals were randomly selected from the whole assemblage data for each sample as follows. Species were randomly drawn from the species pool, weighted with a probability equal to the abundance of that species in the observed sample pool, until there were *n* individuals in the random sample. The sampling technique sampled with replacement (i.e. if an individual of one species was drawn from the total pool, it did not affect the chance of another individual of the same species being drawn). This allowed us to draw random samples that had a total abundance greater than that observed. At large sample sizes, sampling species with replacement gives essentially the same result as sampling individuals without replacement. Our observed sample sizes ranged from just under 500 individuals to nearly 2000 individuals. Results were aggregated to present the number of species that occurred in the sub-sample, and the number of individuals of each species in the sub-sample for each site. This was repeated for 1000 randomisations and the mean and 95% confidence intervals were generated for each of the sampling effort levels.

This 'rarefied' data was then used in the ANCOVA model comparing Behana and Babinda Creeks, and the data from different sampling efforts were compared by determining the ability of each sampling effort level to detect differences between these two streams in the same way as for testing between indices, described above.

4.4 Results

A total of 118 invertebrate taxa were collected in the four streams sampled. Not all taxa were identified to species level – for example, the Chironomidae were only identified to family – so it is clear that the actual species diversity was considerably higher than 118. Of special interest were records of two new species: a confirmed new species of Ephemeroptera (Leptophlebiidae WT sp.6) and a confirmed new species of Psephenidae (*Sclerocyphon* sp.). There were high numbers of species of Leptophlebiidae mayflies (13 species), the trichopteran families Hydropsychidae (11 species), Philopotamidae (5 species) and Leptoceridae (5 species), and the beetle family Elmidae (> 20 species). The assemblages were dominated numerically by high numbers of individuals of a few taxa. The Baetidae mayflies were generally very abundant as was *Austrophlebioides* sp. (Leptophlebiidae). Other abundant taxa included the Chironomidae and Simuliidae and some of the trichopteran species, for example, *Cheumatopsyche* sp. AV15, *Cheumatopsyche* sp. AV16 and *Chimarra* sp. AV5. These taxa were generally most abundant at sites located in the mid reaches of each stream, whereas the Elmidae, particularly *Austrolimnius* Type A, were very abundant at sites located in the lower reaches of Behana and Babinda creeks.

The physical character of all four study streams changed gradually along their length as they flowed out from the foothills and across the floodplain. The most notable change in the riffle habitat was the gradual reduction in the grain size of substratum sediments, reducing from large boulders in the uppermost sites at the very base of the range, to smaller cobbles in mid reaches, and eventually to coarse gravels and sands in the lower reaches of Behana and Babinda Creeks (Figure 4.1). Chapter 2 provides a more detailed description of sediment sizes and general geomorphology of the streams.

The gradient in the macroinvertebrate assemblages was correlated with the changes in riffle sediment sizes along the stream. A strong upstream-downstream gradient in macroinvertebrate composition was apparent for all four streams, with a decline in taxon richness downstream and away from the range (Figure 4.2). The decline in the number of taxa with distance downstream was most striking in Behana and Babinda Creeks, where taxon richness changed by a factor of four between the most upstream and most downstream sites. The Little Mulgrave River and Woopen Creek were much shorter than Behana and Babinda Creeks, with the upstream-downstream gradient truncated, essentially making these streams only comparable with the upper reaches of the two longer streams.

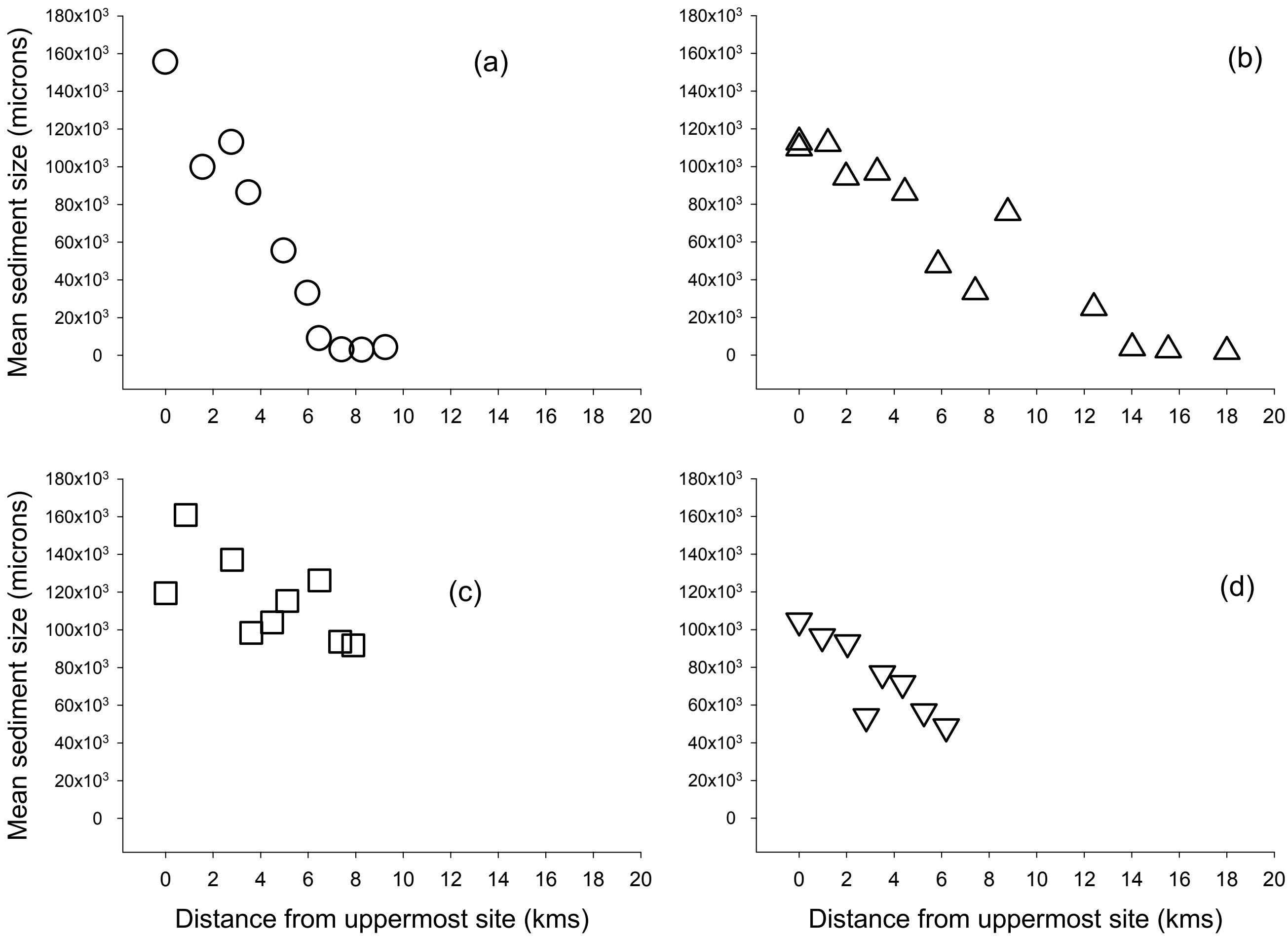

Figure 4.1. The mean substratum particle sizes recorded at sample sites in relation to the distance from the uppermost site, positioned at the base of the range, for each of the four streams surveyed: (a) Behana Creek (b) Babinda Creek (c) Little Mulgrave River (d) Woopen Creek.

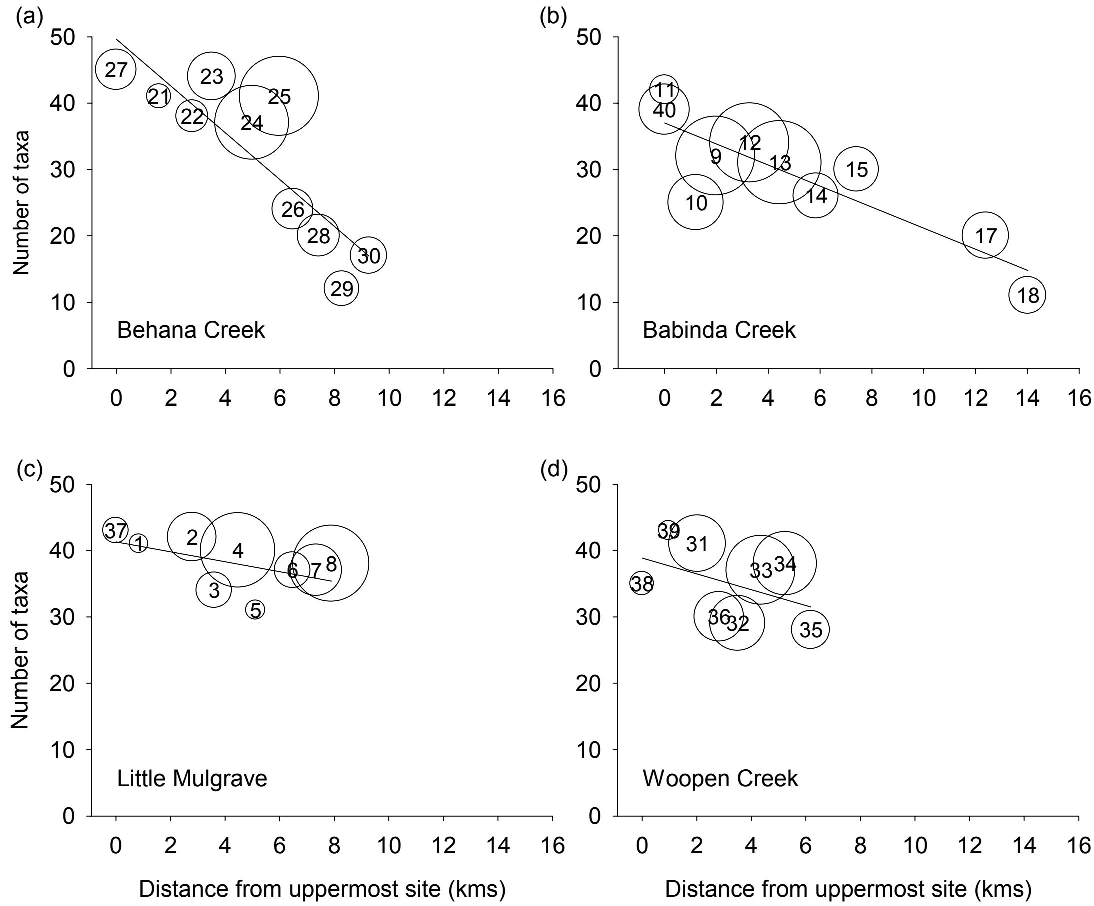

Figure 4.2. The number of macroinvertebrate taxa recorded at sites in relation to the distance from the uppermost site, positioned at the base of the range, for each of the four streams surveyed: (a) Behana Creek (b) Babinda Creek (c) Little Mulgrave River (d) Woopen Creek. Sizes of bubbles are proportional to the number of macroinvertebrates collected at each site, indicated by the key. Numbers in bubbles indicate site numbers.

4.4.1 Site 16

The substratum at Site 16 in Babinda Creek had been modified by the introduction of large cobbles and boulders close to a railway bridge. The substratum sediment sizes in this reach were otherwise quite fine, consisting of gravels and sand. The introduction of larger particles created an anomaly in the substratum particle sizes at that site relative to its position along the stream length (Figure 4.1b). Consequently, it was not plotted in Figure 4.2b, and does not contribute to the regression line shown. However, the macroinvertebrate assemblage responded to the presence of the larger particle sizes with several species being present (e.g., several of the Hydropsychidae and Leptophlebiidae) that were otherwise absent or in very low numbers at adjacent sites, in reaches with only smaller sediment particle sizes. These taxa contributed to a high diversity at this site, with 38 taxa recorded. However, Site 16 did group with adjacent sites with UPGMA classification and in a position relative to the upstream-downstream sequence of sites (see below), thus having components of the assemblage consistent with these downstream sites.

4.4.2 Classification and Ordination

Classification of sites using the macroinvertebrate assemblage data grouped the sites from the four streams in an upstream-downstream order, with the longitudinal sequence of site positions generally maintained (Figure 4.3). Sites grouped according to upper-, mid- and downstream stream sections, confirming the strong longitudinal gradient in macroinvertebrate assemblage composition along these streams. The upstream-downstream sequence was maintained when data from different streams were combined (Groups 1, 2 and 3 respectively in Figure 4.4). However, within groups 2 and 3 sites clustered according to catchment and stream.

Ordination of sites using the macroinvertebrate assemblage data confirmed the site groups identified in Figure 4.4, with site groupings corresponding to their locations along the streams (Figures 4.5 and 4.6). Principal Axis Coordinates showed that the ordination correlated strongly with sediment particle sizes (Table 4.2), with sites located along a gradient in sediment sizes (Figures 4.5b and 4.6b). The direction of this gradient is described by the mean sediment size vector that had the highest correlation coefficient value (r^2 = 0.862). The ordination also correlated strongly with the number of taxa recorded at sites, as well as the occurrence or relative abundance of several taxa (Figures 4.5a and 4.6a, and Table 4.3).

4.4.3 Group 3 – downstream sites

In Figures 4.5 and 4.6 the downstream sites in Behana and Babinda Creeks, identified as Group 3 in Figure 4.4, separated from other sites most clearly. These were the sites where riffle substratum particles consisted of sands and gravels (Figure 4.1). Except for the anomalous Site 16, many otherwise abundant taxa were absent from these downstream sites where larger particle sizes were absent. Across the survey area, abundances of Trichoptera were very low at sites where the mean sediment particle size was less than 40 mm and all of the Trichoptera, except for *Oecetis* spp., were absent at sites with mean sediment particle sizes less than 30 mm. In contrast, *Oecetis* spp. appeared to have a preference for sites with smaller, sandier substratum.

The Leptophlebiidae were absent from sandy sites but the distribution of *Austrophlebioides* sp. extended to Site 17 in Babinda Creek, with a mean sediment particle size of 25 mm. In contrast, the caenid and the baetid mayflies were present in high numbers at the downstream sites.

Several species responded to the availability of larger sediment particles at Site 16 in Babinda Creek. This response was most notable for the Hydropsychidae, *Cheumatopsyche* sp. AV15, *Cheumatopsyche*. sp. AV8 and *Chimarra* sp. AV5. Conversely, abundances of Baetidae and *Austrolimnius* Type A were lower than might have been expected relative to adjacent sites and the position along the stream.

Although many taxa were absent from the downstream sites in Behana and Babinda Creeks, these sites contained very high abundances of *Austrolimnius* Type A and high abundances of caenid mayflies. Both of these taxa occurred throughout the stream but increased in abundance towards downstream sites. Other taxa, such as Ceratopogonidae, *Graphelmis* sp., Tipulidae sp.10 and Belostomatidae were only found at downstream sites in Babinda and Behana Creeks and were not recorded in the Little Mulgrave River or Woopen Creek.

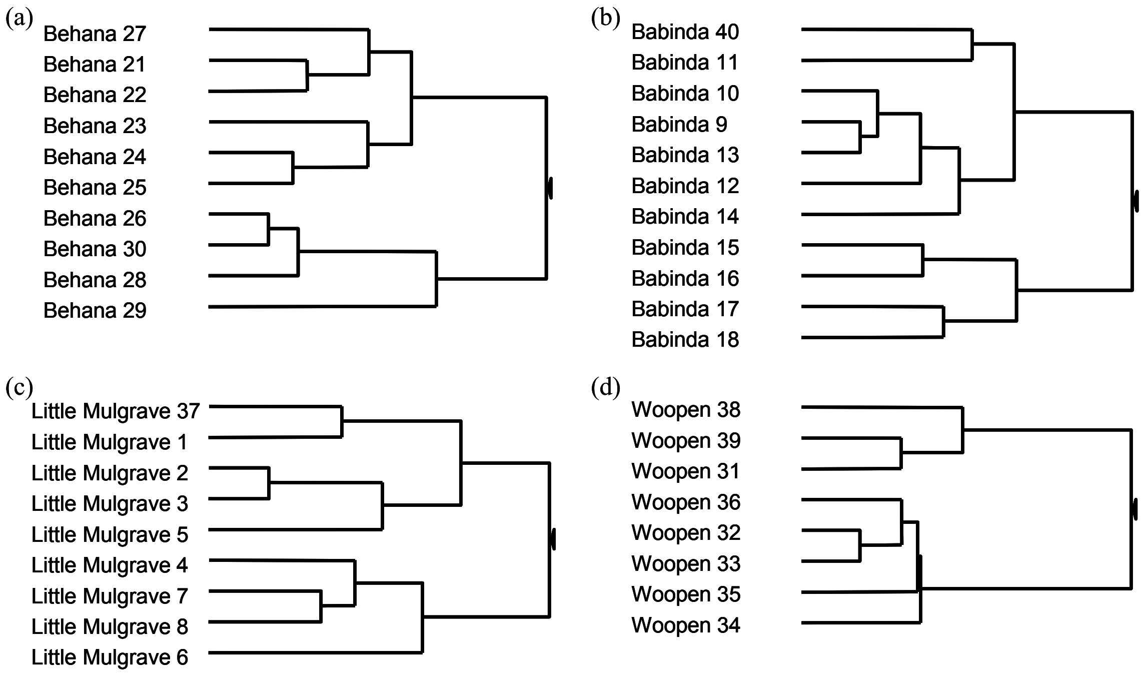

Figure 4.3. A hierarchical classification of sites in each of the four streams surveyed based on macroinvertebrate assemblage data. (a) Behana Creek (b) Babinda Creek (c) The Little Mulgrave River (d) Woopen Creek. Generated using range-standardised data and Flexible Pair-Group Method using arithmetic averages (UPGMA) with $\beta = -1$.

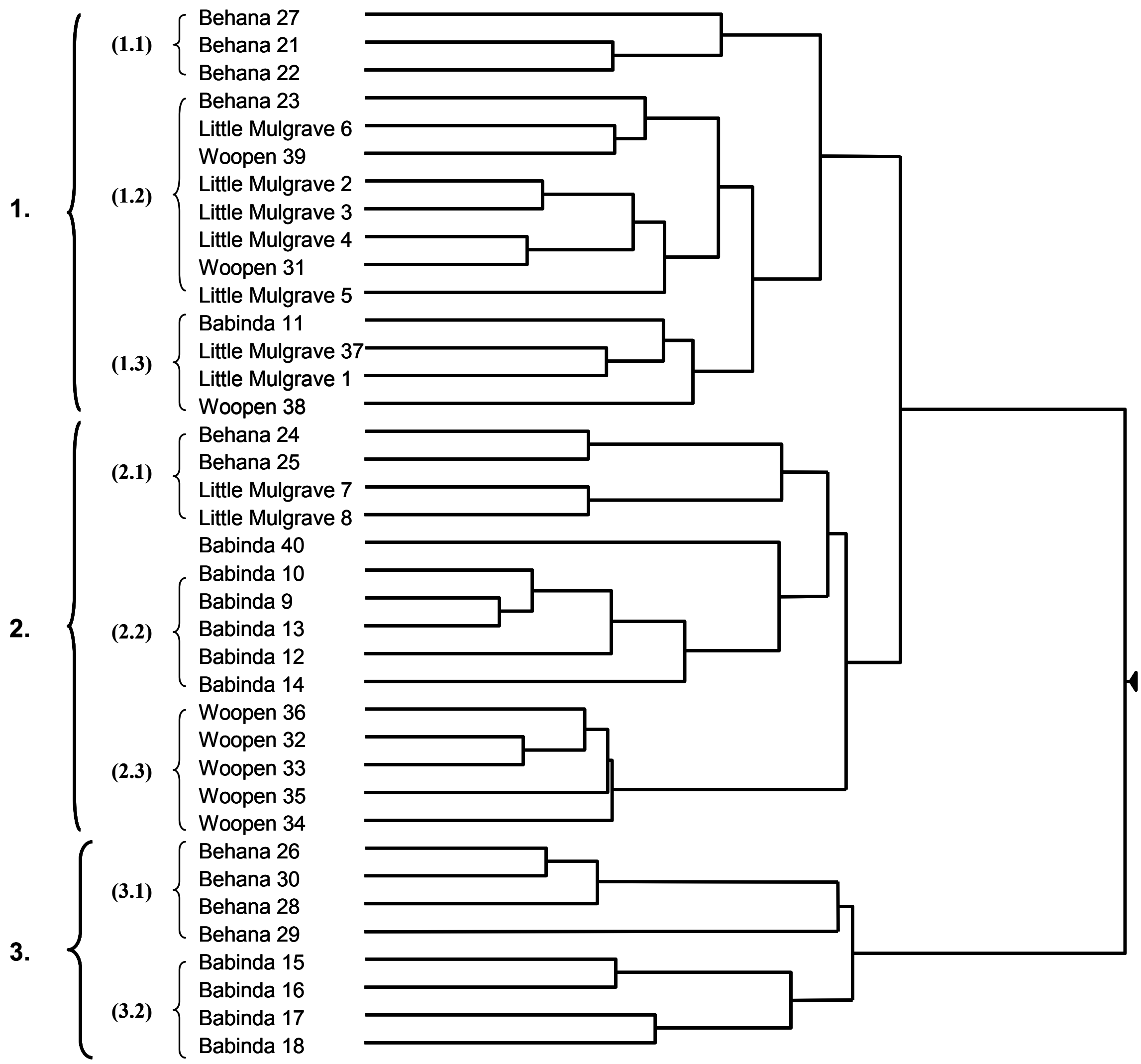

Figure 4.4. A hierarchical classification of sites in all of the four streams surveyed based on macroinvertebrate assemblage data. Generated using range-standardised data and Flexible Pair-Group Method using arithmetic averages (UPGMA) with $\beta = -1$.

Table 4.2. Correlation coefficients of environmental vectors from Principal Axis Coordinates Analysis (PCC) carried out on semi-strong hybrid multidimensional scaling (SSH MDS) ordination in three dimensions and with a cut-off value of 0.835 using range-standardised species abundance data. Vectors that were determined as significant in the Monte-Carlo Attributes and Ordination (MCAO) procedure in PATN after 1000 random permutations are shown. % Medium Boulders and Riparian cover are also included because they are potentially important parameters and were marginally non-significant at the 0.05 level.

Variable	r^2	% Permuted r^2 > Actual r^2	p
Phi MEAN	0.862	0.0	<0.001
Phi D50 (f):	0.846	0.0	<0.001
Geo MEAN	0.841	0.0	<0.001
Geo D50 (micron):	0.813	0.0	<0.001
Distance	0.691	0.0	<0.001
% Sm cobble	0.662	0.0	<0.001
% FINE GRAVEL	0.640	0.0	<0.001
% V COARSE SAND	0.639	0.0	<0.001
% V FINE GRAVEL	0.620	0.0	<0.001
% Lg cobble	0.596	0.0	<0.001
% MED SAND	0.532	0.0	<0.001
% Sm boulder	0.517	0.0	<0.001
% V COARSE SILT	0.504	0.0	<0.001
% COARSE SILT	0.504	0.0	<0.001
% MED SILT	0.504	0.0	<0.001
% FINE SILT	0.504	0.0	<0.001
% V FINE SILT	0.504	0.0	<0.001
% CLAY	0.504	0.0	<0.001
% V FINE SAND	0.488	0.0	<0.001
% FINE SAND	0.486	0.0	<0.001
% COARSE SAND	0.469	0.0	<0.001
CPOM Dry Wt. (g)	0.334	0.6	0.006
% COARSE GRAVEL	0.275	1.2	0.012
Catchment Area (sq. km)	0.223	3.2	0.032
% MED GRAVEL	0.192	5.2	0.052
Geo (St Dev)	0.184	6.2	0.062
% Med boulder	0.176	7.8	0.078
Riparian cover	0.316	8.9	0.089

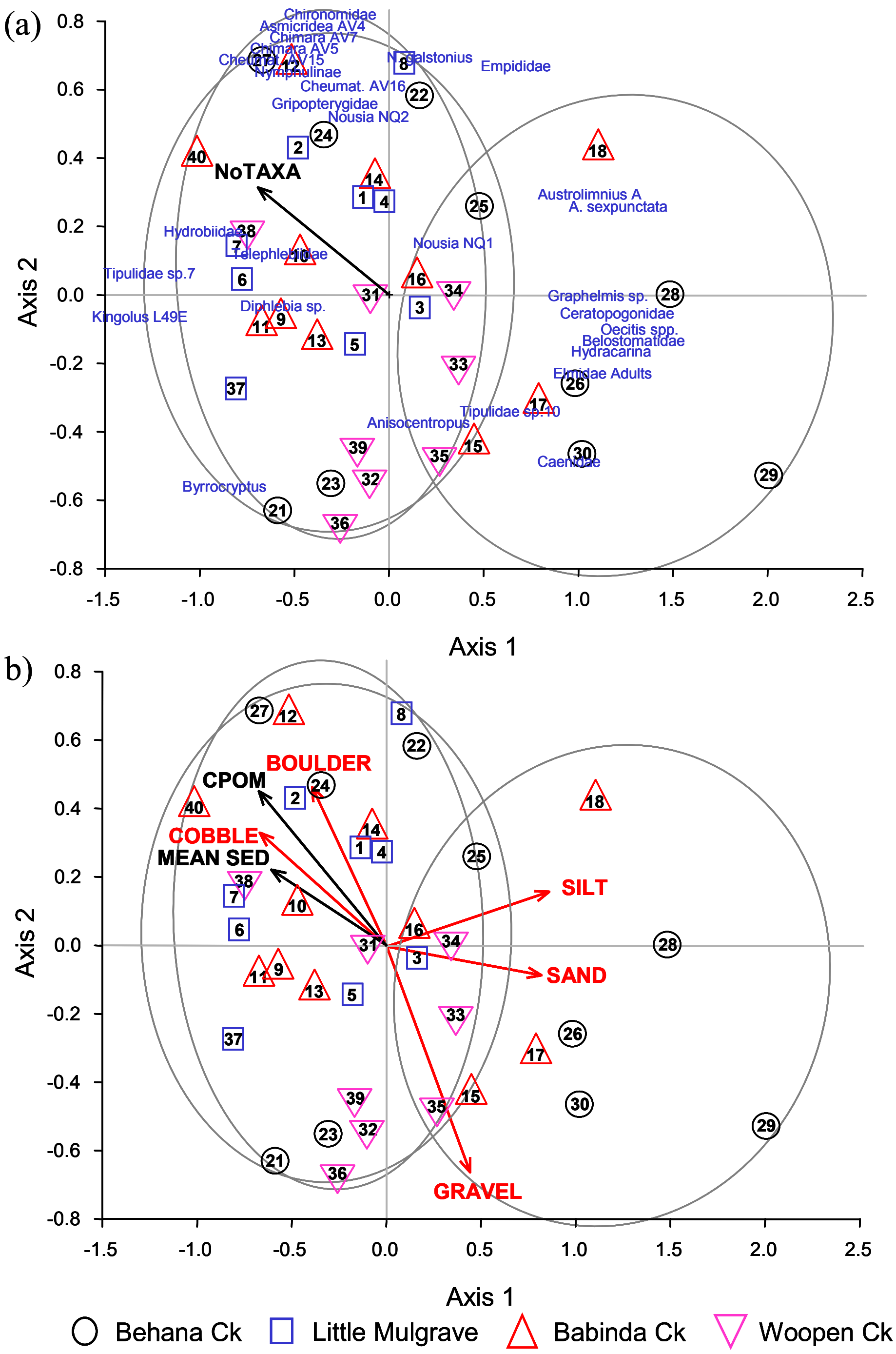

Figure 4.5. Position of sample sites plotted on Axis 1 and 2 of semi-strong hybrid multidimensional scaling (SSH MDS) ordination in three dimensions and with a cut-off value of 0.835 using range-standardised species abundance data. Ellipses refer to the groups 1, 2 and 3 identified in Figure 4.8. (a) Ordination in two dimensions showing direction of taxon vectors and for the parameter species richness, derived by Principal Axis Correlation (PCC). (b) Ordination in two dimensions showing direction of environmental vectors. Only vectors that were determined as significant in the Monte-Carlo Attributes and Ordination (MCAO) procedure in PATN after 1000 random permutations are shown.

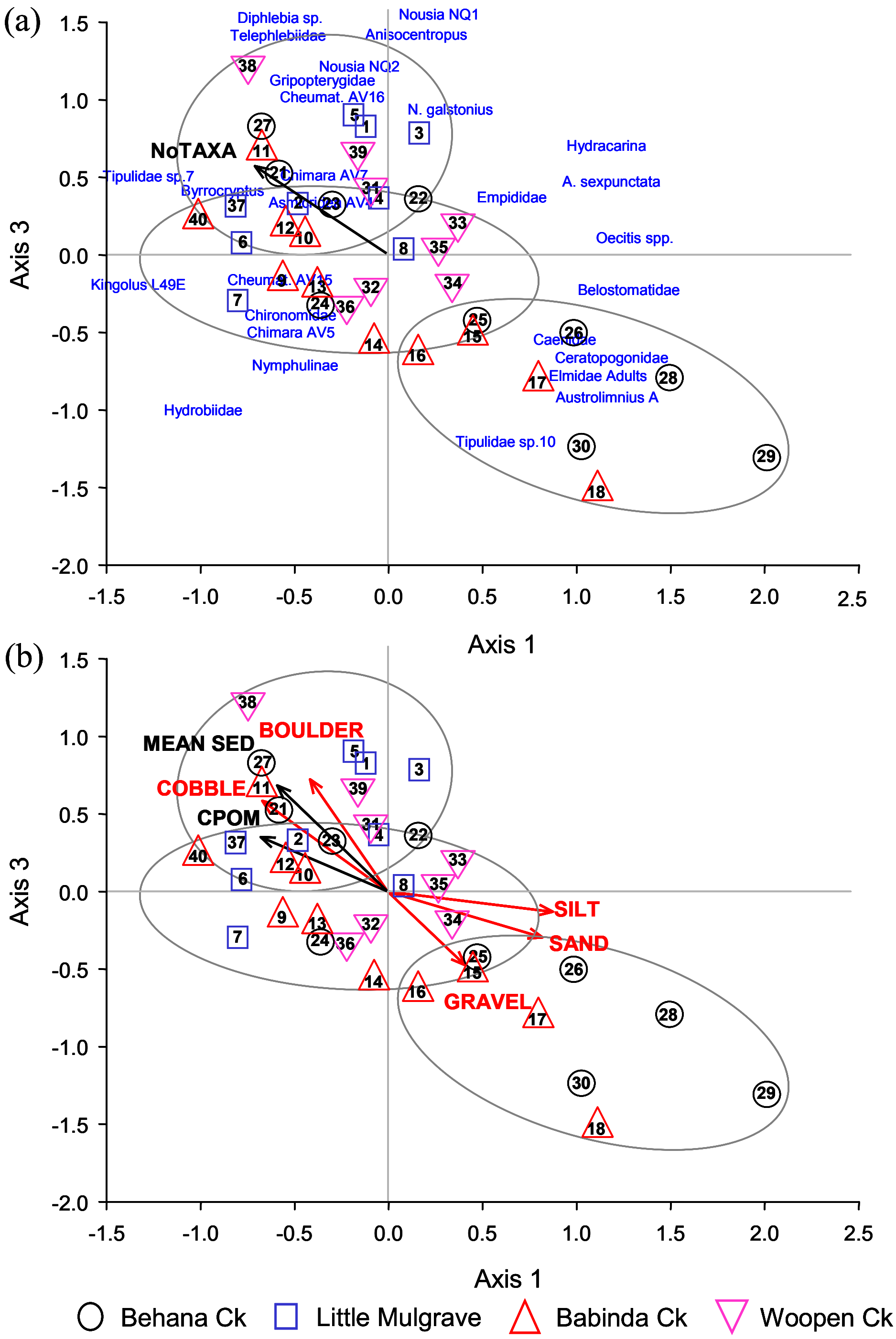

Figure 4.6. Position of sample sites plotted on Axis 1 and 3 of semi-strong hybrid multidimensional scaling (SSH MDS) ordination in three dimensions and with a cut-off value of 0.835 using range-standardised species abundance data. Ellipses refer to the groups 1, 2 and 3 identified in Figure 4.8. (a) Ordination in two dimensions showing direction of taxon vectors and for the parameter species richness, derived by Principal Axis Correlation (PCC). (b) Ordination in two dimensions showing direction of environmental vectors. Only vectors that were determined as significant in the Monte-Carlo Attributes and Ordination (MCAO) procedure in PATN after 1000 random permutations are shown.

Table 4.3. Correlation coefficients of taxa vectors and the parameter species richness from Principal Axis Coordinates Analysis (PCC) carried out on semi-strong hybrid multidimensional scaling (SSH MDS) ordination in three dimensions and with a cut-off value of 0.835 using range-standardised species abundance data. Only vectors that were determined as significant in the Monte-Carlo Attributes and Ordination (MCAO) procedure in PATN after 1000 random permutations are shown.

Variable	r^2	%Permuted r^2 > Actual r^2	p
Number of species	0.636	0.0	0.000
Elmidae adults	0.636	0.0	0.000
Cheumatopsyche sp. AV16	0.565	0.0	0.000
Austrolimnius type A	0.519	0.0	0.000
Anisocentropus sp.	0.456	0.0	0.000
Caenidae	0.445	0.0	0.000
Kingolus L49E	0.435	0.0	0.000
Gripopterygidae	0.391	0.0	0.000
Nousia sp. NQ1	0.375	0.0	0.000
Cheumatopsyche sp. AV15	0.339	0.1	0.001
Chironomidae	0.335	0.1	0.001
Empididae	0.304	0.1	0.001
Byrrocryptus sp.	0.341	0.2	0.002
Ceratopogonidae	0.337	0.2	0.002
Notriolus ?galstonius	0.334	0.2	0.002
Belostomatidae	0.304	0.2	0.002
Diphlebia sp.	0.332	0.8	0.008
Oecetis spp.	0.248	0.9	0.009
Pyralidae: Nymphulinae	0.288	1.3	0.013
Asmicridea sp. AV4	0.269	1.3	0.013
Chimarra sp. AV5	0.266	1.4	0.014
Telephlebiidae	0.238	1.9	0.019
Tipulidae sp. 10 (Str. Class)	0.215	2.3	0.023
Chimara sp. AV7	0.212	2.3	0.023
Tipulidae sp. 7 (Str. Class)	0.204	2.4	0.024
Nousia sp. NQ2	0.24	2.5	0.025
Hydrobiidae	0.207	3.4	0.034
Graphelmis sp.	0.213	4.1	0.041
Conoesucidae Genus Con Bsp. AV1	0.182	4.6	0.046
Hydracarina	0.206	5.0	0.050
Aethaloptera sexpunctata	0.151	5.0	0.050

4.4.4 Group 1 – upstream sites

The Group 1 sites (Figure 4.4) grouped together when plotted on Axis 1 and Axis 3 of the ordination, separating clearly from Group 3, with Group 2 overlapping Groups 1 and 3 (Figure 4.6). Group 1 consisted of sites from the upper reaches of each stream. These sites are positioned in the direction of the vectors for cobbles and boulders and are sites with large sediment particle sizes, and are also the sites with high diversity. Surprisingly, however, Sites 40 and 10 in Babinda Creek classified into Group 2, but their position in the ordination was consistent with Group 1 sites, positioned where Groups 1 and 2 overlap. However, when the classification was carried out using data from Behana and Babinda creeks only, the upper Babinda Sites 40 and 11 classified with the upper Behana Sites 27, 21 and 22 (Figure 4.7).

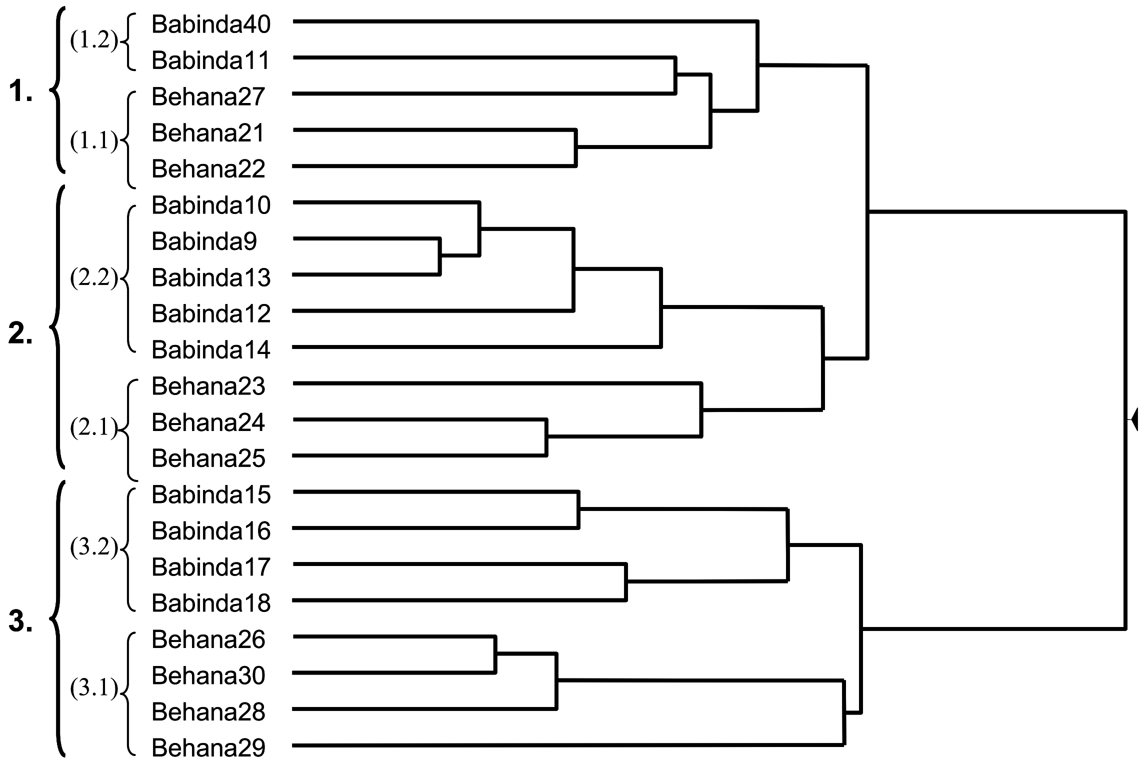

Figure 4.7. A hierarchical classification of sites in Behana and Babinda Creeks based on macroinvertebrate assemblage data. Generated using range-standardised data and Flexible Pair-Group Method using arithmetic Averages (UPGMA) with $\beta = -1$.

4.4.5 Relationships between the number of taxa and mean sediment particle size

The SSH MDS ordination correlated strongly with the number of taxa and the mean sediment ($r^2 = 0.636$ and $r^2 = 0.862$ respectively – Tables 4.2 and 4.3). Both variables were correlated and varied systematically along the stream lengths in all four streams (Figures 4.12(a,b), 4.5 and 4.6).

The relationship between mean sediment size and the number of taxa recorded was logarithmic in form (Figure 4.8a), and when plotted using the $\log_2$ Phi scale to measure sediment particle size, the relationship was approximately linear (Figure 4.8b). This relationship was consistent in all four streams, although the number of taxa differed between streams (Figures 4.9a,b). The relationship between taxon richness and mean sediment size was similar in both Behana and Babinda creeks, but the number of taxa was consistently lower in Babinda Creek sites by about 20-25% (Figure 4.9a). An analysis of covariance, using mean sediment grain size as a covariate, found a significant difference between the number of taxa in these two streams ($F_{1,17} = 17.222$, $p = 0.001$). Site 16 was excluded from this analysis because of its anomalous sediment size.

Figure 4.9b shows that Woopen Creek had smaller sediment particle sizes than the Little Mulgrave River, and a concordant lower number of taxa.

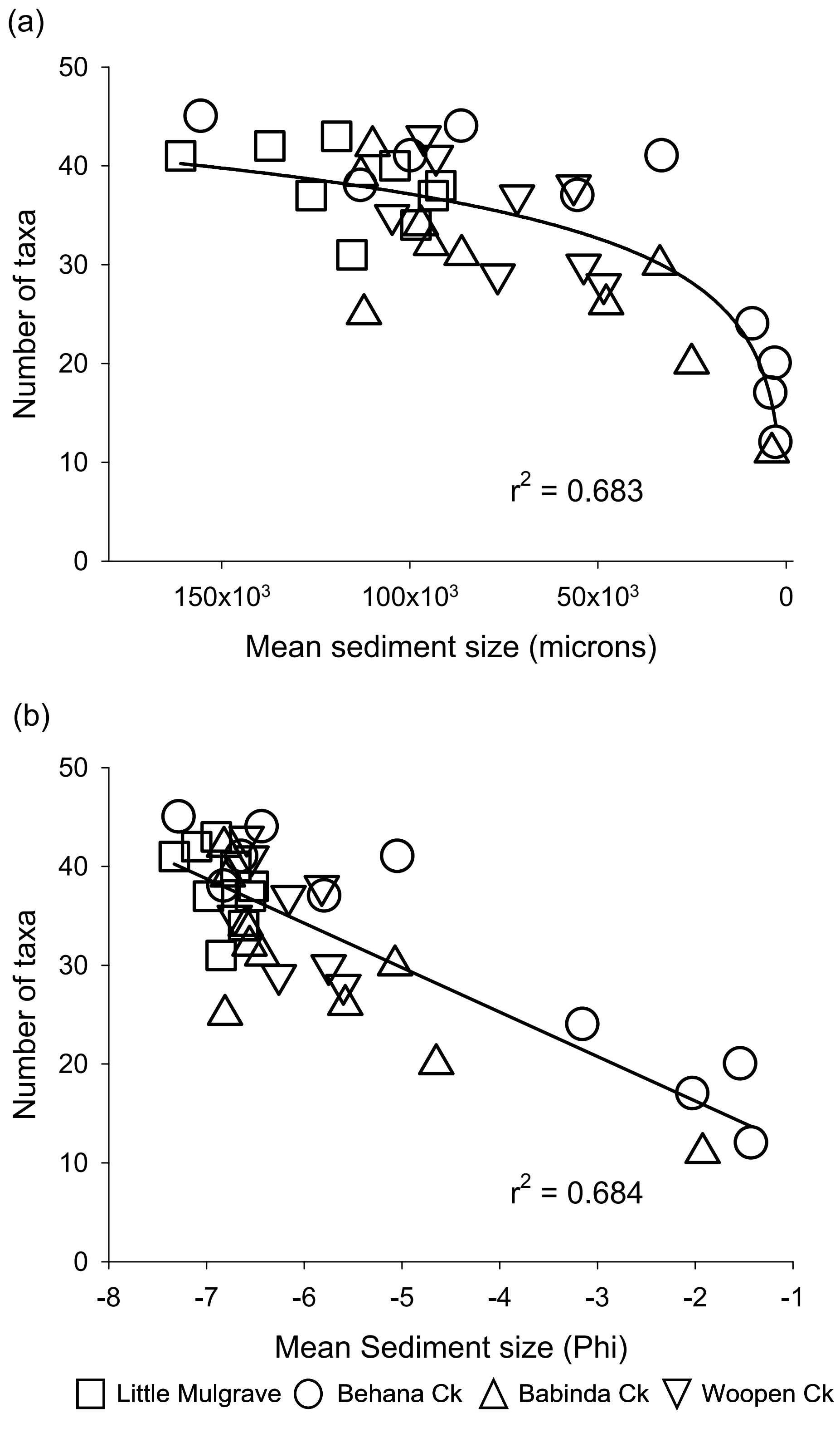

Figure 4.8. The relationship between the mean sediment size and the number of taxa recorded at sites in all four streams. (a) Using mean sediment size measured as microns; regression is based on the equation $y_1 = y_0 + a.\ln(x)$. (b) Using mean sediment size measured using a $\log_2$ scale, Ph; Regression is based on the linear equation $y_1 = y_0 + a(x)$.

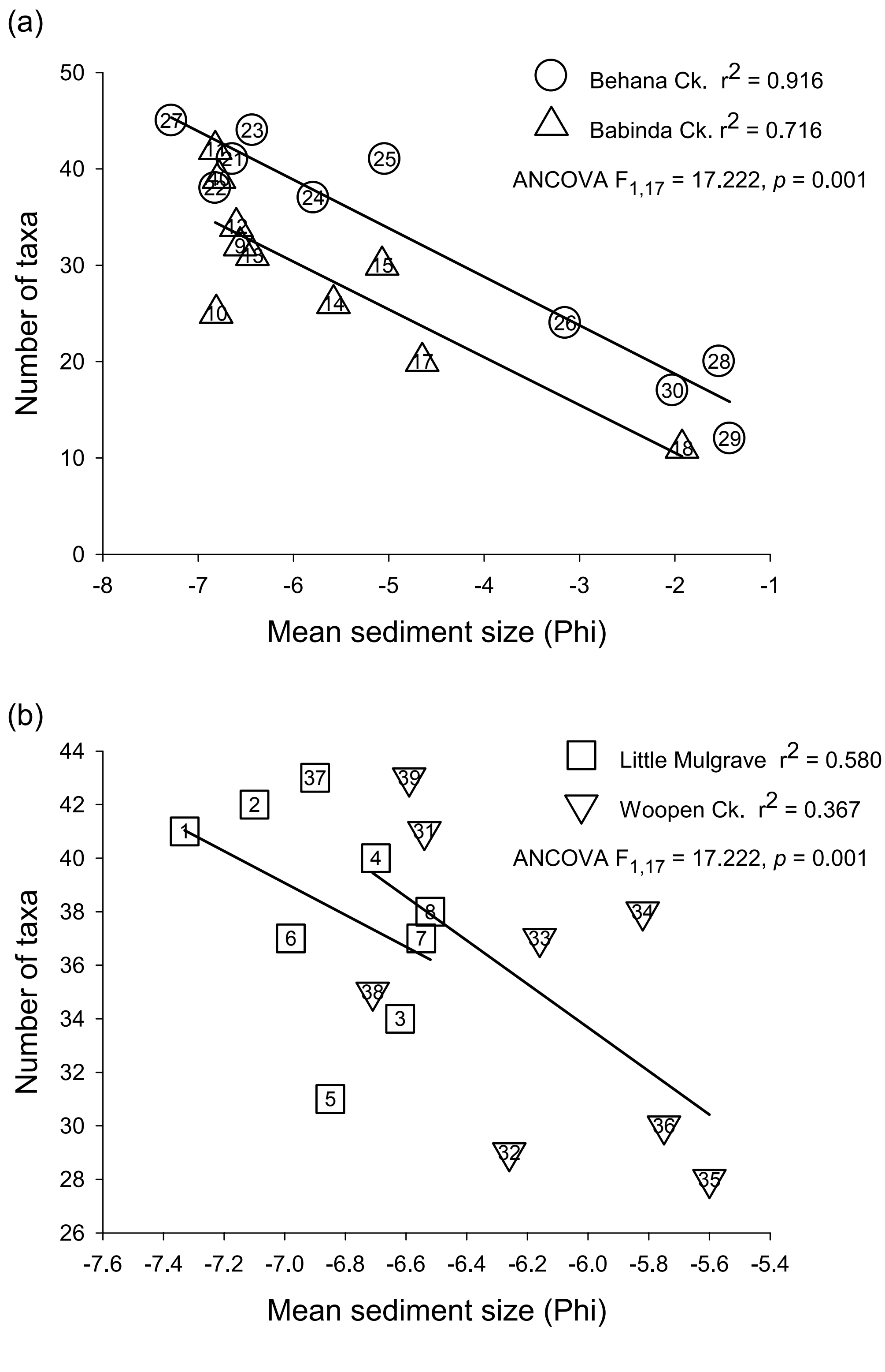

Figure 4.9. The relationship between the mean sediment size and the number of taxa recorded at (a) Behana and Babinda creek sites (b) The Little Mulgrave River and Woopen Creek sites. Regressions are based on the linear equation $y_1 = y_0 + a(x)$. The significant difference between streams is shown by the results of an analysis of covariance (ANCOVA) using mean sediment size (Phi) as a covariate.

4.4.6 Using Behana Creek as a reference

Figure 4.10 compares the diversity in the four streams relative to the diversity in Behana Creek. When the mean sediment particle size was accounted for, the numbers of taxa at sites in Babinda Creek were consistently lower than in Behana Creek, except at the uppermost sites, 40 and 11, which were within the Wooroonooran National Park estate, and upstream from any influence of changed land use. The upstream sites in the Little Mulgrave River fell close to the regression line for Behana Creek but further downstream there was variability, with several sites having a lower number of taxa; however, only Site 5 fell outside the prediction intervals. The number of individuals in the sample from Site 5 was low compared with the numbers in other samples (Figure 4.2c) and this may account for low diversity being recorded at this site. The positions of the sites in Woopen Creek were variable, with several sites positioned close to the regression line for Behana Creek, whereas several others had a lower number of taxa than would be expected from the Behana Creek regression.

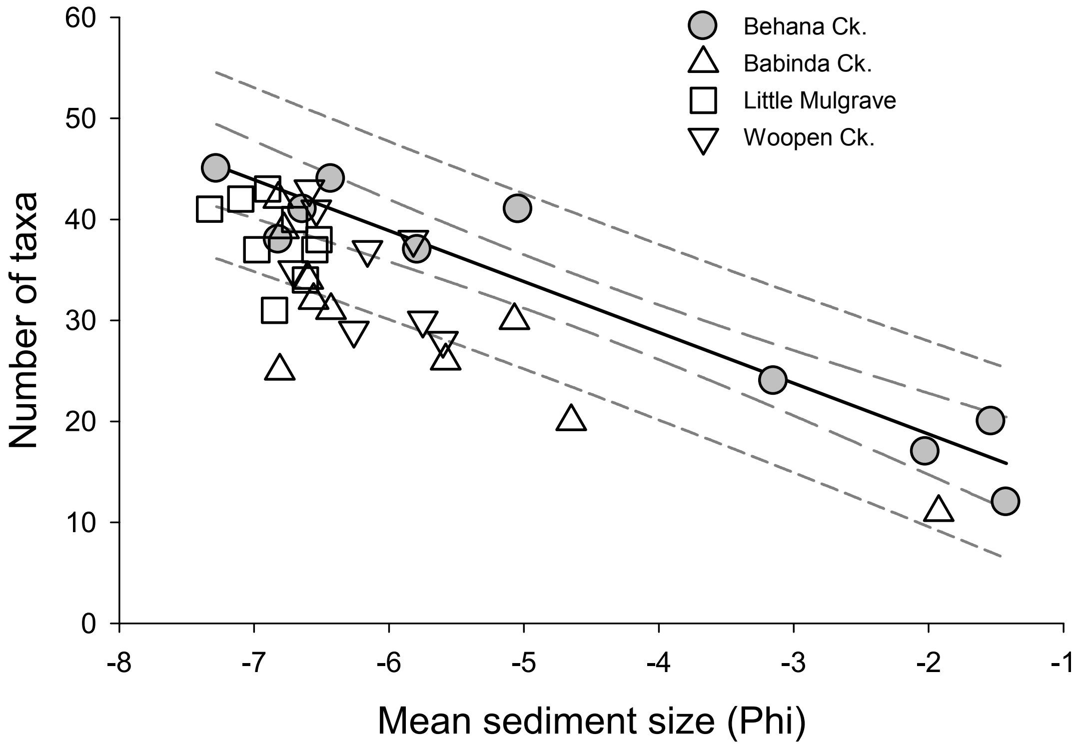

Figure 4.10. A comparison of the taxon richness in the four streams relative to the diversity in Behana Creek. Solid line represents the linear regression for Behana Creek. Long-dashed lines represent the 95% confidence intervals for this regression (i.e. the range where the regression line will fall 95% of the time for repeated measurements of the same population). Short-dashed lines represent the 95% prediction intervals (confidence intervals for the population) for this regression (i.e. the range where the data values (sites) will fall 95% of the time for repeated measurements of the same population).

4.4.7 Differences in riparian vegetation and CPOM

A description of riparian condition in the study area is given in Chapter 3. There was a trend of riparian cover declining with distance downstream in all streams (Figure 11a-d). However, the riparian cover differed significantly among the four streams ($F_{3,25} = 17.743$, $p < 0.001$) when sites within the Wooroonooran National Park boundaries (the two most upstream sites in each stream) were excluded from the analysis (Figure 4.12). Tukey post hoc tests showed that the difference was due to significant differences between the streams in the Mulgrave River catchment compared with the streams in the Russell River catchment. There was also a marginally significant difference in the amount of coarse particulate organic matter (CPOM) collected in macroinvertebrate samples between these streams ($F_{3,25} = 2.043$, $p = 0.054$) (Figure 4.12). The mean levels of CPOM corresponded to the mean riparian vegetation cover (Figure 4.12) and there was a significant correlation between the riparian vegetation cover and the CPOM in the samples, although the correlation was not strong ($r^2 = 0.263$) (Figure 4.13).

There was a tendency for macroinvertebrate richness to increase with riparian cover across the study area (Figure 4.14). However, it was difficult to identify the response of the macroinvertebrate assemblages to the riparian vegetation cover because of the dominance of the longitudinal gradient in determining the macroinvertebrate assemblages. When the substratum sediment size was accounted for, there was a significant difference in the number of macroinvertebrate taxa at sites with or without riparian vegetation: there were more taxa at sites with greater than 50% riparian cover compared with sites with less than 50 % riparian cover, when mean sediment size was used as a covariate ($F_{1,32} = 4.896$, $p = 0.034$) (Figure 4.15). The relationship between taxon richness and CPOM was logarithmic with the inflection occurring at approximately 5 g CPOM (Figure 4.16) and there was a significantly greater number of taxa at sites with greater than 5 g of CPOM than at sites with less than 5 g of CPOM in the sample ($F_{3,25} = 11.665$, $p = 0.002$) when mean sediment size was accounted for (Figure 4.17).

The sample from Site 10 was excluded from these comparisons because it had an unexpectedly low diversity given that of adjacent sites and because it was located upstream from land-use influences.

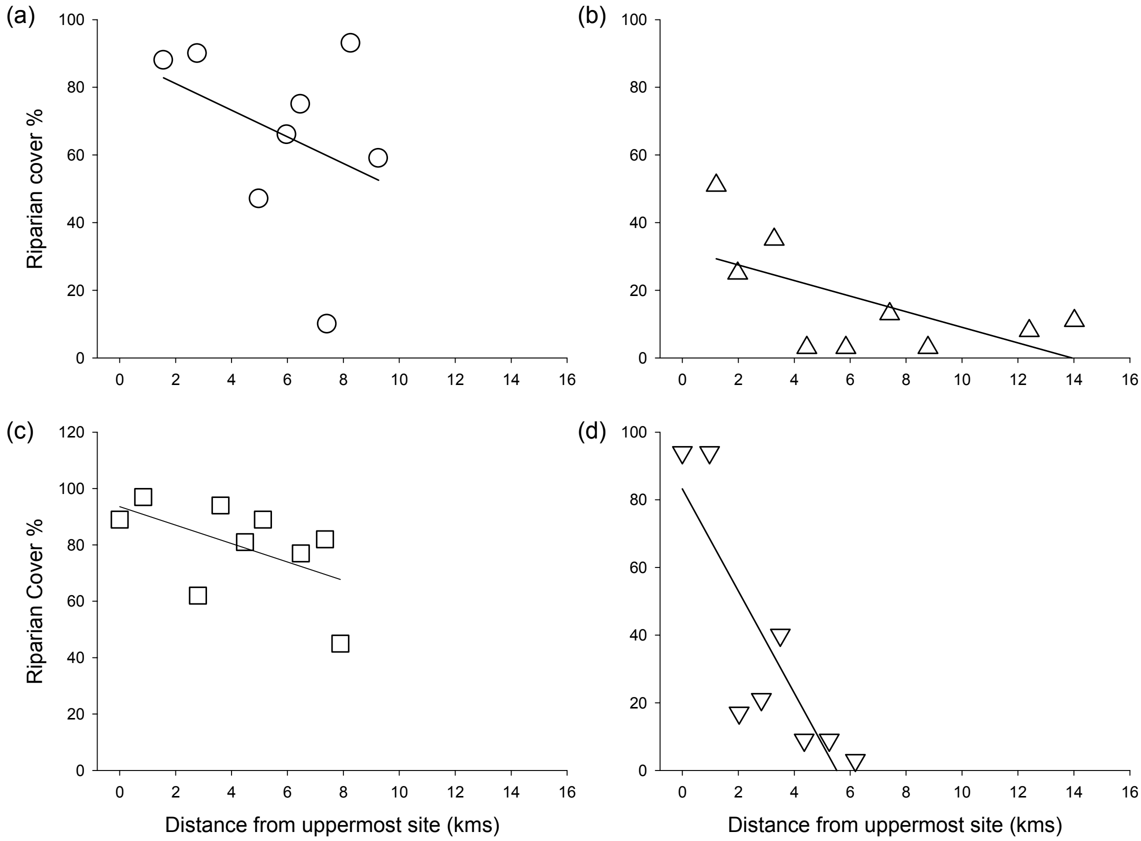

Figure 4.11. The riparian vegetation cover recorded at sample sites in relation to the distance from the uppermost site, positioned at the base of the range, for each of the four streams surveyed: (a) Behana Creek (b) Babinda Creek (c) Little Mulgrave River (d) Woopen Creek.

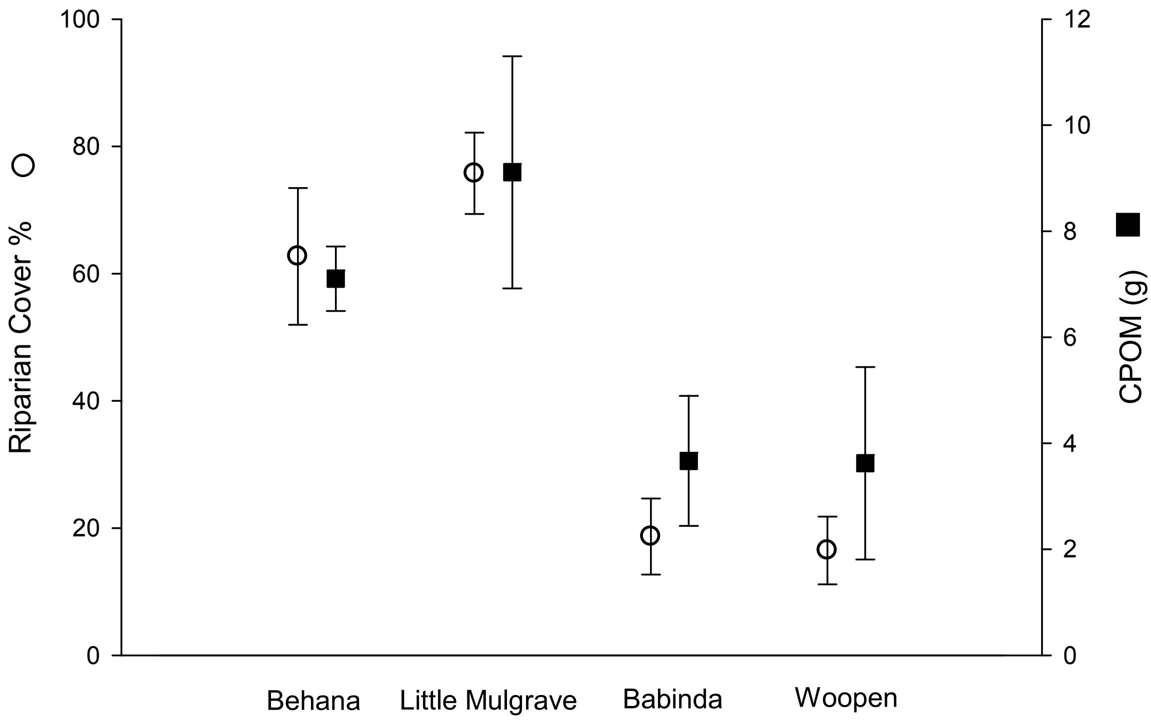

Figure 4.12. Mean % riparian cover and mean dry weight of coarse particulate organic matter (CPOM) collected in macroinvertebrate samples for each of the four streams. Error bars represent the standard errors of the means. The upper two sites are omitted from each stream because they were within Wooroonooran National Park and therefore upstream of agricultural land use.

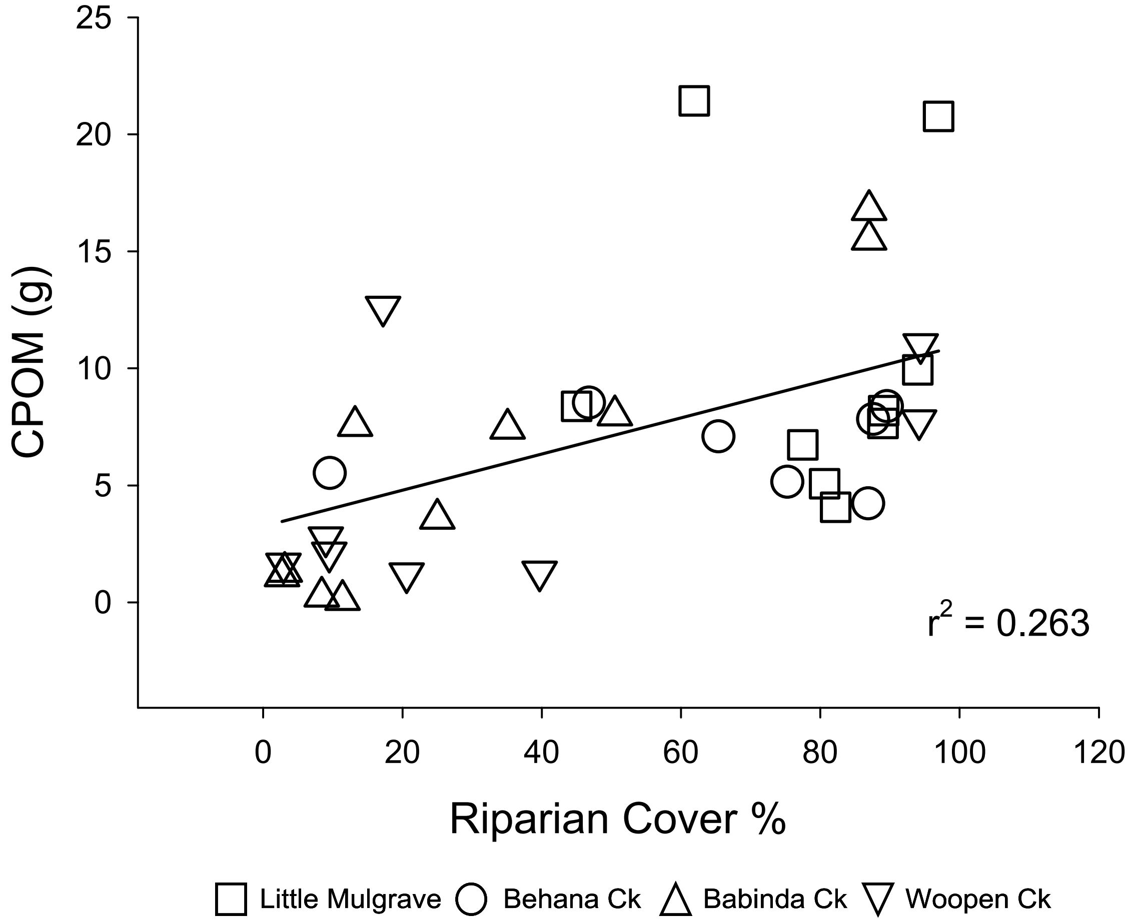

Figure 4.13. The relationship between riparian cover and coarse particulate organic matter (CPOM) collected in macroinvertebrate samples in each of the four streams.

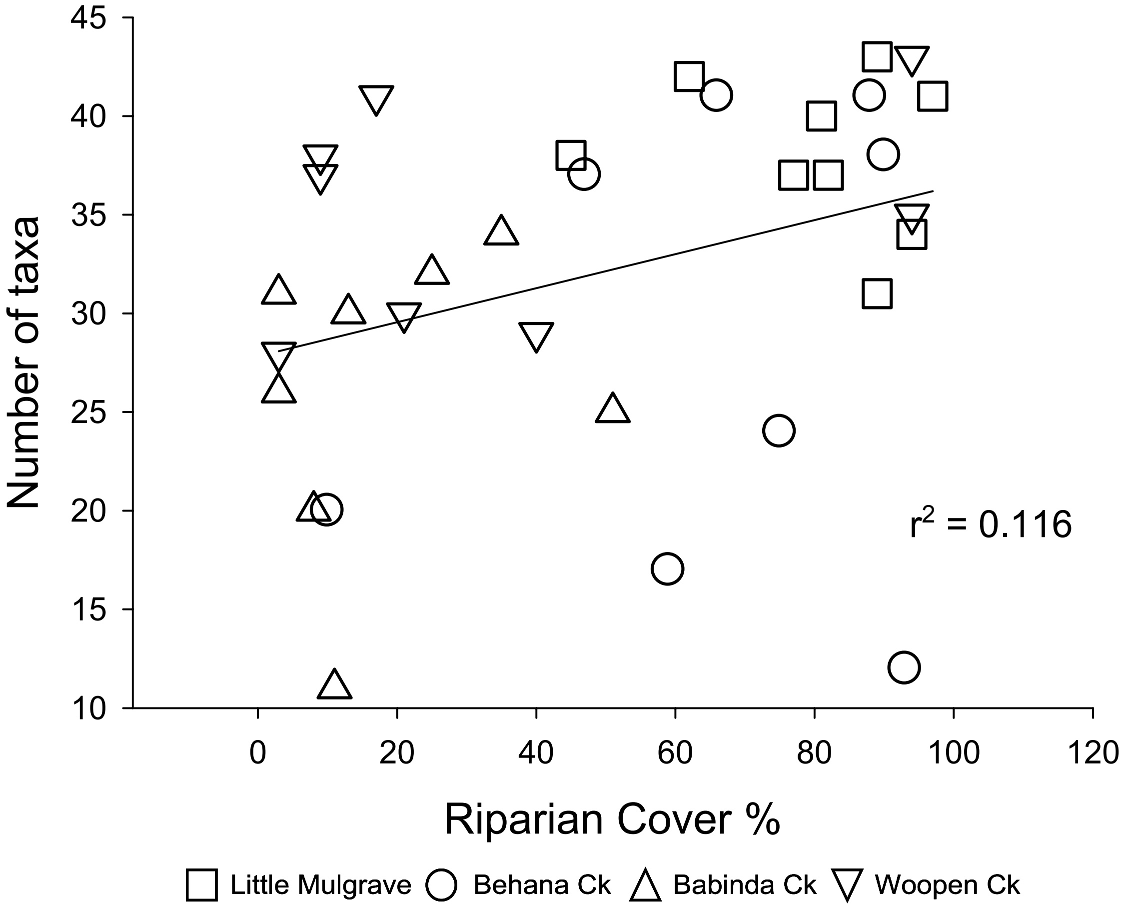

Figure 4.14. The relationship between riparian cover and the number of macroinvertebrate taxa collected in each of the four streams.

4.4.8 Testing Indices

All of the tested indices correlated strongly with taxonomic richness (Figure 4.18a-c), so they all varied systematically along the stream, as did richness. Therefore, the longitudinal gradient in each stream needed to be accounted for when comparing the streams using any of these indices. The ability of each of the invertebrate indices (species richness, family richness, SIGNAL 2 and PET) to detect disturbances to the macroinvertebrate fauna was tested by comparing the significance values (F ratio and p value) obtained for the ANCOVA model. The number of Babinda Creek sites that fell outside the 95% prediction intervals of the regression of number of taxa and mean sediment size for Behana Creek sites also gave an indication of the ability to identify sites in Babinda Creek that were different from those in the reference stream, Behana Creek.

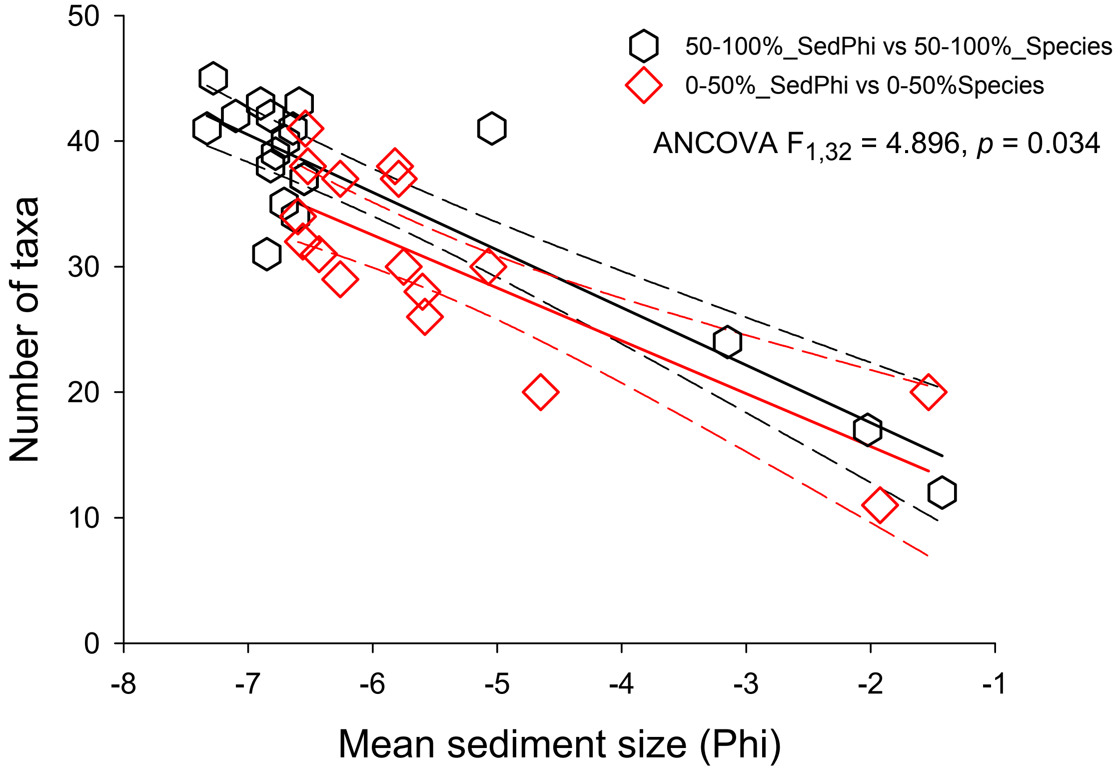

Figure 4.15. The relationship between mean sediment size and the number of macroinvertebrate taxa recorded at sites with greater and less than 50% riparian cover. Solid lines represent the linear regressions for each group. Dashed lines represent 95% confidence intervals for the regression lines. The significant difference between groups is shown by the results of an analysis of covariance (ANCOVA) using mean sediment size as a covariate.

Species richness had the greatest ability to detect differences between the two streams and SIGNAL 2 had the least ability. A significant difference between the sites in Behana and Babinda Creeks could be detected using the number of species, the number of families or the number of PET species (Figure 4.19a,b,d). However, a significant difference could not be detected using the SIGNAL 2 index (Figure 4.19c). The F-ratio value was similar for the analyses using species richness and family richness; however, it was considerably lower for the PET index, suggesting that it may not detect differences resulting from smaller effects.

A greater number of sites from Babinda Creek fell outside the 95% prediction intervals from the Behana Creek regression model when data was summarised into species richness compared with the other three indices (Figure 4.20a-d). When family richness was used, the regression coefficient for the Behana Creek model was lower and no sites from Babinda Creek fell clearly outside the prediction intervals (Figure 4.20b). Both SIGNAL 2 and PET had high regression coefficients for the Behana Creek regression models, but only Sites 10, 12, and 17 fell outside the prediction intervals when SIGNAL 2 was used, and only Sites 10 and 17 fell outside the prediction intervals when PET was used.

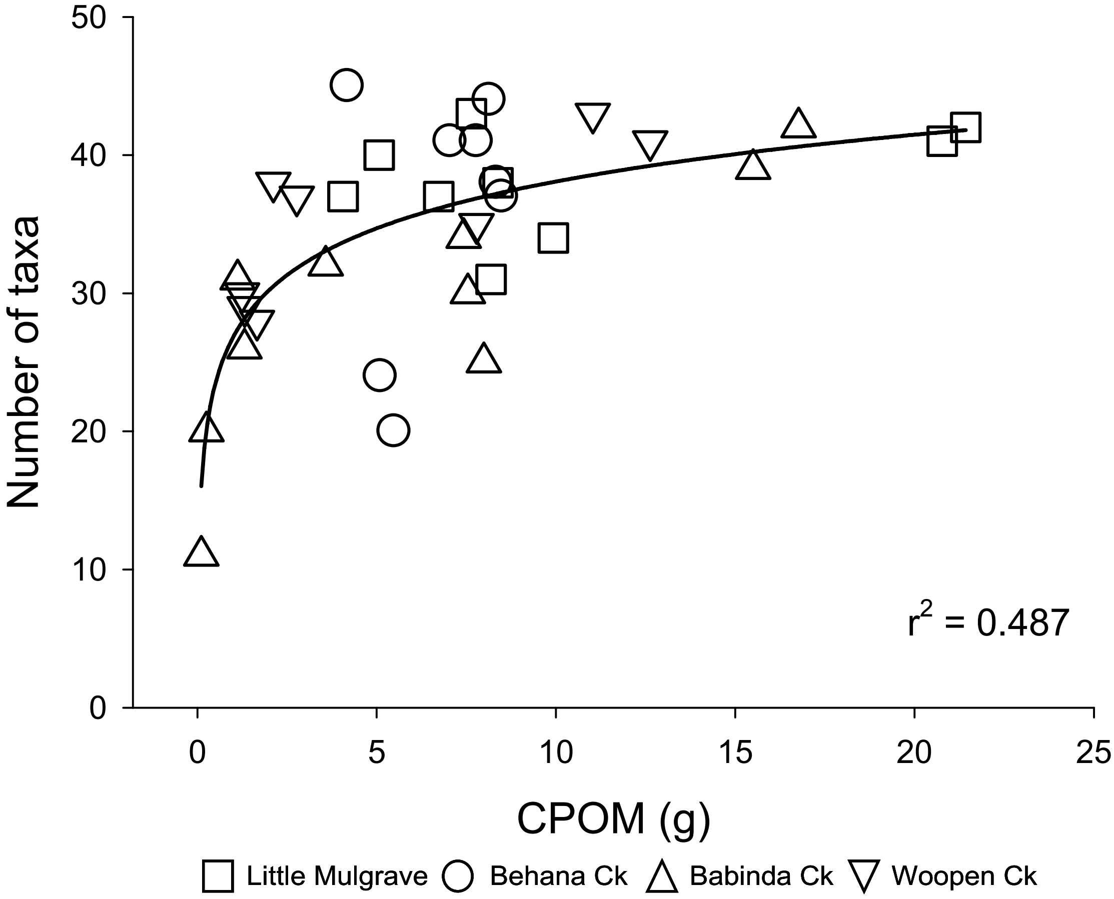

Figure 4.16. The relationship between coarse particulate organic matter (CPOM) collected in macroinvertebrate samples and the number of macroinvertebrate taxa for each of the four streams. Regression is based on the equation $y_1 = y_0 + a.\ln(x)$.

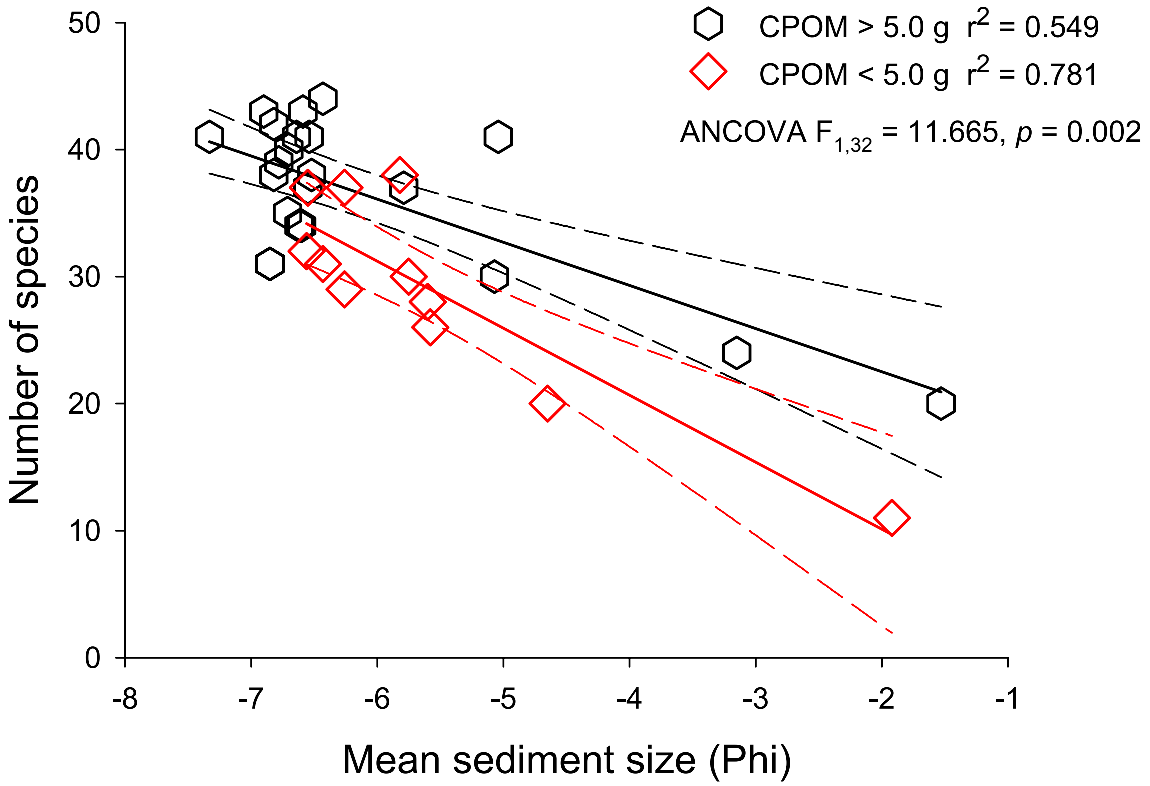

Figure 4.17. The relationship between mean sediment size and the number of macroinvertebrate taxa recorded at sites with greater and less than 5 g of coarse particulate organic matter (CPOM) collected in macroinvertebrate samples. Solid lines represent the linear regressions for each group. Dashed lines represent 95% confidence intervals for the regression lines. The significant difference between groups is shown by the results of an analysis of covariance (ANCOVA) using mean sediment size as a covariate.

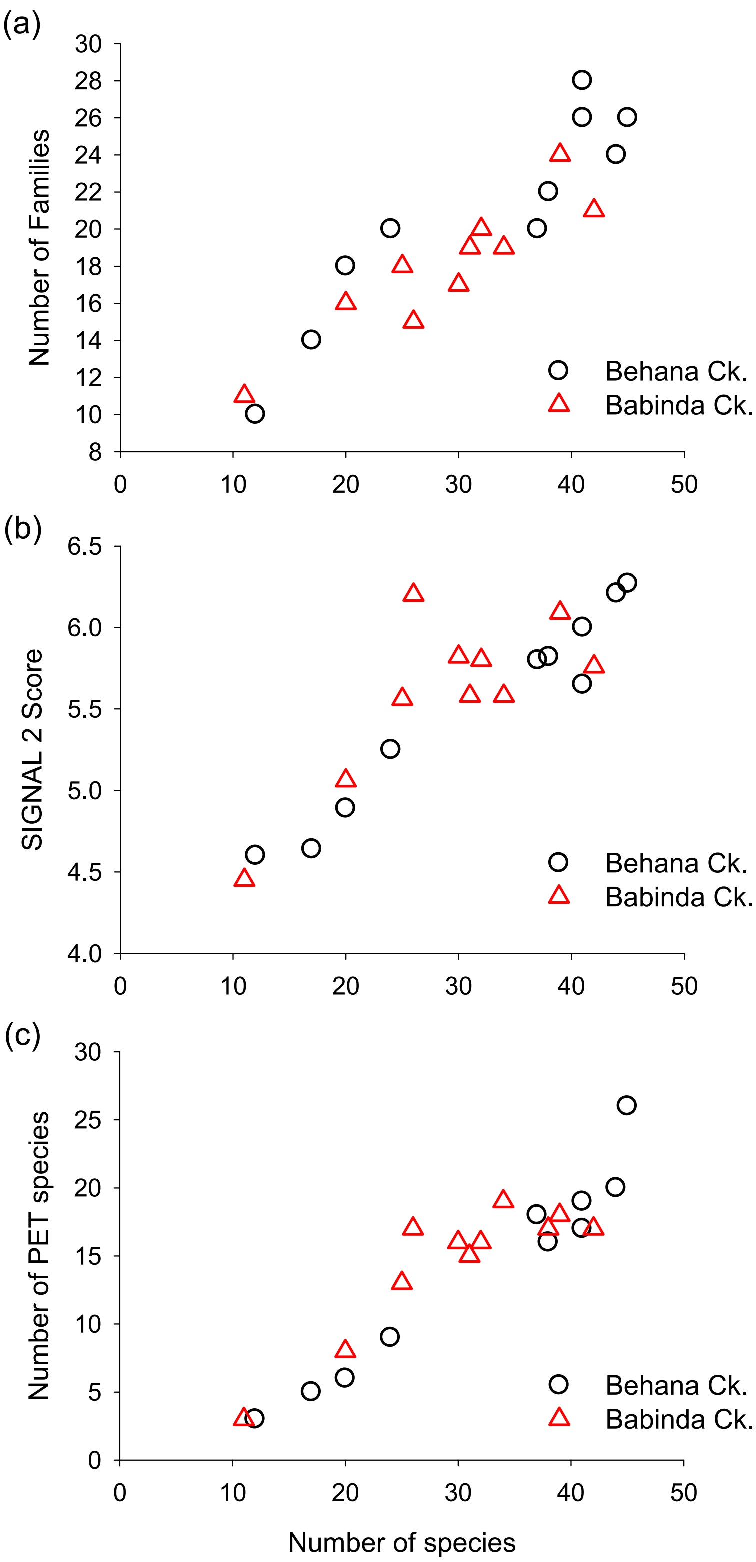

Figure 4.18. The relationship between the number of taxa and the three macroinvertebrate indices: (a) number of families (b) SIGNAL 2 score (c) number of PET species.

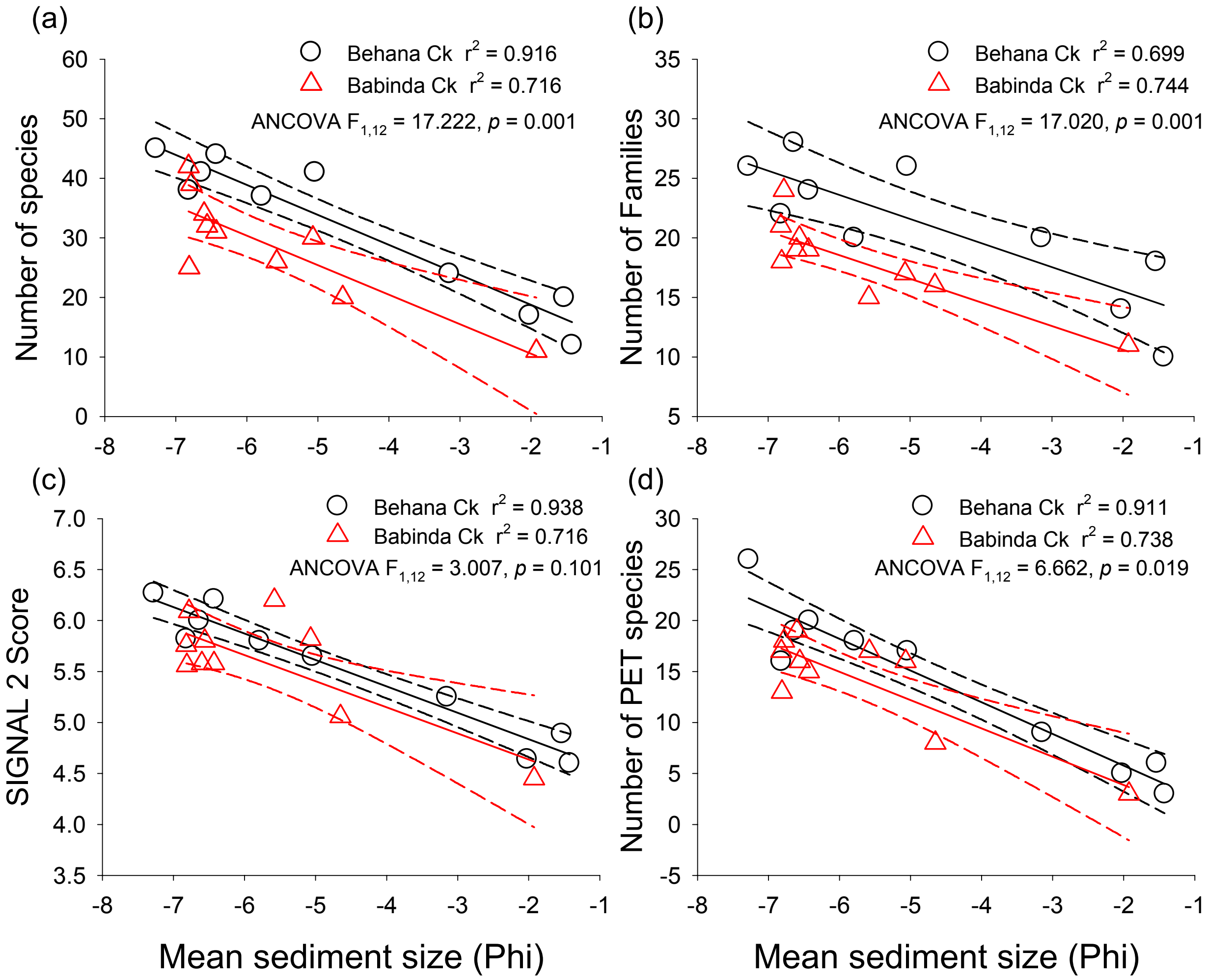

Figure 4.19. The relationship between the mean sediment size and the number of taxa recorded at Behana and Babinda creek sites comparing the ability of different macroinvertebrate indices to detect differences between the streams. (a) Species richness (b) Family richness (c) SIGNAL 2 scores (d) number of PET species. Regressions are based on the linear equation $y_1 = y_0 + a(x)$. Solid lines represent the linear regression. Dashed lines are the 95% C.L. The significant difference between streams is shown by the results of an analysis of covariance (ANCOVA) using mean sediment size as a covariate.

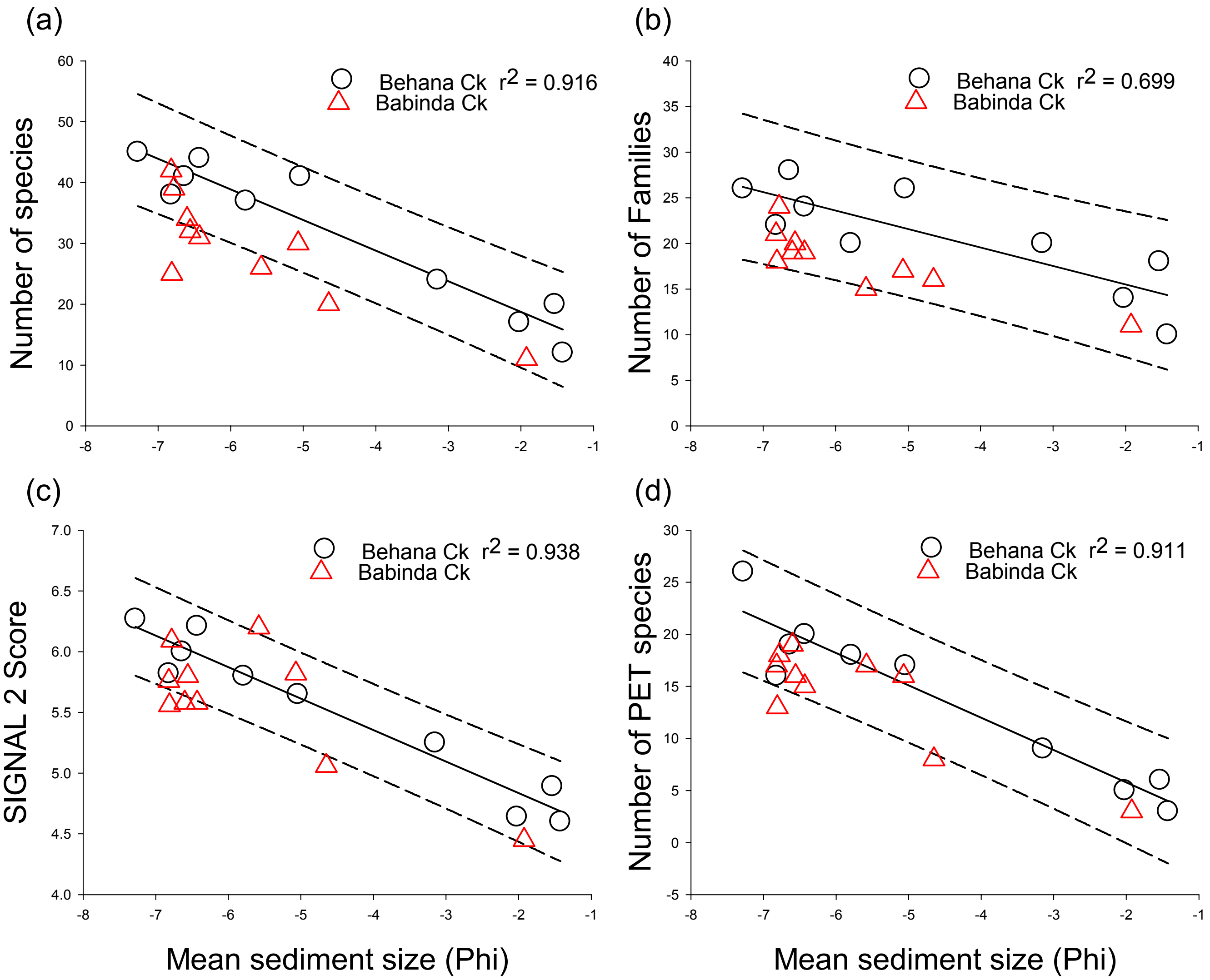

Figure 4.20. A comparison of the taxon richness in Babinda Creek and Behana Creek using each of the macroinvertebrate indices. (a) Species richness (b) Family richness (c) SIGNAL 2 scores (d) number of PET species. Solid line represents the linear regression of mean sediment size and the number of taxa recorded for Behana Creek. Dotted lines represent the 95% prediction intervals (confidence intervals for the population) for this regression (i.e. the range where the data values (sites) will fall 95% of the time for repeated measurements of the same population).

4.4.9 Testing sample size

The ANCOVA model, using mean sediment size as a covariate, comparing sites in Behana and Babinda Creeks, was used to test the effect of sample size (number of individuals) on the ability to detect differences in macroinvertebrate assemblages. The number of individuals in a sample had a large effect on the number of taxa recorded at a site with the greatest effect of sample size occurring at the most diverse sites.

The mean estimate of sample size using the randomised resampling method, underestimated species richness, with fewer taxa being represented in the 1000-individual sample level compared with the observed diversity in actual samples, which generally contained about 1000 individuals, ranging from just under 500 to nearly 2000 individuals. This is an inherent problem with resampling methods and was the result of the underestimation of the very rare taxa in the simulated samples, which occurred across all sites. To alleviate the problem of underestimating rare taxa, the upper 95% confidence interval value for species richness was used to compare the effect of sample size. This gave a much closer estimate of the observed data when a sample level of 1000 individuals was used.

A significant difference between Behana and Babinda creeks was detected for all sample sizes (1000, 500, 250, 100 and 50 individuals) although the F ratio became smaller with smaller sample sizes, indicating that the ability to detect smaller effects would decline with the number of individuals in the sample (Figure 4.21a-e).

The correlation coefficient for the regression of species richness against mean sediment size for Behana Creek reduced with sample size, as did the number of Babinda Creek sites that fell within the 95% prediction intervals (Figure 4.22a-e). This shows that as sample size declines, the ability to distinguish a single Babinda Creek site from the Behana Creek regression model is reduced.

Overall, a sample size of 250 individuals appeared to be a reasonable trade-off between sampling effort and ability to detect differences, with several sites in each stream, using ANCOVA, or identifying a single site as different from the reference model. At a sample size of 100 or 50 individuals, analysis of the macroinvertebrate assemblage will detect a difference between these two streams if several sites are sampled, enabling an ANCOVA. But if only one or few sites are compared, then this level of sampling may be inadequate to detect differences.

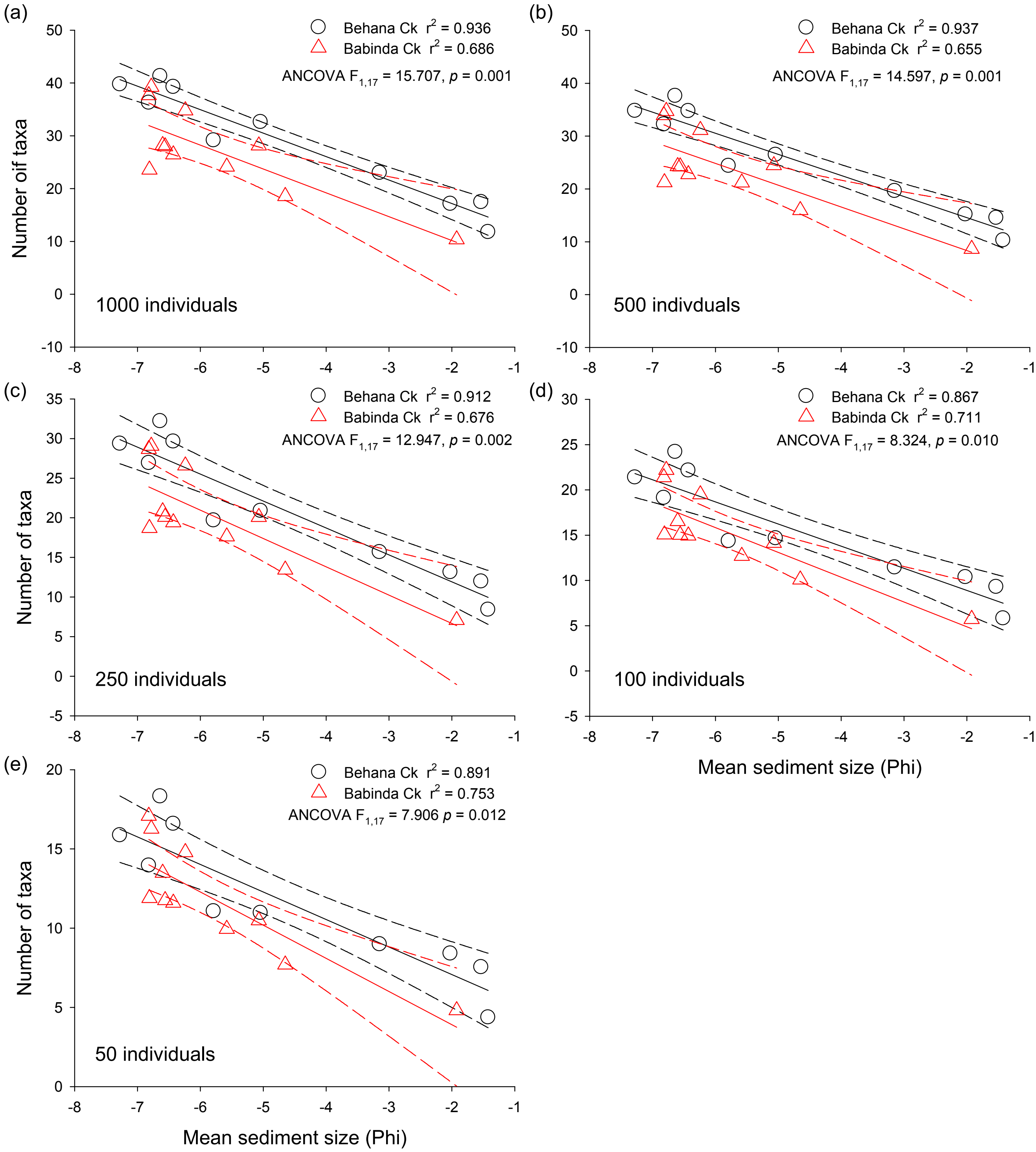

Figure 4.21. The relationship between the mean sediment size and the number of taxa recorded at Behana and Babinda creek sites comparing the ability of different sample sizes to detect differences between the streams, using the upper 95% confidence interval value of the mean estimated number of taxa for each sample size level. (a) 1000 individuals per sample (b) 500 individuals per sample (c) 250 individuals per sample (d) 100 individuals per sample (e) 50 individuals per sample. Regressions are based on the linear equation $y_1 = y_0 + a(x)$. Solid lines represent the linear regression. Dashed lines are the 95% C.L. The significant difference between streams is shown by the results of an analysis of covariance (ANCOVA) using mean sediment size as a covariate.

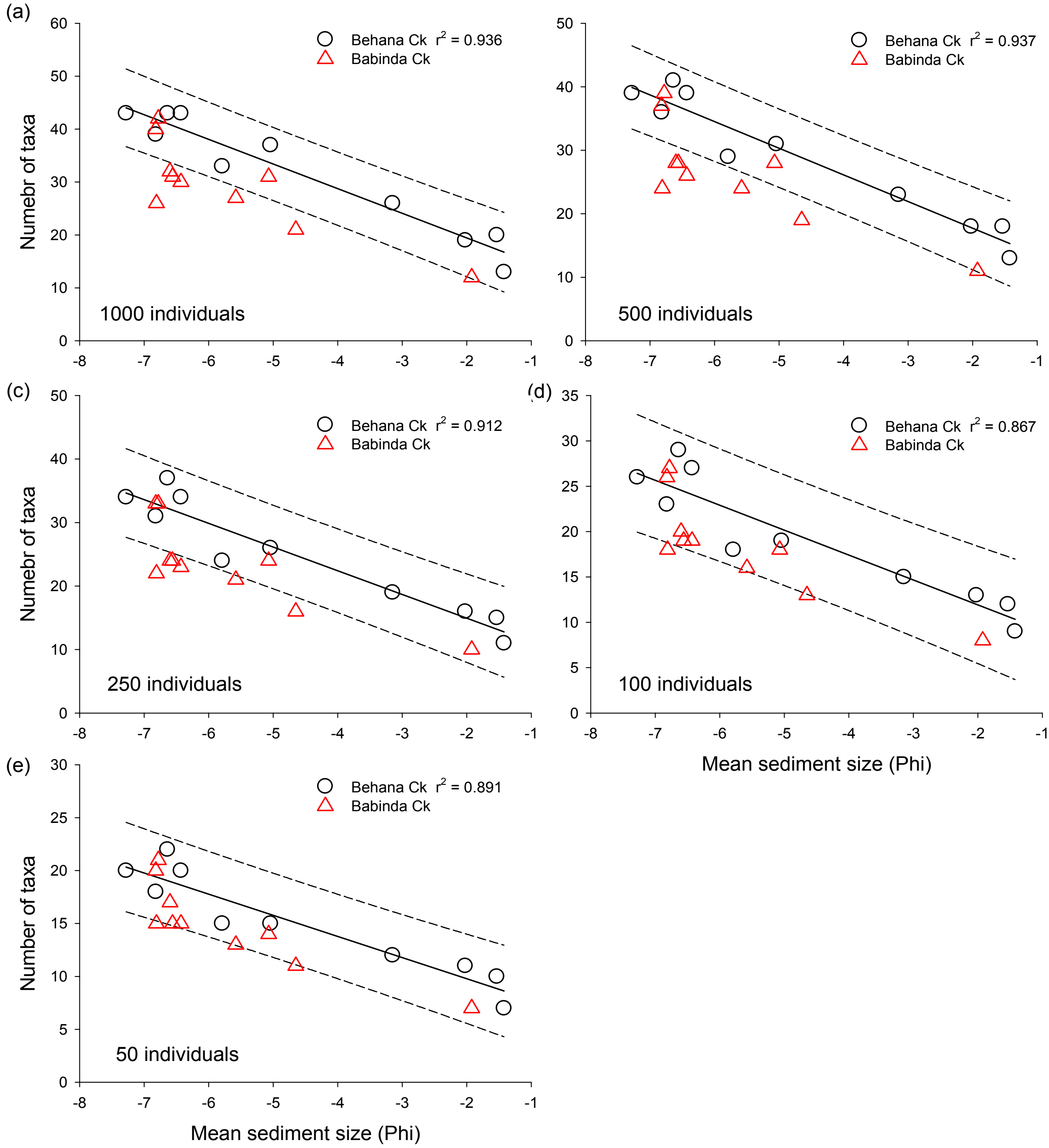

Figure 4.22. A comparison of taxon richness in Babinda Creek and Behana Creek for each of the macroinvertebrate indices, using the upper 95% confidence interval of the mean estimated number of taxa for each sample size. (a) 1000 individuals per sample (b) 500 individuals per sample (c) 250 individuals per sample (d) 100 individuals per sample (e) 50 individuals per sample. Solid line represents the linear regression of mean sediment size and the number of taxa recorded for Behana Creek. Dotted lines represent the 95% prediction intervals (confidence intervals for the population) for this regression (i.e. the range where the data values (sites) will fall 95% of the time for repeated measurements of the same population).

4.5 Discussion

4.5.1 General patterns

The macroinvertebrate diversity collected in these streams represents a significant fraction of the aquatic macroinvertebrate biodiversity currently known to exist in wet tropics streams. For example, the 13 species of Leptophlebiidae collected in this survey represent 45 % of the Leptophlebiidae species currently known to exist in wet tropics streams (F. Christidis pers. comm.; Connolly *et al.* (in press)). Eleven species of Hydropsychidae were collected, with 21 species known from the wet tropics (Dean 1999). The Elmidae were also a very diverse group that potentially provide useful information for biomonitoring in these streams.

The data collected fills a gap in knowledge of the macroinvertebrate fauna in these lowland streams. Previous to this survey there had been no comprehensive description of the distribution of macroinvertebrate species in the lowland streams of the wet tropics. The discovery of new species, even of the well studied Leptophlebiidae (Christidis 2003), highlights that few studies have surveyed the lowland sections of these streams in detail.

4.5.2 Longitudinal gradient

It was very clear that the longitudinal gradient in substratum particle size was a strong determinant of species richness and assemblage structure in all four streams surveyed. Sites were clearly distinguishable along the stream gradient by the number of taxa, but there were also clear changes in composition and presence of species associated with particular sediment sizes. The anomalous Site 16 in Babinda Creek, where large cobbles had been added, provided a useful ‘experiment’ and confirmed the importance of substratum particle size in determining the composition of the macroinvertebrate assemblage.

Longitudinal gradients in streams and the effect of sediment particle size have long been recognised (Allan 1975; Vannote *et al.* 1980; Minshall 1984; Bapista *et al.* 2001). However, this gradient is particularly strong in these high-energy streams of the wet tropics, with rapid changes in substratum over short distances as a product of high rainfall, steep ranges and relatively short streams crossing a narrow floodplain. The strength of the longitudinal gradient, and the strong association between the substratum particle sizes and the macroinvertebrate assemblage composition, provided predictability that facilitated comparisons between streams that will be useful in developing predictive models to use in monitoring programs.

Having a clearly defined gradient that was consistent in each stream enabled comparisons between sites in different streams. The use of the mean sediment particle size as a covariate in analysis of covariance proved to be a very robust way of detecting differences between these streams.

4.5.3 Comparisons between streams

The results clearly show that the macroinvertebrate assemblages in Behana and Babinda Creeks are different, with approximately 20-25% less diversity at Babinda Creek sites downstream from National Park boundaries. This difference was only detectable by sampling a number of sites along each stream and accounting for the substratum particle sizes in the comparisons. If only a few sites had been compared, and site selection had not been carefully stratified by sediment particle sizes, differences would probably not have been apparent and different conclusions could have been made.

The importance of consideration of the substratum gradient was also apparent in comparisons between the Little Mulgrave River and Woopen Creek. Originally it was intended to compare the macroinvertebrate assemblages in these two streams to contrast the effects of different land use and riparian cover, as for Behana and Babinda Creeks. Both the Little Mulgrave River and Woopen Creek occur in similar general locations in their catchments and comparisons between these streams seemed appropriate. However, these two streams had different sediment size ranges, with resulting differences in macroinvertebrate assemblages, precluding direct comparisons between them. If these streams were assumed to be a valid contrasting pair, differences in the macroinvertebrate assemblages could easily be attributed to factors other than substratum particle size. These results highlight how an understanding of the natural gradients is vital to prevent inappropriate comparisons and spurious conclusions.

4.5.4 The effect of riparian vegetation and CPOM

The difference in the macroinvertebrate assemblages between streams appears attributable to differences in riparian vegetation cover affecting the amount of coarse particulate organic matter (CPOM) available in the streams. The critical quantity of CPOM in the macroinvertebrate samples was approximately 5 g. Although the riparian vegetation affected the amount of CPOM at a site, it may be determined at a larger scale: our results suggest that this is the case because the relationship between CPOM and the riparian cover at the site scale was not very strong. However, when the streams were compared at a large scale, using the means of the site values, the relationship was striking.

The amount of CPOM at a site was estimated from material collected in macroinvertebrate samples. A more quantitative measure should reveal a clearer picture of the relationship between CPOM and the riparian vegetation at a site, so quantitative sampling of CPOM should be undertaken during future surveys.

The relationship between the number of taxa and the amount of CPOM was stronger than for the number of taxa and the riparian cover, suggesting that the macroinvertebrate assemblage is influenced indirectly by the riparian vegetation. The shape of the relationship between the number of taxa and the amount of CPOM in the samples suggests that only a certain quantity of the organic material is required to support the full macroinvertebrate assemblage and, at the time of the sampling survey at least, was available in excess quantities at some sites, although these large quantities may be required to support large resilient populations.

These results are tentative because the categories used to test differences in riparian cover (< or > 50%) and the amount of CPOM (< or > 5 g CPOM) meant that the sites were separated into streams in different catchments, which were already shown to be different. Although we sampled two streams in each catchment, we do not have replication at the catchment scale, so differences could be due to differences between catchments. Nevertheless the results do provide strong evidence for the importance of riparian vegetation and CPOM input to the maintenance of the macroinvertebrate assemblage.

4.5.5 Water Quality

None of the water quality parameters correlated with macroinvertebrate distributions, despite there being very high concentrations of agricultural nutrients in all streams surveyed. For example, the concentrations of dissolved inorganic nitrogen generally increased with distance from the range proportional to the area of catchment used for sugar cane production (Chapter 2). Phosphorus concentrations were low in Behana Creek, but in Babinda Creek there was a large spike in phosphorus concentrations in the upper reaches, in an area adjacent to banana farms (Chapter 2). However, it is not clear what effect this enrichment has had on the macroinvertebrate assemblages.

Dissolved inorganic nitrogen concentrations increased exponentially with distance along the streams, suggesting that there is little adsorption or uptake of this nitrogen within the stream. Conversely, the decrease in phosphorus concentrations downstream from a peak in the upstream reaches of Babinda Creek suggests that adsorption or uptake occurs; however, there is no further source of input downstream from these sites so dilution is also likely. Because other water quality parameters were generally unremarkable, with high flows and reasonably cool water, and there was no obvious increase in algae or microphyte growth, enrichment is likely to be having only minor impacts. For example, it may result in increased densities of macroinvertebrates without necessarily causing a loss of diversity as might otherwise be associated with eutrophication, as we have demonstrated experimentally (Pearson and Connolly 2000).

The abundances of macroinvertebrates were high at sites a few kilometres downstream from the foothills in all streams surveyed. Samples from these sites contained nearly twice the number of individuals compared with others, because of very high numbers of a few taxa. However, in Behana and Babinda Creeks abundances then declined further downstream as sediment particle sizes reduced to gravels and sands, creating a roughly unimodal pattern of abundance. An increase in density of macroinvertebrates would be expected to occur naturally as the streams widen and more light is available to support the growth of primary producers on the substratum surfaces. But it appears that there is an interaction between having sufficient light to allow primary production and having sediment particle sizes that are large enough to provide a stable substratum for the primary producers to attach and grow. Thus, abundances declined as the substratum particles reduced to gravels and sands.

The very high abundances of some taxa may be indicative of nutrient enrichment, but because nutrient levels in all streams were enriched it is impossible to compare between streams or to know what natural abundance levels would be. The abundances of some taxa did appear to be higher in Babinda Creek compared with Behana Creek and this may be because of phosphorus enrichment in Babinda Creek. In experiments we have shown that phosphorus enrichment can greatly increase the density of macroinvertebrates in wet tropics streams, whereas nitrate has little effect (Connolly and Pearson, unpublished data).

4.5.6 Testing indices

The analysis of covariance model, comparing sites in Behana and Babinda creeks, gave us a useful benchmark to test the ability of the macroinvertebrate indices to detect differences in macroinvertebrate assemblages. Indices are rarely tested against a benchmark where sites are of known condition. Instead, indices are compared with each other or correlated with assumed physico-chemical disturbance variables (e.g. Marshall *et al.* 2001), but their actual ability to discriminate between sites of known condition seems rarely tested because there is seldom an independent measure of the condition at the sites being compared.

Species richness had the greatest ability to detect differences between the two streams and SIGNAL 2 had the least ability. While family richness and PET could detect a significant difference between Behana and Babinda Creeks when several sites in each stream had been sampled (using the ANCOVA model), the ability of the reference models using family richness or PET to identify individual impacted sites in Babinda Creek was lower than when using species richness. Essentially this result shows that there is a trade-off between the number of sites sampled and the detail collected at each site. That is, the number of families or PET can be used to detect differences if several sites are sampled in each stream, allowing the ANCOVA model to be used. However, if only a few sites are sampled such that the ANCOVA model cannot be used, or if a single stream such as Behana Creek is to be used as a reference model, then species richness is the measure that will have the best likelihood of detecting differences between the streams.

Therefore, if taxonomic expertise is not available, it may be beneficial to sample more sites but limit identification to specific groups or to the family level. However, expertise is required to design the appropriate sampling schemes and to decide on the number and locations of sites. This will need some form of prior knowledge or pilot study in the region to assess the variability and the need to stratify sample locations or account for gradients in macroinvertebrate distributions.

4.5.7 SIGNAL

The inability of SIGNAL 2 to detect differences between the macroinvertebrate assemblages in Behana and Babinda Creeks is significant given that this index is currently very popular and was one of the indices recommended for use in the Ecosystem Health Monitoring Program in south-east Queensland (Marshall *et al.* 2001).

SIGNAL was unable to discriminate a difference between these streams largely because Sites 14 and 15 in Babinda Creek had relatively high scores owing to the presence of some high-scoring taxa, even though richness was otherwise lower than in Behana Creek. This indicates that the score values allocated to these taxa may be inappropriate. SIGNAL assumes that the macroinvertebrate community will shift from an assemblage consisting of sensitive taxa to one consisting of only tolerant taxa. We found it difficult to identify taxon-specific responses. Although Babinda Creek sites had lower richness, the species that were present were not limited to those classified with low SIGNAL scores. Instead our results showed a general loss of species across all groups. Further, because the sites are ranked by averaging the SIGNAL scores for each taxon, if the assemblage had not shifted from a sensitive to a tolerant assemblage, but just had lower numbers of taxa, the averages of the taxon scores may be expected to be very similar.

The average value also poorly represents the highly non-normal distribution of the macroinvertebrate assemblage being sampled (and the highly skewed distribution of allocated scores themselves). Mean values do not describe skewed or multimodal frequency distributions adequately and will vary greatly with repeat sampling, especially if sample size is small, and no measure of variance around this mean value is considered in SIGNAL. When confidence limits were placed on the SIGNAL mean value for each site, based on the distribution of scores in the sample, sites were invariably not statistically different, unless comparing sites from very different habitats.
SIGNAL values tend to cluster around certain levels depending on the microhabitat being sampled, with a fast-flowing riffle habitat scoring higher than slower-flowing habitats, but struggle to discriminate samples taken from the same habitat types (Connolly 2003). Thus, site scores vary over very small ranges for each habitat type, with each habitat type scoring in a different band. This is not in itself a problem except that in reality the scale of resolution is much smaller than the nominal 1 to 10 scale suggests. This also suggests that taxon scores reflect supposed impacts beyond water quality *per se*, but also incorporate expectations from impacts such as changes in flow dynamics and changes in microhabitat, although it is not clearly stated that this is intentional.

SIGNAL also tended to correlate with taxon richness, with more diverse samples scoring higher (Connolly 2003), and was sensitive to the longitudinal gradient in the streams in this study, with downstream sites scoring lowest. Because measures of taxon richness are strongly affected by sampling effort, especially the number of samples or individuals being sampled, it is thus important that sampling effort be similar across sites and adequate to be representative of the assemblage being sampled.

Thus, the response by SIGNAL could easily be masked by these effects and there is the need to properly classify study sites so that valid comparisons are being made. Special attention is required when comparing samples across gradients in habitat or diversity, such as with altitude or along a river.

It should be remembered, though, that SIGNAL was designed as a water quality indicator and water quality did not have a large effect in the streams surveyed in this project. It may be unreasonable to expect SIGNAL to detect impacts that operate through processes other than toxic effects of pollutants. However, it was worth testing SIGNAL in this situation because SIGNAL is often used in situations where the main impacts are not water quality *per se*. For example, SIGNAL is one of several indices used in condition assessments across Queensland for water resource planning, but despite its lack of power to detect effects in the way it is applied (Grinter *et al.* 2000, Connolly 2003, 2006), it is still being used for this purpose.

4.5.8 Testing sample size

The strategy of basing sampling effort on the number of individuals was good and samples in this survey generally were large enough to represent the diversity at sites and to detect effects.

The importance of the number of individuals in determining the species richness and other characteristics of biological assemblages is well known (Gotelli and Colwell 2001) but often poorly accounted for in macroinvertebrate monitoring surveys. The stochastic nature and highly skewed frequency distributions of macroinvertebrate assemblages, with large numbers of rare taxa, means that the level of sampling effort, in terms of the number of individuals collected, is very important to get accurate, repeatable estimates of species richness and other assemblage attributes. Also, as we have demonstrated, many macroinvertebrate indices used in monitoring programs are sensitive to species richness, though this is rarely explicitly stated.

Again it was very useful to be able to use the analysis of covariance model comparing Behana and Babinda Creeks to test the effects of sample size, in terms of the numbers of individuals. The resampling method used, essentially using rarefaction values for each sample size level, seemed to work well, except that it did underestimate the number of rare taxa. This was an unavoidable problem because the rare taxa that occur in the original sample will have a very low probability of occurring when resampled, so some will be sporadically absent in many simulated samples, lowering the mean estimate. In the actual assemblage other rare taxa would exist that did not happen to be represented in the original sample. Therefore, if repeatedly sampling from the original assemblage, there is the opportunity for other rare taxa that did not occur in the first observed sample, to be included. This means that species richness estimates derived from resampling a single sample will generally be lower than estimates derived if resampling the original assemblage. For the purpose of our comparisons, this problem was ameliorated by using the upper 95% confidence value instead of the mean estimate in the comparisons to boost the inclusion of the rare taxa.

As with the level of taxonomic resolution, the ability to detect differences between the macroinvertebrate assemblages at sites in Behana and Babinda Creeks appears to be a trade-off between the number of sites sampled and the number of individuals collected at a site. In this study, we found that the analysis of covariance was very resilient to sample size with significant differences being detectable at all sample size levels tested, even down to 50 individuals. However, F-ratio values did reduce with sample size, indicating that the ability to detect smaller effects would be impeded with smaller sample sizes, and the ability to detect differences between fewer sites was likely to reduce with sample size, as shown by the number of sites that fell outside the prediction intervals. Therefore, if only a few sites are sampled, or if a particular stream is to be used as a reference model (e.g. Behana Creek), then a higher number of individuals need to be collected at each site.

Some methods recommend 100 individuals or greater (Metzeling and Miller 2001; Chessman 2003 b). Our data would suggest 250 individuals are necessary to discriminate differences in the streams surveyed. However, we need to be cautious in recommending this as a sample size for surveys. This level was derived from a random selection of a large sample that had sampled over the entire extent of the riffle, and represented the variety of microhabitats and patches that existed within that riffle. This is essentially equivalent to taking a large sample throughout the riffle and then mixing the individuals and taking a random subsample, somewhat similar to some fixed count subsampling methods in use (Sorvell and Vondracker 1999). However, this is not analogous to taking a smaller sample in the field because individuals are patchily distributed in a riffle. If a small sample were to be taken in the field, fewer patches and microhabitats would have been represented. This would under-represent the diversity in the riffle and result in high variance between repeated samples.

It is clear that species richness is an effective community measure. However, while it is well accepted that sample size affects the estimate of species richness, quantifying the richness of a diverse assemblage with many rare, patchily distributed taxa can be difficult. The number of species accumulates at an ever decreasing rate as sample size increases (number of samples or number of individuals) (Gotelli and Colwell 2001). If sample sizes are large enough such that all species are collected, then comparing species richness between samples is simple. However, if the sample size is such that species are still accumulating at a high rate, then difficulties can arise. If the shape of the species accumulation curves are very similar for the samples being compared, then by standardising the number of individuals, either by using rarefaction or random fixed count subsampling, the samples should be reasonably comparable given the usual sampling error. However, if species accumulate at different rates because of differences in heterogeneity or patchiness at different sites (which could be one of the impacts under investigation), then it is more difficult. It is feasible that changes in heterogeneity and patchiness could occur with the loss of CPOM input and/or nutrient enrichment, as in our study. In this study we deliberately collected large samples so that there was the greatest likelihood that most of the species pool was represented.

Connolly *et al.* (in press) discuss how greater sampling effort is required in tropical streams compared with temperate streams because the macroinvertebrate assemblages have a more heterogeneous distribution in the tropics, and include many more rare taxa, such that species accumulate in samples at a slower rate. This suggests that the recommended sample size of 100 individuals, derived in temperate systems, would need to be modified in the streams of the wet tropics. Our results support this suggestion.

We recommend taking samples large enough to collect 250 individuals as a minimum but, because species were still accumulating at this level, samples of 500 individuals, from the whole riffle, are preferable. Large samples can be subjected to random fixed-count subsampling of 250 individuals, if necessary, as a means to minimise effort in identification, as long as the whole riffle is sampled to overcome the inherent patchiness of the macroinvertebrate distribution.

4.6 Conclusion

There were clear patterns of macroinvertebrate distributions in the streams surveyed, because of the strong gradient in substratum particle sizes along each stream and differences in particle sizes between streams. The macroinvertebrate assemblages were useful in classifying the streams into upper, middle and lower reaches and demonstrated a consistent longitudinal gradient of assemblage structure. The consistency of these patterns enabled comparisons between streams using analysis of covariance and this proved to be a robust approach in detecting differences between streams.

The approach of taking many samples across gradients was very successful, and was demonstrated to be of high utility in developing monitoring protocols. However, our testing of indices and sample size demonstrated that to detect differences there was a trade-off in the amount of detail and effort applied at the site scale and the number of sites used in comparisons. Understanding this trade-off is valuable in that effort can be concentrated to suit individual constraints. For example if high taxonomic expertise is available fewer sites can probably be sampled. Similarly, if fewer sites are available for sampling, because of access or other limitations such as time, then large numbers of individuals at each site should be sampled and the best taxonomic resolution should be used. On the other hand, if a high level of taxonomic expertise is not available then this may have to be compensated for by sampling more sites to include in the contrast, and identifying to family level. Nevertheless, site selection will be critical to avoid confounding effects and will depend on prior knowledge of the macroinvertebrate distributions or require a pilot study.

Our results describing the impacts of loss of riparian vegetation and CPOM are interesting and provide further evidence of the importance of riparian vegetation. However, our conclusions are tentative because we have not yet replicated catchments. The perceived differences could be the result of inherent differences between the two catchments rather than to impacts. Therefore, there is a need for similar surveys to be carried out in different catchments to generalise our conclusions. Moreover, further surveys should encompass other types and degrees of impact to test our approach across different levels of disturbance (rather than just contrasting good with bad).
Nonetheless, our results suggesting that riparian vegetation is a key determinant in maintaining instream diversity is encouraging, as this is the most common remediation currently being applied in these streams and our results confirm that maintaining and rehabilitating riparian vegetation is a worthwhile activity.

We demonstrated the efficacy of different monitoring indices, with species richness being the clear winner, although identification at a higher taxonomic level (typically family) was still more effective than using the commonly adopted SIGNAL index. This highlights the need to test indices under the situations in which they are to be applied and to ensure that appropriate measures are being used to answer the question being asked. Choosing an inappropriate or insensitive measure will waste time and use resources that may not be easily obtainable to repeat the exercise.

The holy grail of monitoring – a cheap, easy, one size fits all, silver bullet index – is probably not attainable, and methods that promote this ideal are typically not very useful as diagnostic tools. We aim to develop the methods outlined here into a scientifically robust and efficient tool for monitoring stream health and develop a framework to create sound foundations of information that can be built upon.

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Chapter 5

Freshwater Fish as Indicators of Ecosystem Health in Wet Tropics Streams

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5.1 Introduction

5.1.1 Freshwater fish of the wet tropics region

The Australian freshwater fish fauna is generally considered to be of low diversity when compared with the rich faunas of South America or south-east Asia (Allen *et al.* 2002), although the number of species per river basin is not greatly different from that seen elsewhere in the world after scaling for differences in catchment area or mean annual flow (Bishop and Forbes 1991; Oberdorff *et al.* 1995; Pusey *et al.* 2004a,b). The fauna is acknowledged as being distinctive by virtue of the absence of many primary freshwater fish families and the presence of many endemic species (70% of continental total) and genera (42% of continental total) (Allen *et al.* 2002). Even in this context of biogeographic distinctiveness, the fish fauna of the wet tropics region stands out as being especially distinctive (Pusey and Kennard 1996; Unmack 2001; Pusey *et al.* 2004b; Pusey *et al.* in press).

To date, 103 native and four alien species have been recorded from fresh waters (<2000 $\mu S.cm^{-1}$) of the wet tropics region (Russell *et al.* 1996, 2000; Pusey *et al.* 2004b). These 107 species are contained within 37 families with almost half (52/107) within six families: Eleotridae (gudgeons – 15 spp.), Gobiidae (gobies – 13 spp.), Chandidae (glassfish – 9 spp.), Melanotaeniidae (rainbowfish – 6 spp.), Terapontidae (grunters – 5 spp.) and Plotosidae (eel-tailed catfish – 4 species). These families typically contribute the majority of freshwater fish biodiversity in northern Australian rivers (Bishop and Forbes 1991; Allen *et al.* 2002). Of the 107 species, 49% are entirely restricted to fresh water or have a limited estuarine phase in the life history, 34% have an obligate estuarine or marine life history interval, with the remainder being composed of species within marine families that occasionally may be found in freshwater rivers (e.g. Sillaginidae, Leiognathidae, Platycephalidae and Mugilidae). Self-sustaining populations of four alien species – *Oreochromis mossambicus* (Cichlidae), *Tilapia mariae* (Cichlidae), *Poecilia reticulata* (Poeciliidae) and *Xiphophorus maculatus* (Poeciliidae) – have been recorded from rivers of the region. As many as 36 alien and native species have been stocked in streams, farm dams and impoundments of the region (Burrows 2004) and such translocations frequently have adverse effects on native species, assemblages and ecosystem function (Arthington 1991; Arthington and McKenzie 1997; Canonico *et al.* 2005; Kennard *et al.* 2005; Pusey *et al*. 2006).

Very clear spatial organisation of freshwater fish assemblages is evident in rivers of the WT region with landscape-scale features being important in determining the distribution and abundance of fish (Pusey *et al.* 1995a; Pusey and Kennard 1996; Pusey *et al.* 2005). Large barriers to movement are important in maintaining biodiversity through the isolation of endemic species and distinctive phylogenetic lineages (Pusey and Kennard 1996; Hurwood and Hughes 1998). Other landscape-scale features such as stream elevation, distance from the river mouth and stream channel width are also important in determining the distribution of species within the stream system network. Notably, the structure of fish assemblages based on presence/absence of particular species is determined by landscape-scale features, whereas species abundances are influenced by additional local habitat-scale features such as stream depth, velocity and substrata composition (Pusey *et al.* 2000; Arthington *et al.* 2004).

5.1.2 Impacts of human land use on stream fish

Human land use (agriculture, urban, industrial, water resource development, recreation, waste treatment and disposal) and indirect effects associated with human activities (such as those wrought by translocated or alien pest species) all have the potential to make an impact on stream ecosystems and on stream fish communities (Arthington *et al.* 1983; Bunn and Arthington 2002; Allen 2004; Kennard *et al.* 2005; Dudgeon *et al.* 2006). Moreover, the impact of different land use may be spatially segregated from areas of impact (e.g. because of the downstream transport of material and the barriers to upstream fish movement caused by lowland dams and weirs) and may be extremely long lasting in effect (Allen *et al.* 1997). Typically, and ignoring the influence of flow regime manipulation, human activities affect stream systems only when the by-products of those activities move from the terrestrial to the aquatic biome, as represented in our conceptual model of healthy and degraded stream ecosystem (Fig. 5.1). For example, increased nutrient applications to farmlands to enhance crop production influence stream primary production only when those nutrients find their way into the stream system, either in groundwater or in overland flow. Similarly, the movement of sediment is only of concern to aquatic systems when that movement is from the terrestrial to the aquatic biome (see Arthington *et al.* 1997). In most naturally functioning aquatic systems (ignoring arctic or highly xeric environments), the interaction between these two biomes and the extent to which land-based activities can have an impact on stream and riverine fish is regulated by the riparian zone (Pusey and Arthington 2003). Bank-side trees and shrubs intercept, store and sequester nutrients and inorganic particles thus preventing them from reaching the stream environment (Fig. 5.1).

The riparian zone directly shades the stream environment thus regulating the transfer of solar energy to the wetted portion of the stream as well as the unwetted stream banks. In the absence of this shade, often owing to riparian clearing, the growth of aquatic and marginal plants is enhanced. This is of particular concern in the wet tropics region, where introduced ponded pasture grasses such as para grass (*Urochloa mutica*) and other alien weeds are encouraged by the altered light environment and favourable temperature and water regimes (Bunn *et al.* 1997; Pusey and Arthington 2003). The riparian zone also helps to stabilise bank-associated structures such as undercuts whilst simultaneously providing complexity to the aquatic habitat (i.e., root masses, woody debris and leaf litter). In addition, the fruits of riparian trees and the insects that feed in and on riparian foliage are important to aquatic food webs, particularly in the wet tropics region (Pusey *et al.* 1995b). Clearly, the riparian zone is very important in maintaining the health of stream ecosystems (Bunn 1993; Fig. 5.1).

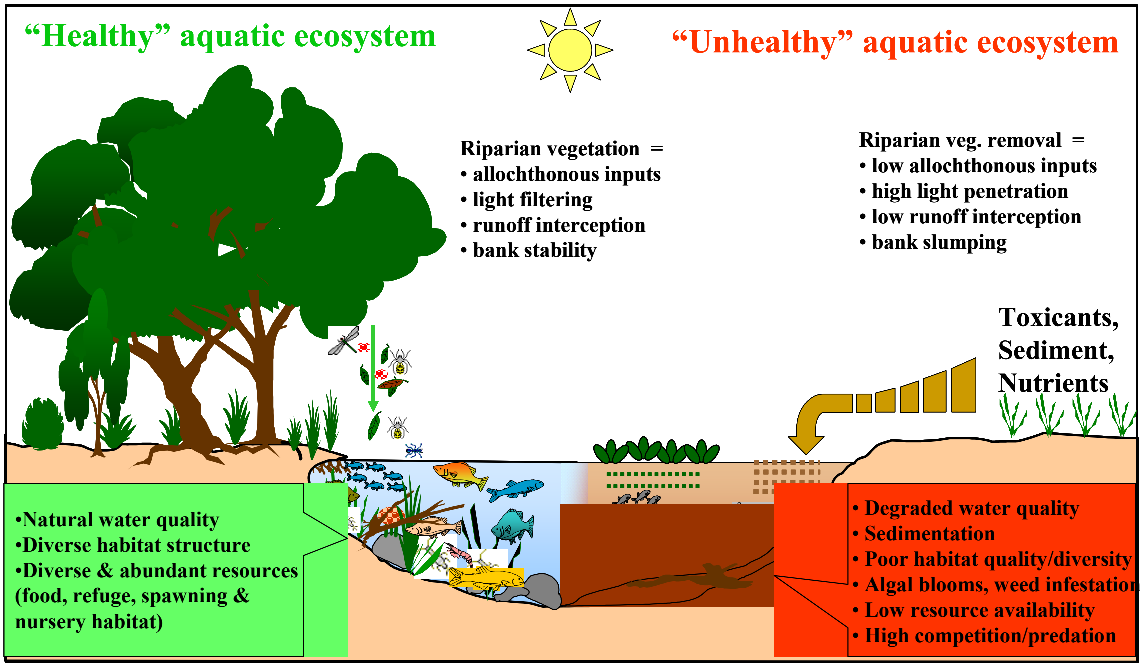

Figure 5.1 Conceptual model of predicted changes in physical and biological characteristics of aquatic environment that potentially affect fish assemblages with increasing levels of disturbance due to human land use practices. Healthy aquatic ecosystems would be expected to contain a diverse array of habitats and resources for fish refuge, feeding, spawning and larval development. Fish assemblages would be characterised by species from a range of habitats and trophic guilds (varying spatially and temporally with local and landscape scale environmental features), few or no exotic fish and a diverse range of size/age classes. With increasing levels of anthropogenic disturbance, a decrease in the availability and/or suitability of habitat and other resources may be expected, leading to increased potential for biotic interactions and intolerable conditions for sensitive fish species. Fewer native species and more exotic fish would be expected and the fish assemblage may be characterised by low structural and functional diversity (figure modified from Kennard et al. 2001).

5.1.3 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 and Thoms 1999, Gergel *et al.* 2002, Niemi and McDonald 2004). The choice of physical, chemical or biological indicators depends largely on the reasons for undertaking the work or the type of anthropogenic impact(s) to be assessed. A wide range of aquatic organisms has been used as indicators of stream ecosystem health including algae, macrophytes, macroinvertebrates and fish (Norris 1995; Mackay *et al.* 2003). More recently, indicators of ecosystem processes (e.g. benthic metabolism) have been developed and applied (e.g. Bunn 1995; Bunn and 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 and Chu 1999; Kennard *et al.* 2005, 2006a,b) 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 because of factors other than disturbance;
3. site by site differences may be difficult to interpret owing 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 fish 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 fish 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 fish are used widely as indicator organisms (Harris 1995, Kennard 1995; Simon 1999; Kennard *et al.* 2006a,b) and are the focus of this chapter.

5.1.4 Approaches to ecosystem health assessment using freshwater fish

There are two approaches available for examining the role of human disturbance on fish communities in the chosen study area, the Russell/Mulgrave basin: comparative and referential. First, because Behana Creek and the Little Mulgrave River are relatively undisturbed compared with Babinda Creek and Woopen Creek (see Chapters 1 and 2 of this report) we can directly compare various attributes of fish community structure between rivers. However, while Behana Creek and the Little Mulgrave River are comparatively less influenced by human activities, that is not to say that they are entirely without any anthropogenic disturbance, given that their catchments are not entirely forested, free of any agricultural development or residential pressures and associated infrastructure such as roads and bridges, nor are they unaffected by water resource use. Moreover, given that position in the landscape has such an important influence on freshwater fish communities in the Russell/Mulgrave basin (Pusey *et al.* 1995a; Pusey *et al.* 2000) and the wet tropics region in general (Pusey and Kennard 1996), downstream changes in land-use impacts (see Chapters 1 and 2) have significant potential to confound any comparisons between seemingly intact catchments and degraded catchments.

An alternative approach is to use a referential system, in which attributes of fish assemblages at test (i.e., assessment) sites are compared with attributes from a range of reference sites. A critical underpinning of use and interpretation of ecosystem health indicators is the ability to accurately define the expected condition for those attributes upon which the indicators are based (Stoddard *et al.* 2006). 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 and Rosenberg 1989; Grossman *et al.* 1990; Kennard *et al.* 2006a,b). 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 affected 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 that they 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 and Norris 2000; Oberdorff *et al.* 2001a), and 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 in-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 and Norris 2000). The development of multivariate predictive models of fish assemblage composition and their utility in stream bioassessment programs in northern Australia has received little attention. However, fish-based predictive modelling methods have been shown to provide a sensitive tool for biomonitoring river health in south-eastern Queensland (Kennard *et al.* 2005; 2006a,b) as well as in Europe (Oberdorff *et al.* 2001b) and New Zealand (Joy and 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;

Barbour *et al.* 1995; Karr and Chu 1999; 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) and 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 ecological condition in the absence of human disturbance differ 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 strengths of multimetric methods (Karr and Chu 1999). Harris (1995) suggested that multimetric methods such as the IBI are potentially applicable to stream ecosystem health assessment in Australia and, to this end, the IBI has been tested and applied in several rivers of southern Australia (Harris and Silveira 1999; Murray Darling Basin Commission 2004).

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 indicators 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 ecosystem health that is not only rigorous and scientifically defensible but, more importantly, is crucial to justify management interventions and acceptance by the community (Kennard *et al.* 2006a,b).

5.1.5 Aims and structure of chapter

In this chapter we aim to evaluate the extent to which present-day agricultural practices and other anthropogenic stresses have an impact on stream fish in four sub-catchments of the Russell/Mulgrave River basin. The study involved a paired catchment comparison, with two streams sampled in both the Russell and Mulgrave catchments (see Fig. 2.1, Chapter 2). Our aim was to investigate the effects, if any, of contrasting land use and management practices in the two catchments. We applied both a comparative and referential approach to this question. The logic and analytical pathway for the investigation is shown in Figure 5.2.

Firstly, we describe how the study sites vary in terms of position within the catchment and the associated longitudinal changes in habitat structure, and with respect to various land-use pressures and changes to riparian and aquatic vegetation communities (see Chapters 2 and 3, respectively). Next, we document and discuss the freshwater fish communities present in the four sub-catchments of interest and quantify the relationship between species abundances and various landscape and local in-stream habitat features. We then directly compare fish assemblage attributes (fish species richness and assemblage structure) between streams and along the land-use disturbance gradient described in Chapter 2. Following this we compare the observed fish assemblage with the expected assemblage (derived from the reference condition) and the deviation between observed and expected is used as a measure of stream ecosystem health. Figure 5.2 presents the logic and analytical pathway used to investigate the influence of human land use on freshwater fish assemblages in the four chosen tributaries of the Russell/Mulgrave River system.

5.2 Methods

5.2.1 Freshwater fish assemblages

Site selection and quantification

Study sites typically constituted an entire riffle-run sequence, and were between 100 and 120 m in length and positioned at the same locations as associated investigations of geomorphic structure, water quality, riparian structure and condition, and microphyte and macroinvertebrate communities. Data collected as part of these investigations were used to quantify stream ecosystem condition and habitat structure and a full description of the methods used in each is given in the appropriate chapters. In addition to these data, the elevation and distance of each site from the river mouth were estimated from GIS maps to provide a means of locating each study site within each drainage basin and of estimating the expected assemblage of fish at each site (see below). Twenty-six sites were sampled, comprising six sites in Woopen Creek, eight in the Little Mulgrave River, five in Babinda Creek and seven in Behana Creek. Site locations are given in Chapter 2**.**

Fish communities in the headwaters of these drainages were not sampled because they typically contain a very distinctive set of species that are not found at lower elevations nor in areas that are potentially exposed to the impacts of land use and human activity (Pusey *et al.* 2000) and were therefore unlikely to aid our investigation of how such activities affect stream fish. Sampling took place between 27/6/2005 and 19/7/2005 and typically occurred concurrently with riparian and aquatic plant sampling and within 1-2 days of sampling for aquatic macroinvertebrates. Significant rainfall (>150 mm in 2 days) occurred in the catchment during this period and stream flow was sufficiently elevated for a period of 3-5 days after such events to prohibit effective sampling because of elevated turbidity, water depth and velocity. In such cases, sampling was discontinued until stream flow and turbidity levels had returned to levels similar to those occurring at the start of the study. Previous research in this basin (Pusey *et al.* unpubl. data) has shown that minor spates, such as occurred during the study period, do not greatly affect fish communities.

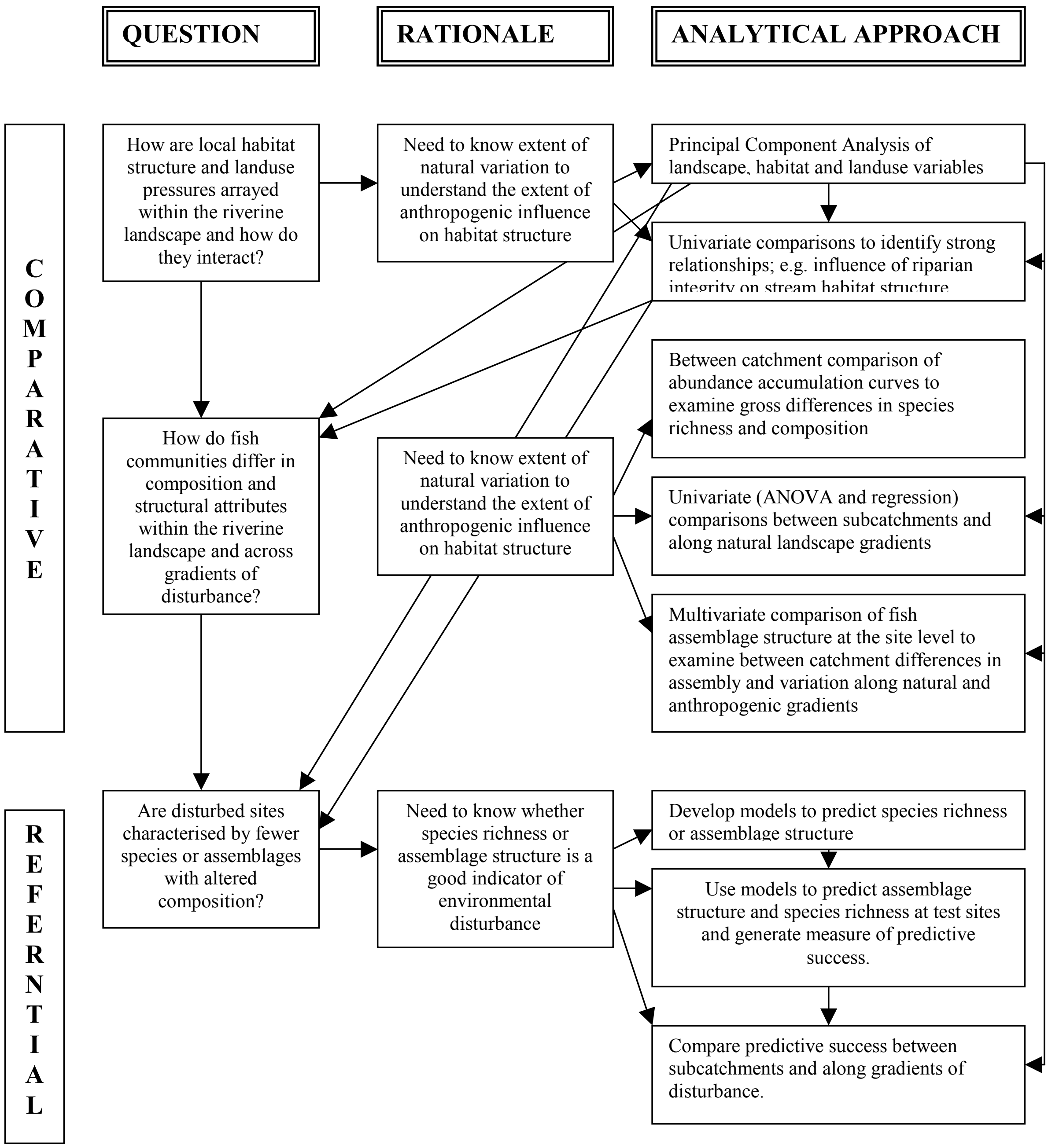

Figure 5.2. Analytical pathway used for investigating the influence of human land use on freshwater fish in the Russell/Mulgrave River

Stream fish sampling procedures and quantification of assemblage structure

Freshwater fish at each site were collected using a combination of electrofishing and seine netting. Electrofishing was conducted using a back-pack DC electrofishing unit (Smith Root MK 12 POW) with a standard Smith Root anode (25cm diameter ring on a 2 m pole) and cathode (3.2 m stainless steel wire cable). The unit was typically operated at 300-400 V, 50-70 Hz frequency and 4 msec pulse width as previous research by the investigators has shown this combination to be most effective in streams of north-eastern Australia (Pusey *et al.* 2004b). A small net was attached to the anode pole to increase capture efficiency. In addition, a second operator with a long handled dip net (50 cm gape, square bottom, mesh size of 6 x 9 mm) accompanied the electrofishing unit to collect all fish stunned but not collected by the primary operator. Two electrofishing passes were conducted in each site and all available microhabitats were sampled during both passes. Typically the operator proceeded along a stream bank for 10 m before moving across the stream bed, whilst sampling, to then sample the opposite bank. This zigzag pattern tends to reduce fright bias of stream fish and lessens the extent to which fish are driven before the operator but not sampled. All fish collected were placed into a large plastic tub and periodically removed and placed in an aerated water container (approx 20 L) on the stream bank. The time required to undertake electrofishing was noted for each site (average = 90.7 ± 6.0 (SE) min) and abundance values for all species were standardised to the number collected in 100 minutes of electrofishing time. Note that this period is the length of time over which electrofishing occurred, not the total “on time” of the unit, which is typically much shorter.
Seine netting was employed after the completion of electrofishing to collect vagile, open water species such as *Melanotaenia splendida* (eastern rainbowfish) and *Craterocephalus stercusmuscarum* (fly-specked hardyhead), which are typically undersampled by multi-pass electrofishing (Pusey *et al.* 1998; Kennard *et al.* 2006c). Two hauls of the seine (30 m length, 2 m drop, stretched mesh size of 11 mm) occurred at each site, except in sites characterised entirely by shallow fast flowing water (i.e., entire site consisted of shallow riffle). In such circumstances, seine netting is not efficient but is not required as open water species are typically collected with sufficient efficiency to preclude its use. The catch by this method was combined with the catch by electrofishing (after standardising to 100 min of effort) to represent the total catch for each site. All analyses were performed on these standardised data. Fish were identified at the time of capture and larger bodied species (*Hephaestus tulliensis, H. fuliginosus, Kuhlia rupestris, Tandanus* sp. and *Neosilurus ater*) were classified as either adults (> 150 mm SL) or juveniles (<150 mm) following Pusey *et al.* (2004b). Eels (*Anguilla reinhardtii*) were divided into four size classes: <10cm, 10-30 cm, 30-60 cm and >60 cm. All fish except alien species were released alive to the area of capture. Alien species were euthanased on site in accordance with the Queensland Fisheries Act, 1994.

5.2.2 Statistical analyses

All statistical analyses other than multivariate examination of spatial variation in fish assemblage structure were performed using the SPSS© 12.0.1 for Windows© statistical package. Multivariate analyses were performed using PATN (Belbin 1995). Details concerning parametric statistical tests are given in the relevant sections.

Site location along natural gradients in the riverine landscape and gradients resulting from human land use.

The structure and functional characteristics of the stream environment change naturally along a river’s longitudinal course. Such natural variation is of profound importance in determining the distribution and abundance of freshwater fish. When human-induced disturbance is imposed on this gradient, any potential impact can only be discerned after natural variation is accounted for. In the present case, disturbance to stream ecosystems may arise from activities that occur within the catchment but are distant from locations of particular interest. Such impacts may therefore indirectly affect the stream environment and may be associated with agricultural development in the surrounding floodplain, industrial activities, water resource use (e.g., upstream dams), or urban and rural residential development. More localised disturbance may occur within the stream channel but not necessarily within the wetted stream itself. Such impacts may be associated with damage to the riparian zone, for example, in which case associated effects may also occur indirectly within the wetted stream (see Pusey and Arthington 2003). Here we are concerned with defining and categorising potential sources of impact associated with human activities and processes occurring within the riverine landscape (catchment and in-stream) and placing them in a context defined by natural variation in stream environments.

When examining spatial variation in parameters along natural gradients, one must be mindful that many variables are intercorrelated. For example, as one moves upstream along a watercourse, variables such as elevation and distance from the river mouth increase whereas other variables such as catchment area decrease. Similarly, we may also reasonably expect variables such as stream width, depth, water velocity and substratum composition to vary along this longitudinal gradient. Moreover, variables describing the nature of the surrounding catchment may also change in a correlated manner. For example, changes in the proportion of catchment that is cleared of natural vegetation, or proportion of catchment used for the production of sugar cane, are likely to be negatively correlated with proportion of catchment covered by intact native forest.

We used Principal Components Analysis (PCA), based on correlations between 26 variables describing position in the landscape, local habitat structure and human land use, to identify a smaller set of variables that were uncorrelated with each other. The initial dataset consisted of four variables describing the position of each site in the landscape, 11 variables describing the physical structure of the in-stream environment, eight variables describing the extent, nature and integrity of the riparian and in-stream vegetation and 11 variables describing the areal extent of various human activities in the catchment (Table 5.1). All variables were measured on-site or estimated from maps or GIS. A description of methods used and their derivation is provided in Chapters 2, 3 and 4. We also had access to a dataset describing water quality conditions (dissolved oxygen concentration, % saturation, conductivity, pH and turbidity – Chapter 2) at each site, but did not include water quality in the PCA for the following reason. Initial examination of the data revealed that the upper and lower limits for the water quality parameters never approached levels that might affect native fish according to information on environmental tolerances contained in Pusey *et al.* (2004b). Their inclusion in a PCA would therefore add little to our ability to discern the factors that were of importance in determining the composition and structure of fish assemblages and, moreover, may have unnecessarily complicated the analysis by increasing the number of Principal Components necessary to describe the full variation in environmental differences between sites and sub-catchments.

5.2.3 Univariate comparison of fish assemblage characteristics

A number of measures of fish assemblage structure were derived from the total catch from each site, including species richness (total number of species collected), native species richness, alien species richness, the proportion of total species richness contributed by native species, total abundance, the proportion of total abundance contributed by native species, diversity (Shannon H) and evenness (Pielou's J). These measures are useful for describing natural variation in fish assemblage structure and have been shown to be helpful in studies of the impact of human disturbance on stream fish communities (Karr 1981). Analysis of variance was used to compare means for each sub-catchment. Levene's test was used to test for variance equality and appropriate transformations were used when variances were shown to be significantly unequal. Some variables differed with position of the site in the riverine landscape and in such cases a General Linear Model analysis of covariance with distance of the site from the river mouth (log transformed) (DISTM) as the covariate was used to test for between-sub-catchment effects. The GLM procedure was first run to test for main effects (sub-catchment and the covariate DISTM) and for an interaction between main effects. In the absence of any significant interaction, the procedure was run again testing only for significance of main effects. In such cases, only the results of the latter test are displayed here.

5.2.4 Multivariate comparison of fish assemblages and relationships with natural environmental gradients and human land-use factors

Ordination of fish taxon (species) abundances using the Semi Strong Hybrid MultiDimensional Scaling (SSHMDS) routine available in PATN (Belbin 1995) was used to compare the structure of fish assemblages at the test sites. Age classes were combined and all alien species were pooled because their low abundances did not warrant analysis of individual species. Abundance data were log(x+1) transformed prior to analysis and the site-by-site association matrix was based on the Bray Curtis measure of dissimilarity. The ordination was based on three vectors in order to reduce the resultant stress below 0.15. The PCC routine, using the Varimax rotated solution, was used to test for significant associations between site position in ordination space and the abundance of individual species, and the various landscape, in-stream and catchment land-use variables detailed above. The significance of such correlations was assessed using the MCAO module available in PATN.

5.2.5 Referential approaches to discerning stream health using native fish as indicators

As discussed in the introduction to this chapter, attempts to discern changes in freshwater fish assemblage structure have been centred on using a referential approach as well as a comparison of sub-catchment variation in fish assemblage attributes (i.e., natural *versus* potentially degraded assemblages, as described in Section 5.2.3 above). We focussed on using existing data to predict how many native species and what array of species occurred at each stream location as a means of providing a reference condition against which departures from expected fish assemblage structure could be assessed and related to existing land-use pressures. We were aided in this approach by having access to data from several different sources currently being assembled to provide an Atlas of Freshwater Fishes of the wet tropics region (Pusey *et al.* in prep.). These data consist of species lists based on over 500 point locations in all of the major basins of the wet tropics region.

Table 5.1. Environmental variables estimated at each sampling location.

Variable (measurement unit)	Description	
Elevation (m.a.s.l.)	Estimated from GIS maps	
Distance from river mouth (km)	Estimated from GIS maps	
Site gradient (%)	Measured for entire study reach using staff, dumpy level and tripod	
Total riparian cover (%)	Assessed using a spherical densiometer. Value represents means of approximately 30 readings taken at Left Bank, Centre and Right Bank at 10 meter intervals	
Width (m)	Average of ten readings	
Depth (m)	Three quadrats within multiple transects (see Section 5 for details)	
Velocity ($m.sec^{-1}$)	Approx. n = 30	
Substrate composition (% surface area)	Visually estimated for multiple quadrats within multiple transects (see Section 5)	
Mud	<0.06 mm (particle size)	
Sand	0.06-2.0mm	
Fine Gravel	2.00-16.0mm	
Gravel	16.0-64.0mm	
Cobbles	64.0-128.0mm	
Rock	>128mm	
Bedrock		
Total riparian condition (%)	See Section 5 for details	
Total aquatic microphyte cover	See section 5 for details	
(Braun Blanqet scale)	Cover rating	Braun Blanquet cover scale
Submerged macrophyte cover	0	0
Emergent macrophyte cover	1	<1%
Terrestrial vegetation	2	1-5%
Paragrass	3	6-10%
Singapore Daisy	4	11-25%
	5	26-50%
	6	51-75%
	7	76-100%

5.2.5.1 Predicting native species richness in test sites

A dataset consisting of 129 sites within the Russell/Mulgrave drainage system was extracted from the fish Atlas (Pusey *et al.* in prep.). Data sources from which data were accessed included: 1) Pusey *et al.* (1995a) – single pass electrofishing over 100-200 m of stream length encompassing multiple hydraulic habitat units; 2) Russell *et al.* (1996) – single pass electrofishing over 100-200 m of stream length encompassing multiple hydraulic habitat units; and 3) Pusey and Kennard (1996), Pusey *et al.* (2000) – multiple pass electrofishing within single hydraulic units. Previously sampled locations within the test streams (i.e., Woopen Creek, Babinda Creek, Behana Creek and the Little Mulgrave River) or within the Alice River and tributaries feeding into Eubanangee Swamp were excluded. Sites within the streams selected for this study were excluded because of concerns that any existing disturbance within these streams at the time of sampling during the 1990s had the potential to bias predictions of species richness, and this would preclude their use as reference streams. Similarly, habitats of the Alice River and associated tributaries, including Eubanangee Swamp, are highly dissimilar to those found within the test wet tropics streams and therefore their inclusion in any predictive model, while increasing our ability to predict species richness over the entire catchment, would in all likelihood decrease or, at least, not enhance our ability to predict species richness in the streams of our test drainages. The remaining data set comprised 78 sites.

Initial exploratory correlation analysis revealed that species richness was significantly correlated with both elevation (log transformed, Pearson's $r = -0.591$, $p<0.001$) and distance from the river mouth ($r = -0.468$, $p<0.001$) paralleling similar findings in Pusey *et al.* (1995) and Pusey *et al.* (unpublished data). These two variables were then used in a multiple regression analysis (simultaneous entry). The resultant model: spp = 15.457 – 0.019(DISTM) – 3.309(logELEV); was highly significant ($F = 20.338$, $p<0.001$, $r = 0.57$). Unstandardised residuals were calculated and inspection revealed that several sites contained abnormally few species (i.e., three or more fewer than predicted). These sites may have been inadequately sampled given that single pass electrofishing, as opposed to multiple pass electrofishing, as used here and recommended in Pusey *et al.* (1998), was the dominant method used for collecting fish at these sites. Furthermore, a proportion of the study sites derived from Russell *et al.* (1996) were located in degraded streams and therefore they were unsuitable as reference sites. All such sites (n=17) were excluded from further analysis.

The remaining sites were then used in a further multiple regression analysis. The resulting model was, as expected, able to account for a higher proportion of the variance in species richness (59.4% *vs* 35.2%) and was highly significant ($F= 42.515$, $p<0.001$). The relationship was: spp = 18.531 – 0.035(DISTM) – 3.979(logELEV). This relationship (Fig. 5.3) predicts a decrease in species richness with increasing elevation and distance from the river mouth but importantly, also predicts that declines in richness with increasing altitude are less in adventitious streams located close to the river mouth. The resultant model was used to predict species richness at sites within the test drainages and the results are depicted in Figure 5.3. The ratio of the number of species observed at each site (O) to that predicted (P) was calculated for each site.

Predicting assemblage composition in test sites

Previous research has shown that elevation and distance from the river mouth are both powerful determinants of assemblage structure in addition to the number of species present at individual locations in streams of the wet tropics region (Pusey and Kennard 1996; Pusey *et al.* 2000; Pusey *et al.* in prep.). This effect is associated with predictable changes in habitat structure associated with increasing elevation and distance from the river mouth (these variables are highly intercorrelated) and the influence of habitat structure on fish assemblage structure. Upstream locations, for example, tend to be of higher gradient with faster current speeds and coarser substrata and contain only those species favoured by such habitats. In addition, a very high proportion (>50%) of the fauna typical of streams of the wet tropics region has a marine or estuarine life history phase (usually the larvae or juveniles) and must migrate upstream to the preferred adult habitat (Pusey *et al.* 2004b). Consequently, the combined effects of increasing distance and elevation on migrating fish sequentially "filter out" such species so that a predictable decline in species richness and change in composition occurs.

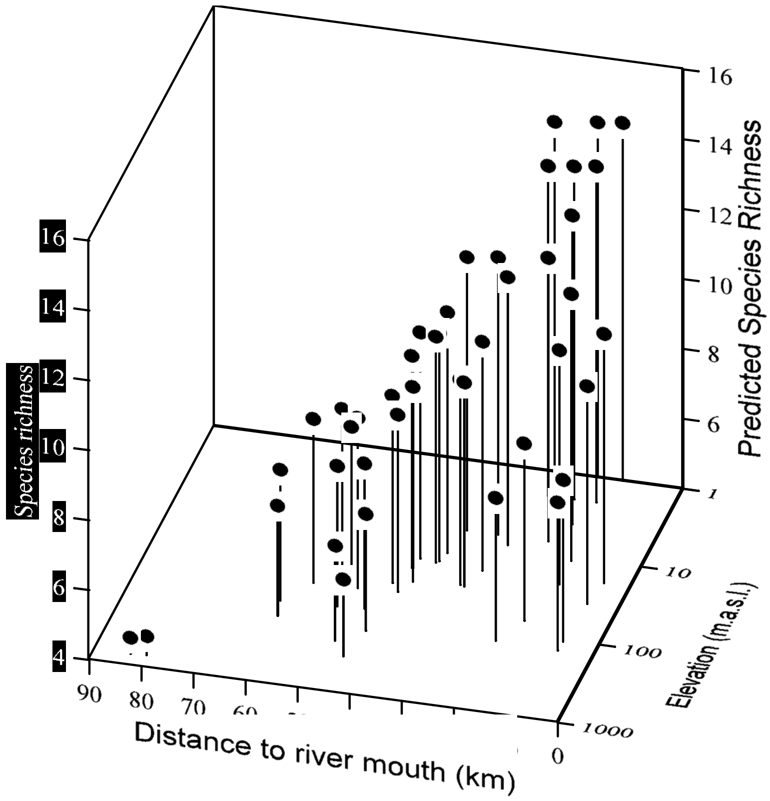

Figure 5.3. Spatial variation in predicted species richness at test sites along the gradients of distance from the river mouth (km) and elevation (m.a.s.l.)

The overriding importance of the two variables of elevation and distance upstream from the river mouth, which are surrogates for a range of other correlated environmental factors such as physical habitat structure and water quality, ensures that a predictive model based solely on these two variables is both simple to construct and has biological relevance. We used the same data-set previously used to predict native species richness in reference and test sites as the basis for a model to predict assemblage composition. The reference sites were plotted against elevation and distance from the river mouth (DISTM) (Fig. 5.4). A series of grids was then imposed upon the distribution in an attempt to partition the total sample in a reasonably small number of groups (to ensure that within-group sample sizes remained as large as possible) whilst ensuring that biologically relevant between-group differences in assemblage composition were maintained. Seven groups (A-G) were recognised. We then used these data in a SSHMDS ordination (PATN) to examine the extent of between-group differences in assemblage composition.

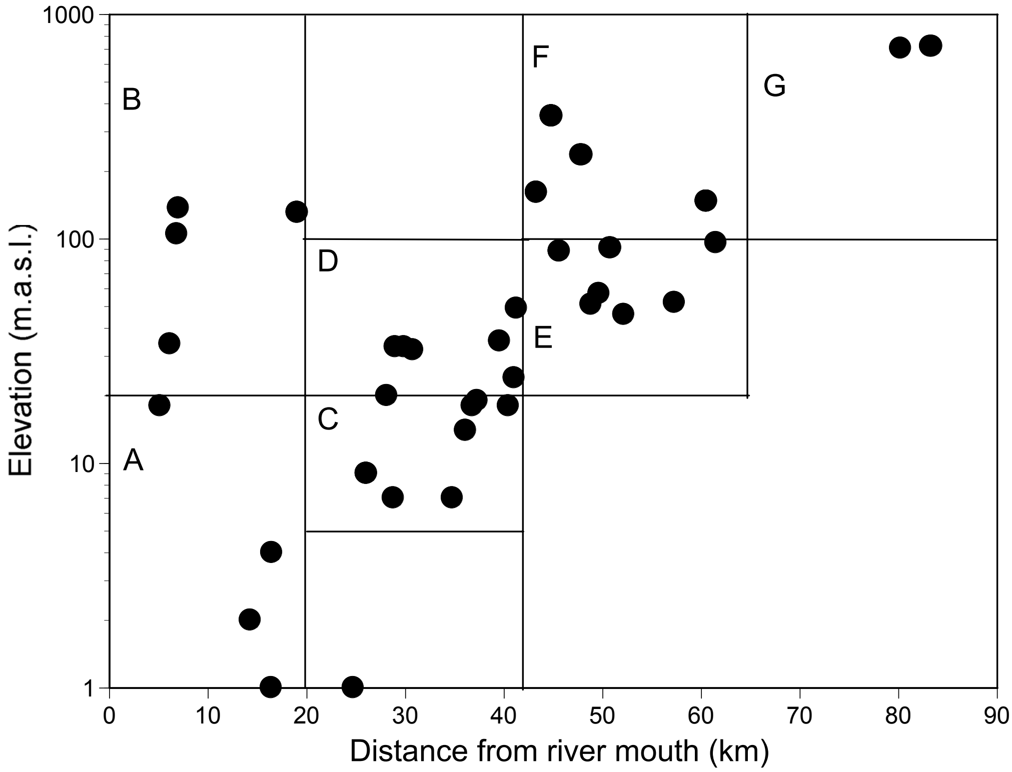

Figure 5.4. The position of reference sites within the riverine landscape as determined by elevation and distance upstream from the river mouth. Note that the some sites situated close to one another are obscured on the figure. The lines on the figure denote sites groups defined on the basis of similarities in proximity to the river mouth and elevation. Sample size for each group is given in Table 7.8.

We also used the group structure revealed in Figure 5.4 to define species useful as indicators of that group, which when combined serve to act as reference indicator communities. A reference community is that collection of species that is reasonably expected to occur at a site that is in good condition and is not affected by human activities. Deviation away from that expected community may provide an assessment of the extent of the impact of human activity on that location. Two levels of frequency of occurrence were used to construct the expected community for each group. The first included only those species present in 50% or more of sites within each group, while the second included species occurring in 30% or more of sites. These two communities were designated E_{50} and E_{30}, respectively. Similar approaches (e.g. Wright *et al.* 1984) have further reduced the expected species richness at reference sites to account for expected levels of spatial variation in species richness within individual reference groups in the following manner. The frequency of occurrence of individual species within each group is calculated and summed for all species present in 50% or more of sites (or 30% or more as in the present case). Thus the expected species richness will always be less than the total number of species occurring in 50% or more of sites (unless of course those species occurred in 100% of sites). We did not use this additional step in the present study. To assess the extent of variation in deviation away from the expected condition within our reference sites, we compared observed (O) with expected (E) communities, calculated O/E_{50} and O/E_{30} scores for each reference site and derived the median, 10^{th} %ile and 90^{th} %ile O/E scores for the entire sample.

5.3 Results

5.3.1 General

A total of 3692 fish from 30 species (27 native; 3 alien) within 15 families was collected during the study (Table 5.2). Fourteen of the native species collected have life histories completed entirely within freshwater, with the remaining 13 species requiring access to estuarine or marine areas for spawning or larval development. Exclusively freshwater species contributed equivalent proportions to the total species richness for each sub-catchment (44.4%, 48.0%, 42.1% and 45% for Babinda Creek, Behana Creek, the Little Mulgrave River and Woopen Creek, respectively). The proportion of the total number of fish collected that were exclusively freshwater species was greatest in the two most upstream sub-catchments: the Little Mulgrave River (60.2%) and Woopen Creek (58.8%); and lowest in the two most downstream sub-catchments: Behana Creek (54.2%) and Babinda Creek (52.6%).

Total native species richness varied between the sub-catchments (Fig. 5.5) with the Little Mulgrave River, Woopen Creek, Behana Creek and Babinda Creek containing 19, 19, 25 and 18 species, respectively. *Cairnsichthys rhombosomoides* and *B. gyrinoides* were recorded from Woopen Creek but not the Little Mulgrave River, whereas *Schismatogobius* sp. and *G. margaritacea* were present in the Little Mulgrave River but not Woopen Creek. All four species were rare, never exceeding 0.2% of the total collected in either drainage. It is probable that these species were present in both sub-catchments but were not sampled in both because of their rarity. Species absent from Babinda Creek but present in Behana Creek (*P. gertrudae* (3.3% of Behana Creek total)*, C. rhombosomoides* (0.7%)*, R. bikolanus* (1.2%)*, M. adspersa* (2.3%)*, E. fusca* (0.6%)*, B. gyrinoides* (0.1%) and *G. margaritacea* (0.4%)) comprised a mix of both rare (<0.2%) and more common species. It is worth noting that only a single adult individual of the jungle perch (*K. rupestris*), a species of conservation significance (Pusey *et al.* 2004b) was recorded from Babinda Creek, whereas both juvenile and adult specimens were recorded from the remaining sub-catchments (6, 8 and 10 individuals from the Little Mulgrave River, Woopen Creek and Behana Creek, respectively).

Six species (*Pseudomugil signifer, Melanotaenia splendida, Glossogobius* sp. 4, *Anguilla reinhardtii, Craterocephalus stercusmuscarum* and *H. compressa* (in decreasing order of abundance)) collectively comprised 78.4% of the total number of fish collected. These species were typically the most abundant species in each of the four sub-catchments although the order of abundance varied between sub-catchments (Fig. 5.5). For example, *P. signifer* was very abundant in Woopen Creek (almost 40% of total) but was only the 3rd, 4th and 6th most abundant species in the Little Mulgrave River, Behana Creek and Babinda Creek, respectively. Only three species of alien fish were collected throughout the study area (*Poecilia reticulata, Xiphophorous maculatus* and *Tilapia mariae*) and these species collectively comprised less than 1% of the total. Alien species were present in Babinda Creek (*P. reticulata* and *X. maculatus*), Behana Creek (*P. reticulata* and *T. mariae*), Woopen Creek (*P. reticulata*) and the Little Mulgrave River (*P. reticulata*), where they collectively comprised 3.6%, 0.8%, 0.4% and 0.2% of the total, respectively.

Table 5.2. Freshwater fish species recorded from Behana, Babinda and Woopen Creek and the Little Mulgrave River. Also shown is the proportional contribution by each species to the total collected during the study and the reproductive mode of each species (based on information in Pusey et al. 2004). FW = entirely freshwater, E/M = estuarine or marine, ? insufficient information available to make an assessment.

Species	Common name	Proportion of total (%)	Reproductive mode/Larval habitat
NATIVE SPECIES			
Pseudomugilidae			
Pseudomugil signifier (Kner)	Pacific blue-eye	22.0	FW
Pseudomugil gertrudae Weber	Spotted blue-eye	0.1	FW
Melanotaeniidae			
Melanotaenia splendida splendida (Peters)	Eastern rainbowfish	17.1	FW
Melanotaenia maccullochi Ogilby	Macculloch's rainbowfish	0.2	FW
Cairnsichthys rhombosomoides (Nichols & Raven)	Cairns rainbowfish	0.2	FW
Atherinidae			
Craterocephalus stercusmuscarum stercusmuscarum (Günther)	Fly-specked hardyhead	6.8	FW
Gobiidae			
Awaous acritosus (Watson)	Roman nosed goby	0.8	E/M
Redigobius bikolanus (Herre)	Speckled goby	1.3	
Glossogobius sp.1	Mountain goby	1.0	E/M
Glossogobius sp. 4	Mulgrave River goby	18.1	E/M
Schismatogobius sp.	Scaleless goby	0.2	?
Eleotridae			
Mogurnda adspersa (Castelnau)	Purple spotted gudgeon	1.9	FW
Hypseleotris compressa (Krefft)	Empire gudgeon	6.5	FW, E/M
Eleotris fusca (Bloch & Schneider)	Brown gudgeon	0.1	E/M
Oxyeleotris aruensis (Weber)	Aru gudgeon	0.2	FW?
Bunaka gyrinoides (Bleeker)	Green back guavina	0.1	E/M
Giuris margaritacea (Valenciennes)	Snakehead gudgeon	0.1	E/M
Ambassidae			
Ambassis agrammus Günther	Sailfin perchlet	0.4	FW, E/M
Kuhliidae			
Kuhlia rupestris (Lacépède) (<150 mm)	Jungle perch	0.4	E/M
Kuhlia rupestris (>150 mm)		0.1	
Terapontidae			
Hephaestus tulliensis DeVis (<150 mm)	Tully grunter	3.8	FW
Hephaestus tulliensis (>150 mm)		0.4	FW
Plotosidae			
Tandanus sp. (<150 mm)	Northern eel-tailed catfish	0.7	FW
Tandanus sp. (>150 mm)		1.7	
Neosilurus ater (<150 mm)	Black catfish	0.1	FW
Neosilurus ater (>150 mm)		0.1	
Porochilus rendahli	Rendahl's catfish	0.1	FW
Anguillidae			
Anguilla reinhardtii Steindachner (<10cm)	Pacific long-finned eel	2.9	E/M
Anguilla reinhardtii (10-30cm)		7.4	
Anguilla reinhardtii (30-60 cm)		1.5	
Anguilla reinhardtii (>60cm)		1.0	
Synbranchidae			
Ophisternon gutterale (Richardson)	Swamp eel	0.4	FW
Apogonidae			
Glossamia aprion (Richardson)	Mouth almighty	0.7	FW
Scorpaenidae			
Notesthes robusta (Günther)	Bullrout	0.9	E/M
ALIEN SPECIES			
Poeciliidae			
Poecilia reticulata (Peters)	Guppy	0.4	FW
Xiphophorus maculatus (Günther)	Platy	0.4	FW
Cichlidae			
Tilapia mariae Boulenger	Tilapia	0.1	FW, E/M?

5.3.2 Land use in the sub-catchments of the Russell/Mulgrave River

Between 85% and 98% of the catchments upstream of each study site within each sub-catchment were covered by forest (Table 5.3) with significantly more of the Woopen Creek catchment cleared than that of Behana Creek and the Little Mulgrave River. The next largest land use in the study area was sugar cane production with between 1.3% and 6.0% of the catchments being devoted to this purpose. Horticulture, principally the production of bananas, was important in Woopen Creek (5.8%) but virtually absent from the remaining three sub-catchments. Similarly, cattle-grazing was effectively confined to this sub-catchment also. The remaining land-use types shown in Table 5.3 were of very minor importance in terms of area covered, although significant between-catchment variation in mean proportion was detected by ANOVA.

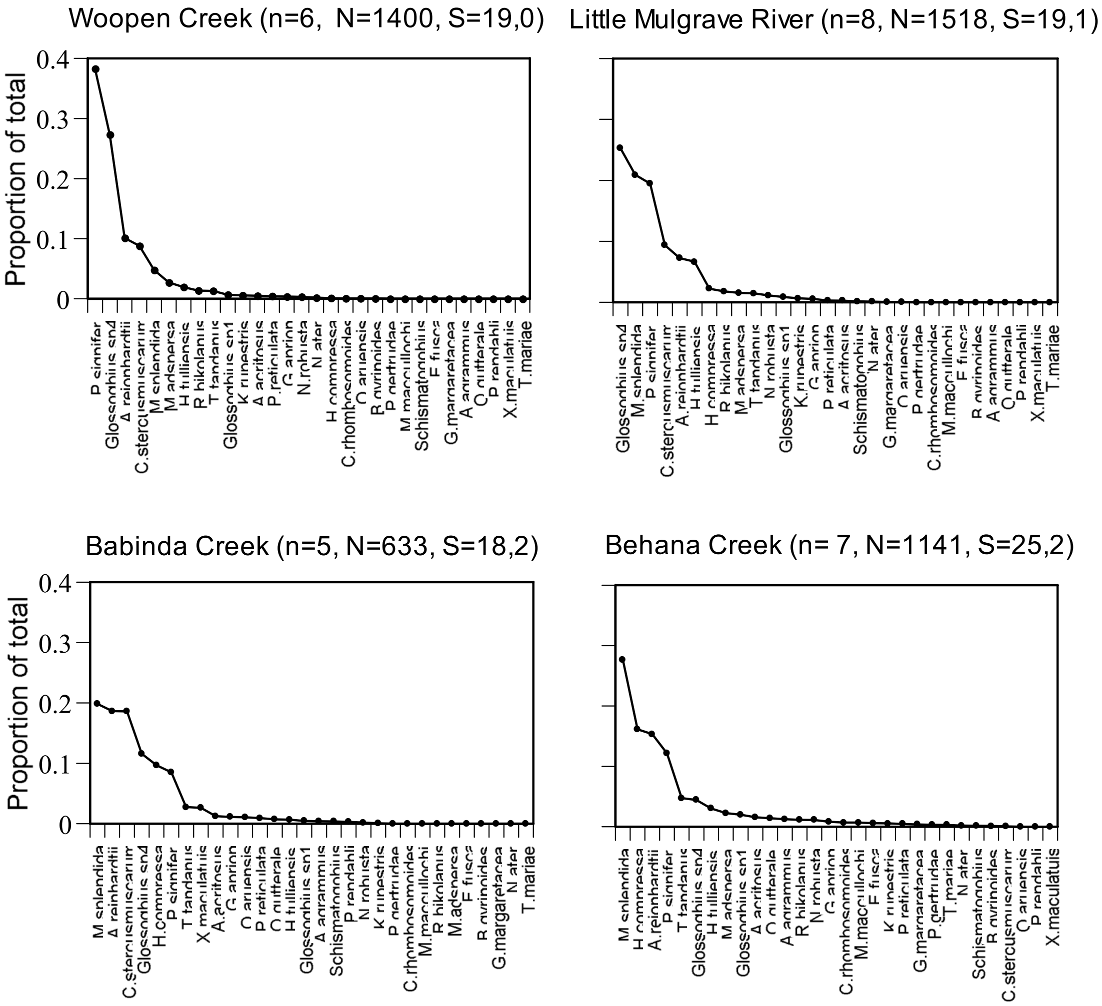

Figure 5.5. Relative contribution by individual species to the total number of fish collected from each of the subcatchments sampled. Also given are the number of locations examined in each subcatchment (n), the number of individuals collected (N) and the number of native and alien species (S), respectively, collected from each subcatchment.

Principal Component 1 (PC1) accounted for 22.2% of the variance observed and equated to a gradient principally in the integrity of the riparian zone and associated changes to in-stream habitat structure. Sites located negatively on PC1 had an intact riparian zone providing good overhead cover, whereas sites located positively had disturbed riparian zones and in the case of some sites in Woopen Creek and especially Babinda Creek, almost no riparian trees at all (Fig. 5.6). As riparian cover and integrity decreased, para grass (*Urochloa mutica*), emergent vegetation and total in-stream vegetative cover increased; this inverse relationship was evidently a consequence of greater light availability. PC1 also represents a gradient in substrata composition that was unrelated to the position of sites within the riverine landscape. Sites located negatively on PC1 had proportionally more rocky substrata whereas sites located positively had proportionally more gravel. In addition, average water velocity was greater in sites with a degraded riparian zone. Sites located in Woopen Creek and Babinda Creek tended to have degraded riparian zones although the most upstream site in Babinda Creek had a relatively intact riparian zone (Fig. 5.6). Sites in the Little Mulgrave River were most frequently characterised by intact riparian zones. Sites within Behana Creek represented a mix of sites with both upstream and downstream sites being of relatively high integrity and two sites located in the middle reaches having a disturbed riparian zone.

Table 5.3. Land use within each of the four sub-catchments. Data shown are the mean and SE of the percentage of the upstream catchment area devoted to each land-use type. Also shown are the F value and associated level of significance (P) for ANOVAs testing for between sub-catchment differences in mean value. Significant differences(P<0.05) determined by LSD multiple comparison tests are indicated by the superscripts.

Land-use	Woopen Creek	Little Mulgrave River	Babinda Creek	Behana Creek	$F_{3,22}$	P
Native forest	84.21 ± 3.76^{1}	97.9 ± 0.56^{2}	$91.48 \pm 3.58^{1,2}$	93.86 ± 2.34^{2}	5.265	0.007
Sugar cane	3.92 ± 1.56	1.29 ± 0.37	5.83 ± 2.71	6.02 ± 2.28	1.709	0.194
Horticulture	5.76 ± 1.29^{1}	0.14 ± 0.03^{2}	0.49 ± 0.16^{2}	0.0 ± 0.0^{2}	21.023	0.000
Grazing	6.03 ± 1.20^{1}	0.0 ± 0.0^{2}	0.88 ± 0.17^{2}	0.0 ± 0.0^{2}	26.423	0.000
Rural residential	0.0 ± 0.0^{2}	0.58 ± 0.15^{1}	0.14 ± 0.05^{2}	0.10 ± 0.04^{2}	7.029	0.002
Urban	0.0 ± 0.0	0.01 ± 0.00	0.36 ± 0.32	0 ± 0	1.794	0.178
Plantation	0.0 ± 0.0^{1}	0.0 ± 0.0^{1}	0.25 ± 0.01^{2}	0.0 ± 0.0^{1}	447.547	0.000
Industrial	0.0 ± 0.0	0.9 ± 0.0	0.02 ± 0.02	0.0 ± 0.0	1.481	0.247
Waste treatment	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.01 ± 0.01	2.143	0.124
Water resources	0.0 ± 0.0^{1}	0.0 ± 0.0^{1}	0.0 ± 0.0^{1}	0.01 ± 0.01^{2}	3.992	0.021
Mining	0.07 ± 0.07	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	1.128	0.359

5.3.3 Site location along natural gradients in the riverine landscape and gradients resulting from human land use

Eight principal components, each with an eigen value greater than 1 and collectively accounting for 86.3% of the total observed variation, were recognised after varimax rotation with Kaiser normalisation. The proportion of the total variation accounted for by each factor is given in Table 5.4.

Table 5.4. Eigen values and proportion of variance (%) accounted for by each of the first eight components after varimax rotation.

Component	Eigen value	% of variance	Cumulative % of variance
1	7.560	22.234	22.234
2	6.504	19.130	41.364
3	3.118	9.170	50.534
4	3.117	9.168	59.702
5	2.833	8.333	68.034
6	2.700	7.941	75.975
7	1.826	5.371	81.346
8	1.659	4.88	86.226

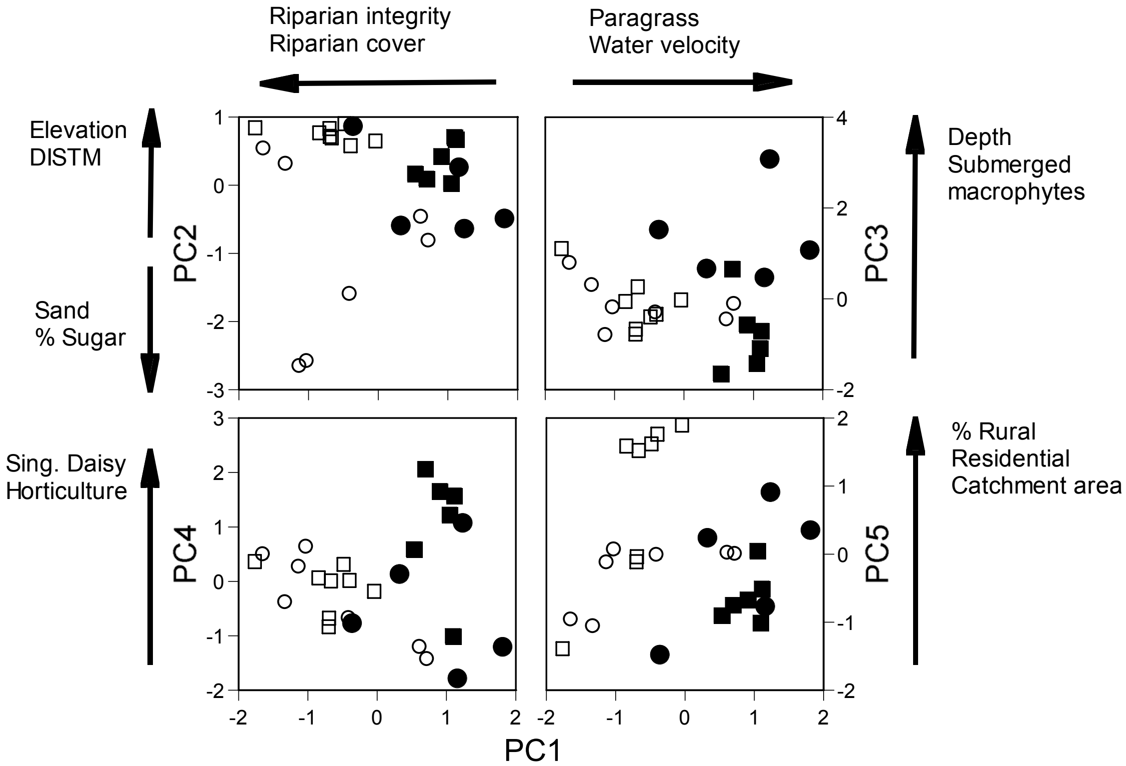

Figure 5.6. Location of the study sites within each sub-catchment within the space defined by the first 5 principal components. ■ = Woopen Creek, □ = Little Mulgrave River, ● = Babinda Creek, and ○ = Behana Creek.

PC2 (19.1%) represents a gradient related to position of the study sites in the riverine landscape (Table 5.5). Sites arrayed positively on this component were situated at high elevations in the catchment and were increasingly distant from the river mouth. Such sites were characterised by stream beds containing higher amounts of cobble. In contrast, sites located negatively on this component had stream beds dominated by sand. This component also represents a gradient in the proportion of the catchment devoted to the production of sugar cane, with a higher proportion of the catchment devoted to sugar cane production at the most downstream sites. It is notable that this component also represents a gradient in the proportion of the catchment upstream of each site devoted to water resource development. The two most upstream sites on Behana Creek had 5% and 3% of their catchments used for water harvesting, reflecting the influence of the impoundment on the upper-most part of this drainage. PC3 (9.2%) represents a gradient in stream depth and the abundance of submerged macrophytes. PC4 (9.2%) represents a gradient in the amount of the catchment upstream of each site devoted to horticultural production, principally bananas, and the extent of invasion of the stream channel by the alien weed known as Singapore daisy. In addition, sites located negatively on component 4 tended to have a higher proportion of their catchment under native forest. It is important to note that, on average, native forests comprised 92.4% ± 1.6% (SE) of the land use for these sub-catchments (Table 5.3). PC5 (8.3%) represents a gradient in catchment size and the extent of rural residential development. The remaining three components represent gradients in human activities in the catchment as well as stream size and stream gradient.

Table 5.5. Loadings of individual variables on components with eigen values >1.

Variable	PC1	PC2	PC3	PC4	PC5	PC6	PC7	PC8
Emergent vegetation	0.912							
Riparian cover	-0.862							
Para grass	0.827							
Riparian integrity	-0.826							
Gravel	0.776							
Velocity	0.639							
Rock	-0.625							
Sand		-0.932						
Water resources		0.828						
Sugar cane		-0.817						
Distance to mouth		0.781						
Elevation		0.779						
Waste treatment		-0.772						
Cobble		0.690						
Depth			0.829					
Submerged vegetation			0.693					
Singapore daisy				0.735				
Horticulture				0.726				
Forest				-0.689				
Rural residential					0.904			
% total catchment area					0.773			
Catchment area					0.719			
Industrial						0.968		
Urban						0.965		
Mining							0.788	
Width								-0.685
Gradient								0.615

5.3.4 Changes in riparian vegetation and its impact on in-stream habitat

The association between riparian condition, in-stream bank-associated weeds and water velocity indicated by loadings on PC1 is particularly noteworthy. Sites with poor riparian condition, exclusively within Babinda and Woopen Creeks, had very abundant infestations of emergent weeds (Pearsons r for correlation between riparian condition and % emergent weeds = -0.760, $p<0.001$), principally para grass ($r = -0.859$, $p<0.001$) but including Singapore daisy in some locations (Fig. 5.7). Sites of poor riparian condition also had elevated water velocities, although it is evident from Figure 5.7 that sites with intact riparian corridors of good condition were occasionally characterised by high average water velocities, in all likelihood because of comparatively higher channel gradients at these sites. However, it is notable that gradient and average water velocity were not associated with the same Principal Component, suggesting that factors other than gradient are more important in determining average water velocities within the stream channel. Average water velocity was, however, significantly negatively correlated with average water depth ($r = -0.507$, $p<0.05$) and it is notable that three sites within Babinda Creek had much higher water velocities than expected from their depths. Moreover, these sites were all located in the most downstream portion of this stream (Fig. 5.7).

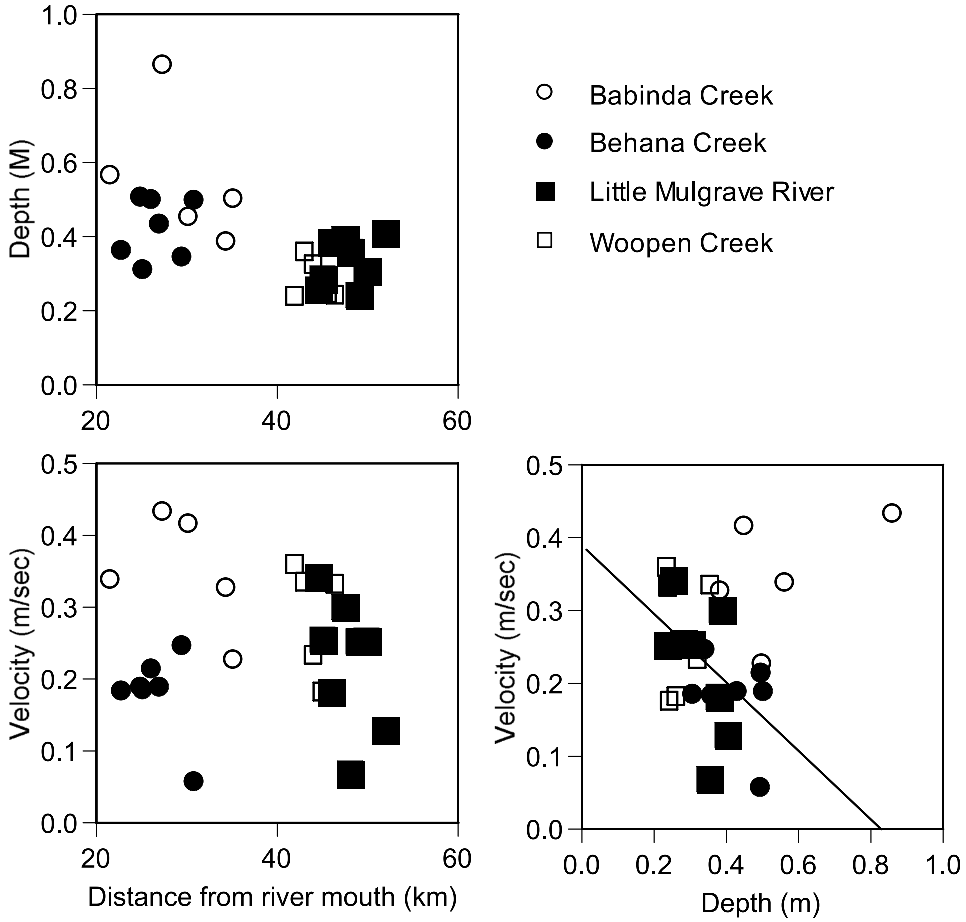

Figure 5.7. Longitudinal variation in mean depth and water velocity and the relationship between the two habitat variables. Significant regression relationships are denoted by the solid line.

Average water velocity was significantly negatively correlated with riparian condition (r = -0.605, p<0.001). Significant positive associations between average water velocity and total emergent vegetation (r = 0.489, p<0.05) and especially para grass (r = 0.523, p<0.01) were also detected (Fig. 5.8). When para grass, Singapore daisy and total emergent vegetation scores plus riparian condition were used as potential predictors of water velocity in a multiple regression analysis with step-wise variable entry, only riparian condition was identified as a significant predictor (i.e., partial correlations of other variables after accounting for the main effect of riparian condition were non-significant). Riparian condition is a collective metric describing a range of factors including total canopy cover, presence of alien weeds etc. (see Chapter 3) and it is important to consider what component of this metric is of importance in influencing stream water velocity. We re-analysed the data using mean riparian cover rather than condition as the predictor variable and this variable accounted for a greater proportion of the observed variation than did condition alone (r = -0.665, p<0.001). Finally, we used riparian cover and site gradient in a multiple regression analysis and the two variables accounted for almost 60% of the total variation observed (gradient accounted for an additional 13.6% after variation resulting from riparian cover had been accounted for: combined $F_{2,23}$ = 15.837, p<0.001). The influence of riparian cover on average water velocity probably arose because the canopy cover is reduced as the riparian zone is degraded, allowing more sunlight to reach the stream channel. Increased sunlight encourages weed growth on the banks, which then extends out into the stream channel, effectively reducing width and channelling flow into a reduced space. As a consequence, average water velocity is increased. In Babinda Creek, bank-associated vegetation increased water velocity and depth (Fig. 5.8).

5.3.5 Sub-catchment differences in fish assemblage characteristics

Analysis of variance of between-sub-catchment differences in assemblage characteristics revealed that significant differences were limited to comparisons of native species richness, diversity and evenness (Table 5.6). Mean native species richness was significantly higher in Behana Creek than in Babinda Creek but all other comparisons were not significantly different (Fig. 5.9). Mean diversity was significantly higher in Behana Creek than in Woopen Creek but all other comparisons were not significantly different. Mean evenness was significantly lower in Woopen Creek than in all other sub-catchments (Table 5.6 and Fig. 5.9). Estimates of abundance (number of fish collected in 100 minutes of electrofishing) of both total and native species only were highly variable. Consequently comparisons of between-catchment differences were not significant for either parameter. Nonetheless, there is some suggestion from the data presented in Figure 5.9 and because significance levels for the appropriate ANOVA comparisons were comparatively low (i.e. ~ p = 0.1), that mean abundance levels in Babinda Creek were depressed in comparison with other sub-catchments. Additional ANOVA comparisons of log (x+1) abundance data elevated the F value but comparisons remained non-significant ($F_{3,22}$ = 2.428, p = 0.092 and $F_{3,22}$ = 2.627, p = 0.076 for comparisons of log(x+1) transformed total abundance and total native abundance, respectively).

Several assemblage characteristics varied with position of the study sites in the riverine landscape (Fig. 5.10). Alien species richness, alien species abundance (log(x+1) transformed), diversity and evenness all declined significantly with log transformed distance from the river mouth (r = -0.567, p<0.01; r = -0.466, p<0.05; r = -0.543, p<0.01, and r = -0.490, p<0.05, respectively), whereas the proportion of total species richness and abundance contributed by native species increased with increasing distance upstream (log transformed) (r = 0.518, p<0.01 and r = 0.434, p<0.05, respectively). Consequently, between-catchment differences may be better explained after removing the co-varying effect of proximity to the river mouth.

The results of ANOVA comparisons in which distance from the river mouth was included as a co-variate indicate that alien species richness was significantly lower in Woopen Creek compared with all other sub-catchments and accordingly the proportion of the total number of species contributed by native species was also significantly higher in this sub-catchment (Table 5.7). Despite failure to identify conclusively which sub-catchment contained the greatest number of individuals of alien species, a significant sub-catchment effect was detected and the mean abundance of alien species in sites within Babinda Creek was approximately four times greater than in any other sub-catchment. The significant interaction between sub-catchment and distance from the river mouth (Table 5.7) is evident in Figure 5.10. The most downstream site in Babinda Creek contained over 20 individuals of alien poeciliids (mostly *X. maculatus*). Similarly, the proportion of the total mean abundance contributed by native species was lowest in the most downstream site of this sub-catchment (Fig. 5.10). Diversity, but not evenness, differed significantly between sub-catchments but this effect was limited to significant differences between Woopen Creek and Behana Creek only. Note that the significant between-catchment differences in evenness revealed by ANOVA (Table 5.7) were no longer evident when co-variation with distance downstream was accounted for.

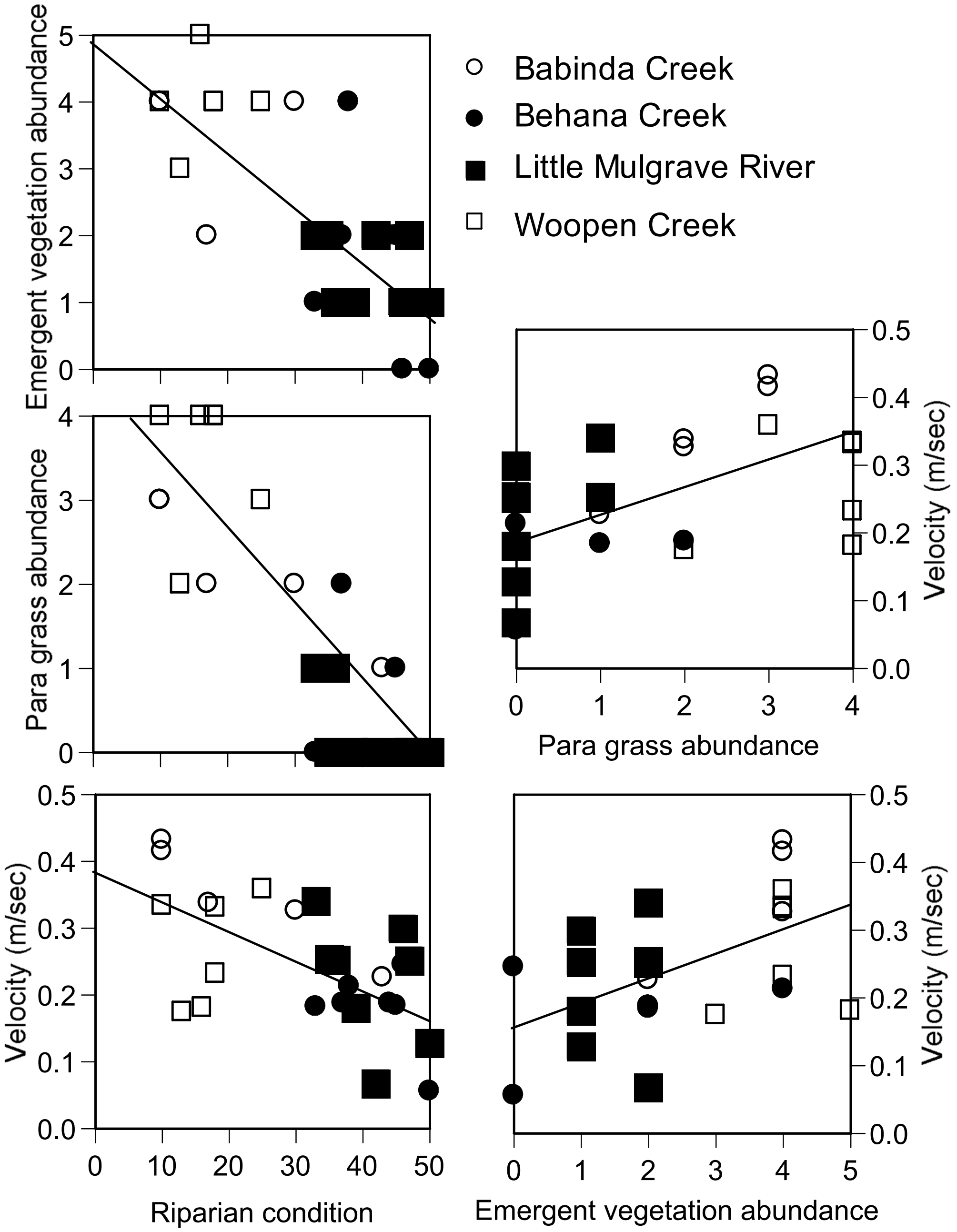

Figure 5.8. Relationships between riparian condition, weed abundance and average water velocity in the Little Mulgrave River and Babinda, Behana and Woopen Creeks. Significant relationships ($p<0.05$) are indicated by the solid lines in each panel.

Table 5.6. F values and associated statistics for ANOVAs testing for between sub-catchment differences in freshwater fish assemblage characteristics. Also shown the results of *post hoc* multiple comparison tests. Significant differences ($P < 0.05$) between groups are denoted by different superscripts. * = $P<0.05$; ** $p<0.01$.

Variable	d.f.	F value	Significance	Post-hoc test
Species richness	3,22	2.841	0.061	
Native species richness	3,22	3.072	0.049*	Bb^{1} WC^{1,2} LM^{1,2} Bh^{2}
Alien species richness	3,22	1.124	0.361	
Proportion native species	3,22	1.009	0.407	
Total abundance	3,22	2.299	0.105	
Total native species abundance	3,22	2.505	0.086	
Total alien species abundance	3,22	0.929	0.443	
Proportion native abundance	3,22	1.064	0.384	
Diversity	3,22	4.882	0.009**	WC^{1} Bb^{1,2} LM^{1,2} Bh^{2}
Evenness	3,22	5.744	0.005**	WC^{1} LM^{2} Bh^{2} Bb^{2}

In summary, although significant differences in assemblage characteristics between sub-catchments were detected by ANOVA and ANCOVA, they were typically small. For example, although significantly fewer alien species were recorded from sites in Woopen Creek, the mean number of alien species in all sub-catchments was less than 1. Similarly alien species contributed less than 5% of the total number of fish collected from each site and, in most cases, only one or two individuals comprised this component. Nonetheless, it is clear that alien abundance is comparatively high in the most downstream site of Babinda Creek where it constituted 17% of the total number of fish collected.

5.3.6 Sub-catchment differences in fish assemblage structure

The various sites examined in this study clearly contained different assemblages of species and to some extent the various sub-catchments differed in the types and abundances of species collected. For example, no overlap in the ordination space defined by axes 1 and 3 (collectively accounting for 74.1% of the observed variance) was discernible between sites within Babinda Creek and the remaining sub-catchments (Fig. 5.11).

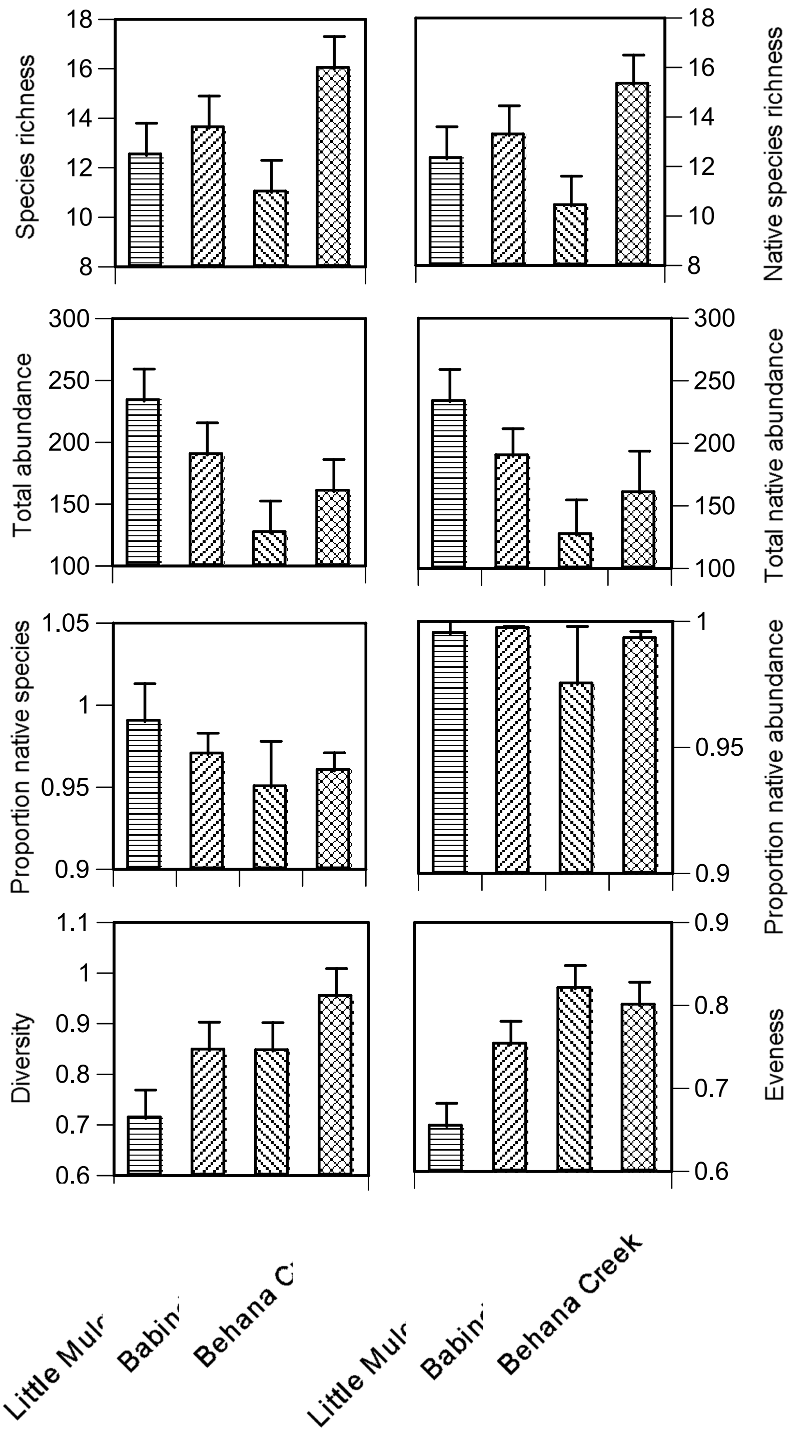

Figure 5.9. Average values (± SE) for fish assemblage characteristics for each of the four subcatchments.

Table 5.7. F values and associated statistics for GLM analysis of covariance of those fish assemblage characteristics in which distance to river mouth was a significant covariate. Mean (and SE) values for each subcatchment are given where ANCOVA revealed significant differences. Significantly different (p<0.05) values are denoted by different superscript numbers. An * denotes that a significant interaction between the main effects (subcatchment and Distm) precluded examination of difference in mean values between subcatchments. . LM = Little Mulgrave River, WC = Woopen Creek, Bh = Behana Creek and Bb = Babinda Creek.

Parameter	Source of variation	d.f.	F	p	Partial eta squared	LM	WC	Bh	Bb
Alien species richness	Log DISTM Subcatchment	1 3	39.36 8.739	0.000 0.000	0.652 0.555	0.38^{1} (0.18)	0.17^{2} (0.17)	0.71^{1} (0.18)	0.60^{1} (0.40)
Alien species abundance	Log DISTM Subcatchment interaction	1 3 3	6.801 3.929 3.740	0.018 0.026 0.034	0.274 0.396 0.030	0.56 (0.33)	1.00 (1.00)	1.25 (0.53)	4.53^{*} (4.3)
Proportion native species	Log DISTM Subcatchment	1 3	34.65 8.445	0.000 0.001	0.623 0.547	0.97^{1} (0.01)	0.99^{2} (0.01)	0.96^{1} (0.01)	0.95^{1} (0.03)
Proportion native abundance	Log DISTM Subcatchment interaction	1 3 3	4.614 7.058 4.403	0.015 0.016 0.017	0.435 0.282 0.423	0.997 (0.000)	0.995 (0.005)	0.993 (0.003)	0.975^{*} (0.023)
Diversity	Log DISTM Subcatchment	1 3	7.155 4.018	0.014 0.021	0.254 0.365	$0.85^{1,2}$ (0.03)	0.71^{1} (0.05)	0.95^{2} (0.05)	$0.84^{1,2}$ (0.03)

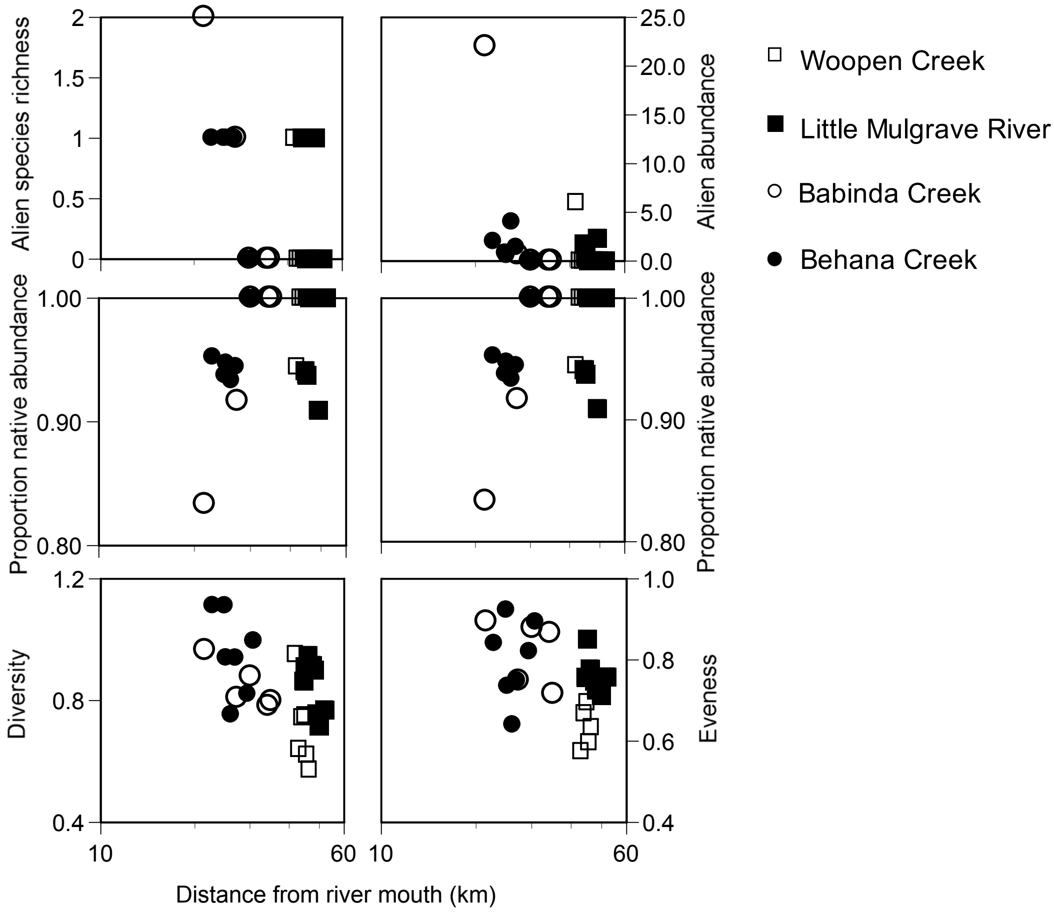

Figure 5.10. Spatial variation in fish assemblage characteristics along a gradient of distance of the study sites from the river mouth.

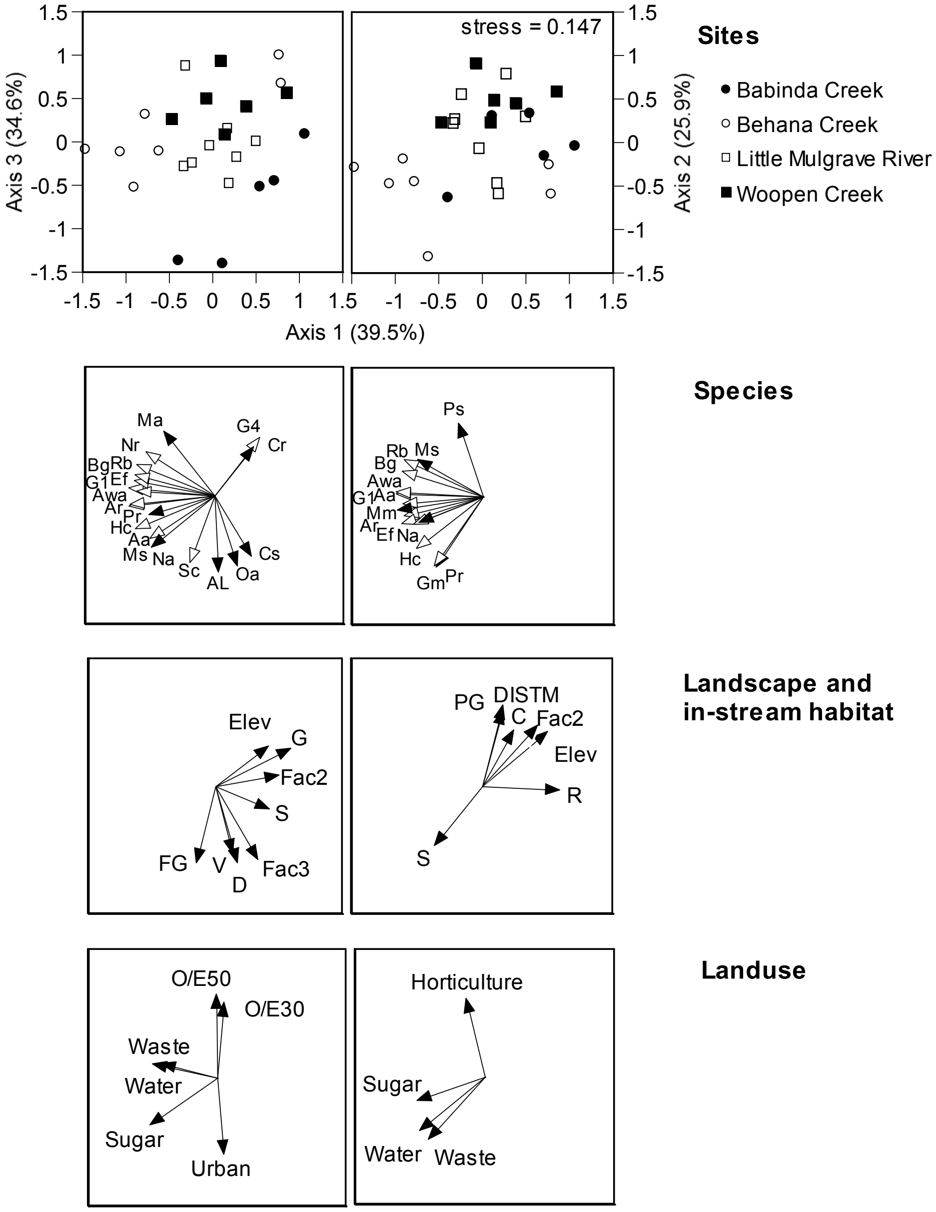

Figure 5.11. Spatial variation in fish assemblage structure based on SSHMDS ordination of sites by species abundance (log transformed). The proportion of the total variance accounted for by each axis is given as a percentage. Significant correlations ($p<0.05$) between species abundance and site position within the defined ordination space are displayed in the species plot. Estuarine or marine dependent species are denoted by the open arrowhead. Species abbreviations are based on the first letters of the generic and species epithets. Significant correlations between landscape-scale and local scale habitat descriptors and site position in ordination space are displayed in the landscape and in-stream habitat plot. Abbreviations are: Elev – elevation; DISTM – distance to the river mouth; Fac2 and 3 – factor scores for Principal Components 2 and 3, respectively; V – velocity; D – depth; S – sand; FG – fine gravel; G – gravel; C – cobbles; R – rocks, and PG – para grass. Significant correlations between land-use descriptors and site position in ordination space are given in land-use plot. Also shown in this plot are significant correlations between site position and O/E_{50} and O/E_{30} scores (see below).

To some extent, sites within the Little Mulgrave River formed a distinct group overlapping only marginally with the group formed by Behana and Woopen Creeks. Distinct grouping was less evident in the space defined by axes 1 and 2 (65.4%), where most sites formed a single large group in the upper right region of the ordination plot. Downstream sites on Behana Creek were distinct from this group.

Sites located positively on all three axes (i.e., the upper right region of both plots) were located at high elevations distant from the river mouth and possessed a coarse substrata component (i.e., habitat variables heavily loaded in PC2 in Figure 5.6). Such sites were characterised by *Glossogobius* sp. 4, *P. signifer* and *C. rhombosomoides*. Downstream sites contained a complex mixture of species dependent on estuarine or marine habitats for larval production, such as *E. fusca*, *G. margaritacea, B. gyrinoides, A. reinhardtii, B. gyrinoides, H. compressa, Glossogobius* sp. 1, *A. acritosus* and some exclusively freshwater species typical of downstream reaches of abundant cover, increased depth and reduced water velocity, such as *M. splendida*, *N. ater* and *P. rendahli*. These estuarine and marine dependent species, with the exception of the almost ubiquitous eel *A. reinhardtii*, were largely absent from both downstream and upstream sites within Babinda Creek, to the extent that lowland sites within this drainage contained assemblages more closely resembling those found in the upper reaches of the Little Mulgrave River and Behana and Woopen Creeks.

It is notable that alien species (pooled across all three species) were strongly negatively correlated with site scores on axis 3 and the two most downstream sites on Babinda Creek contained relatively high abundances of alien species. Also notable is the significant correlation between para grass abundance (Braun Blanquet scale – BB) and site position in ordination space. The effect is primarily driven by the very high abundance of this alien grass on the stream banks of all sites within Woopen Creek and to a lesser extent Babinda Creek. For example, mean BB scores for these two drainages were 3.5 ± 0.34 (SE) and 2.2 ± 0.37, respectively. Note that the maximum possible score is 4. While para grass was present in the Little Mulgrave River and Behana Creek, it was limited to a small number of sites and rarely dominated the stream banks. Mean BB scores for these drainages were 0.25 ± 0.16 and 0.43 ± 0.30, respectively.

Site position in ordination space was also correlated with five aspects of land use. The proportion of catchment devoted to sugar production, water resource use and waste treatment were all negatively correlated with axes 1, 2 and 3. All three land uses were negatively correlated with elevation and distance from the river mouth and loaded negatively on PC2 in Figure 5.7 and Table 5.5. The detection of significant correlations with fish assemblage composition more likely represents co-variation as a result of the effect of site position in the landscape and its influence on fish assemblage structure associated with location of spawning and rearing grounds.

As noted above, assemblages present in the most downstream sites within Babinda Creek differed from those of Behana Creek, which, with respect to distance from the river mouth and elevation, they most closely resemble. Downstream sites were also observed to contain assemblages substantially different from those expected (see below). For these reasons, we compared assemblage characteristics and species abundances in the three most downstream sites of both drainages by t-test of the respective means. Species richness and native species richness were both significantly lower in Babinda Creek (Fig. 5.12).

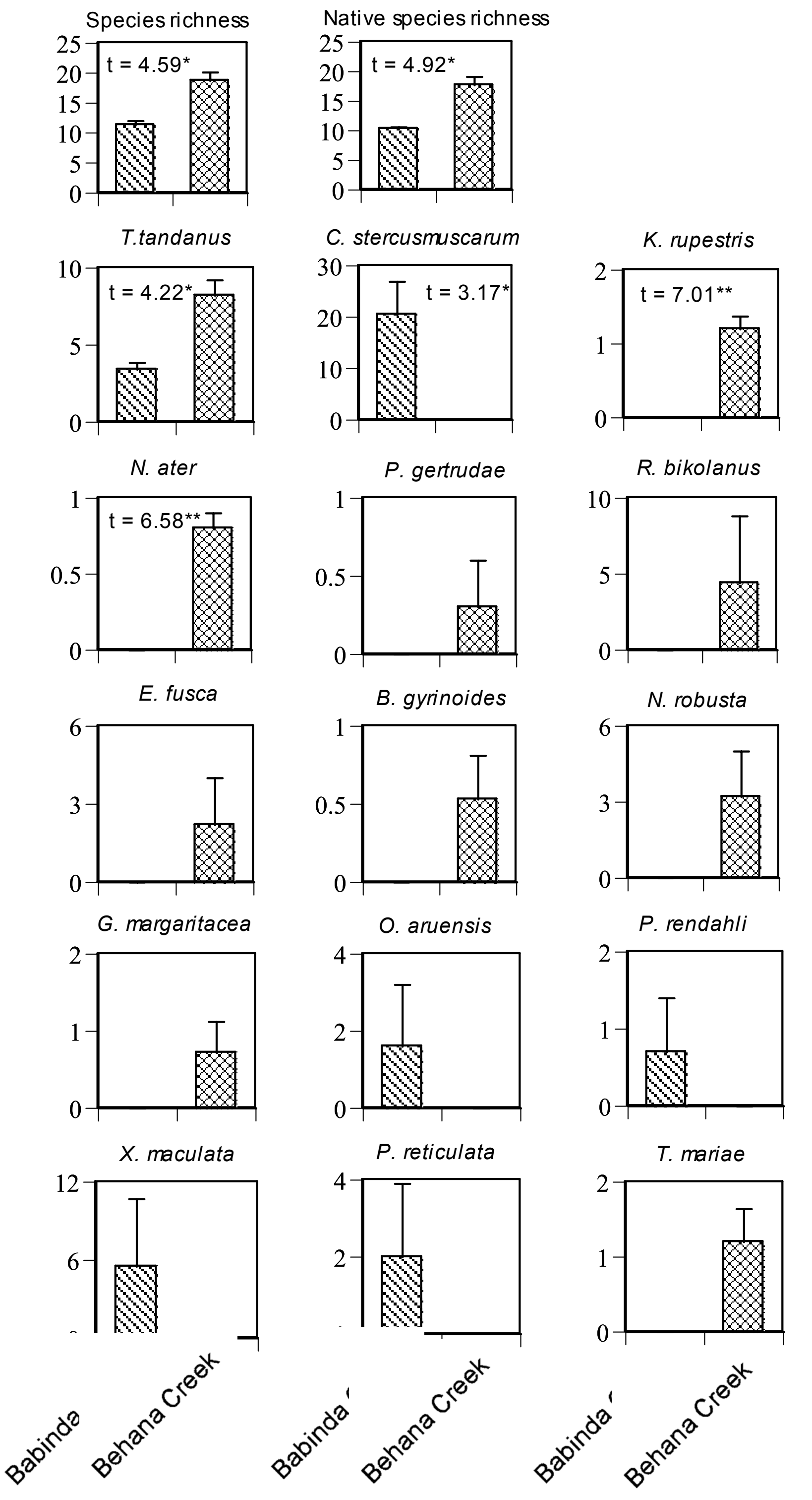

Figure 5.12. Comparison of species richness and abundance of selected species in the three most downstream sites of Babinda and Behana Creek. Species selected for display differed significantly in abundance or were absent from one drainage. The results of t-tests are given where significant.

Four species – *T. tandanus*, *Tandanus* sp., *N. ater* and *K. rupestris* were significantly more abundant in Behana Creek (Fig. 5.12). Both catfish species are reliant on bank-associated woody debris and undercuts (Pusey *et al.* 2004b) and their reduced abundance or absence may reflect the lack of such habitat in Babinda Creek owing to high abundances of alien weeds (e.g., para grass). The jungle perch *K. rupestris* also relies on woody debris extensively, but more importantly migrates between fresh and saltwater habitats to breed (Pusey *et al.* 2004b). *Craterocephalus stercusmuscarum* was more abundant in Babinda Creek. This species can be classified as an open-water run-dwelling species and the changes in habitat associated with para grass proliferation appear to favour this species, as in streams of south-east Queensland (M. Kennard, pers. obs.).

Although significant between-drainage differences in abundance were not detected for the remaining species shown in Figure 5.12, probably because of high variances and lack of statistical power resulting from small sample sizes (n = 3 for both sub-catchments), these species were present in the reaches of Behana Creek encompassed by the range of sites compared, but were not present in Babinda Creek. Of the species recorded from Behana Creek but not Babinda Creek, all except one (*P. gertrudae*) have an estuarine/marine life history phase and the juveniles must migrate upstream to suitable habitat (Pusey *et al.* 2004b). In contrast, *P. gertrudae* is exclusively freshwater in habit but is typically associated with very low water velocities (Pusey *et al.* 2004b). The elevated flows and prolific weed growth in the lower reaches of Babinda Creek may not provide suitable habitat for this species and may prevent or interfere with the upstream migration of small juveniles of the remaining species. *Oxyeleotris aruensis* and *P. rendahli*, in contrast, are very frequently associated with complex habitat such as that provided by para grass infestations.

Of the alien species recorded, neither *X. maculatus* or *P. reticulata* make use of swiftly flowing habitats but will extensively use the marginal slackwater habitats created by para grass in otherwise swiftly flowing habitats. The alien *T. mariae* also prefers more slowly flowing habitats and was only recorded from Behana Creek (Fig. 5.12).

5.3.7 *Prediction of species richness in reference and test sites*

The multiple regression model developed to predict species richness at individual locations based on their position in the riverine landscape (according to site elevation and distance from the river mouth) accounted for over 50% of the observed variation in species richness at the reference sites. Importantly, predicted species richness varied with observed richness at a rate of very close to 1:1 (0.92). The model had a slight tendency to under-predict species richness at sites of high richness (Fig. 5.13). Nonetheless, the mean O/P value across the entire reference set was 0.97, indicating a generally close match between predicted and observed scores (Fig. 5.13). The distribution of reference O/P scores (Fig. 5.14) was positively skewed and, indeed, the median value of 0.88 was somewhat smaller than the mean. Eighty percent of all reference O/P scores fell between 0.62 and 1.32 (10^{th} %ile and 90^{th} %ile, respectively).

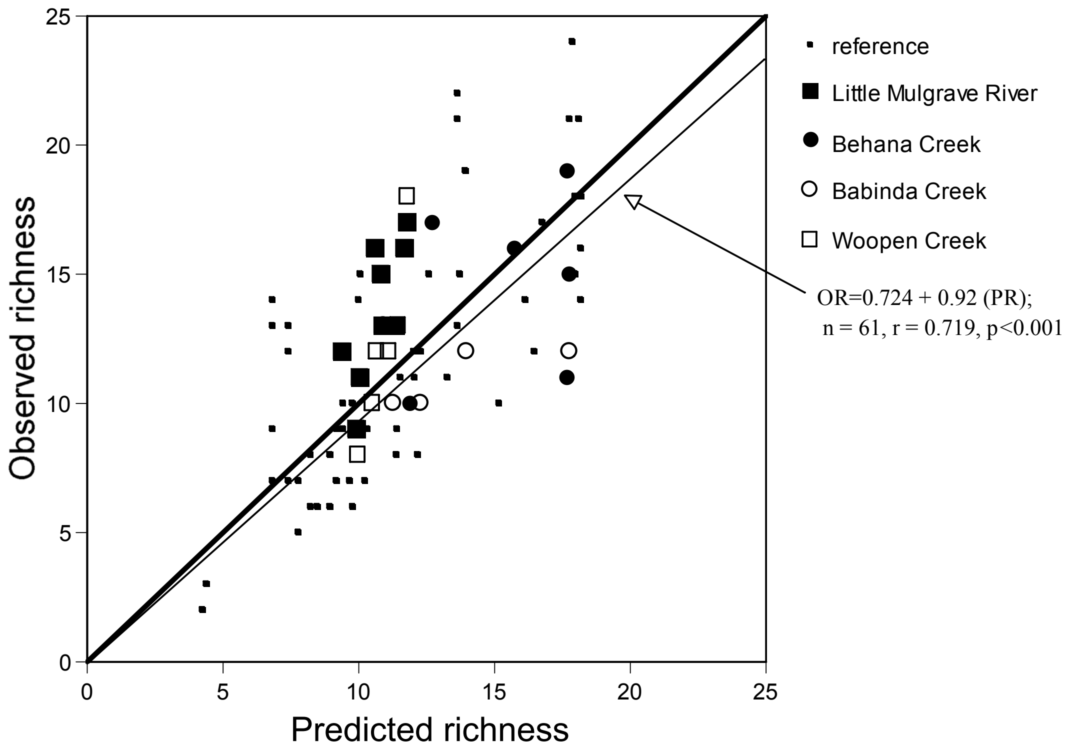

Figure 5.13. Observed species richness compared to predicted species richness for reference and test sites. The heavy diagonal line represents perfect agreement between observed and predicted richness (1:1) whereas the fine line is the regression equation, given immediately below the symbol legend, representing the relationship between observed richness and that predicted by the model.

The distribution of O/P values for each of the test sub-catchments is also shown in Figure 5.14. None of the sites within the test drainages contained significantly fewer species than expected (i.e. less than the 10^{th} %ile) but one site in each of Behana and Woopen Creeks and four sites in the Little Mulgrave River contained significantly more species than expected. Overall, there was a tendency for O/P values to increase with increasing distance upstream ($r = 0.58$, $p<0.01$) (Fig. 5.15); however, this relationship was not evident for all sub-catchments when examined separately. No significant relationship was detected for Behana Creek or the Little Mulgrave River. O/P values declined significantly with distance downstream in Babinda Creek becoming increasingly less than predicted. In contrast, species richness increased with increasing distance downstream at rates greater than predicted in Woopen Creek (Fig. 5.15).

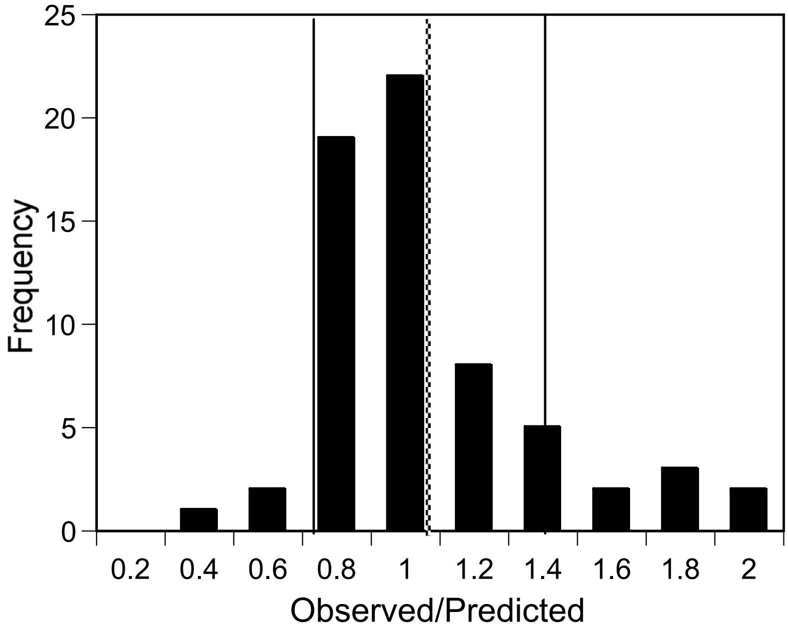

Figure 5.14. Frequency distribution of O/P scores for reference sites. The heavy broken line indicates the mean score for reference sites and the light broken lines represent the 10th and 90th percentiles. Note that the x-axis is categorical (i.e. different O/P classes with a width of 0.2) for the distribution of test and reference scores but is continuously varying for mean, 10th and 90th values.

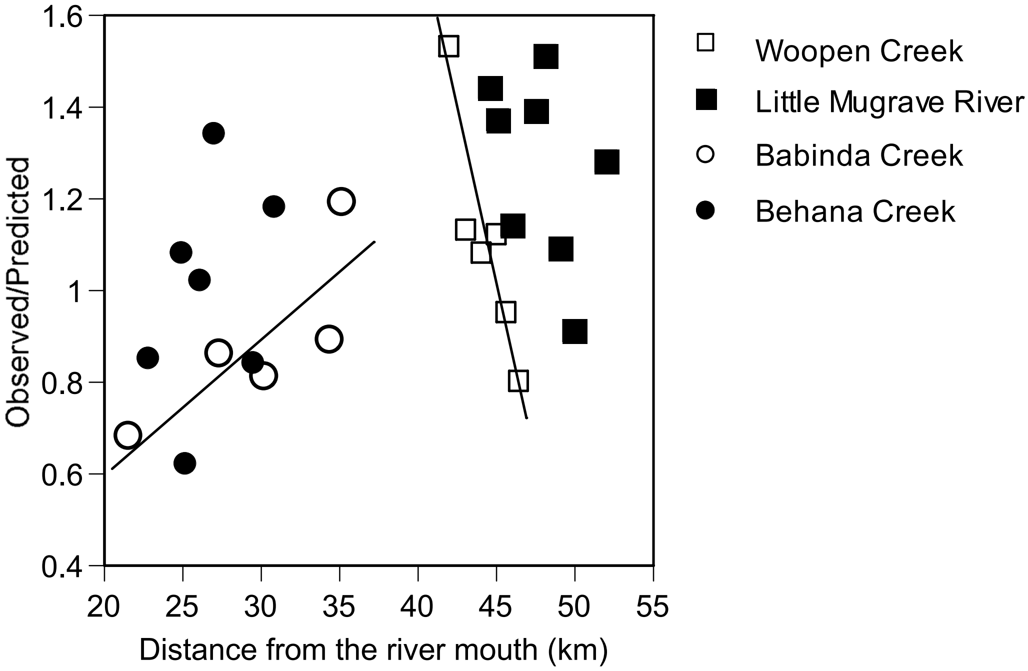

Figure 5.15. The relationship between O/P scores and distance from the river mouth for each of the test streams. Solid lines represent significant ($p<0.05$) relationships determined by simple linear regression.

5.3.8 Prediction of fish assemblage structure

In order to derive the expected composition for each site within our test drainages (Woopen, Babinda and Behana Creek and Little Mulgrave River), test sites were allocated to reference groups based on their position within the riverine landscape as defined by elevation and distance from the river mouth (Fig. 5.16). The majority of test sites were allocated to either groups C, D or E. However, five sites (four in Behana Creek and one in Babinda Creek) were not placed in any existing group. All were located at very low elevation (<3 m.a.s.l.) but more than 20 km distant from the river mouth. It was decided that these sites should be allocated to group A after comparison of species present with those expected in group A or C. The species comprising each of the reference groups are given in Table 5.8. As might be expected from the overlap of groups evident in Figure 5.16, many species are distributed across more than one group (Table 5.8), but the combinations of species, at both levels, are unique to individual site groups.

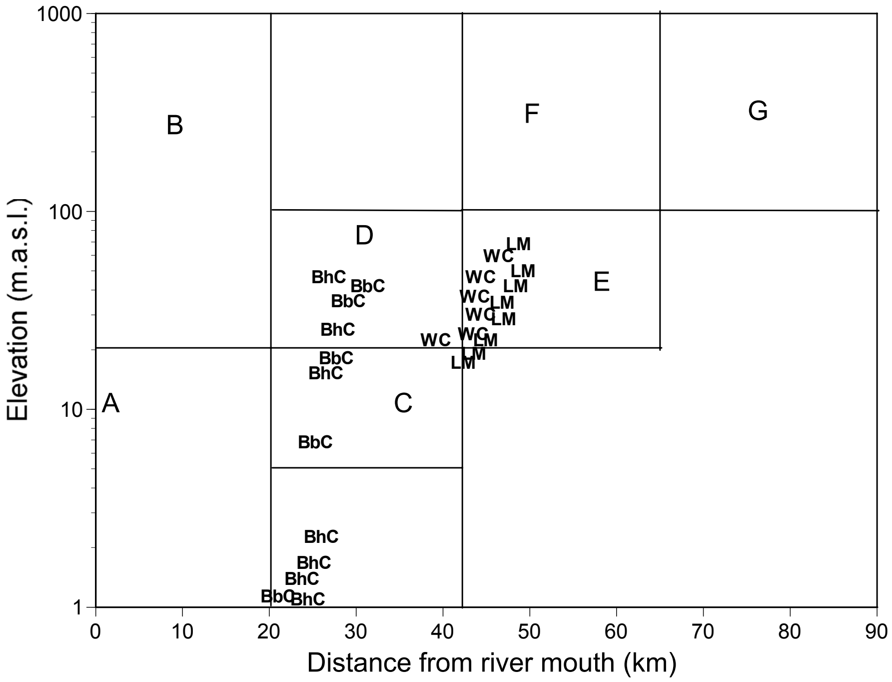

Figure 5.16. Allocation of test sites to reference groups based on elevation and distance from the river mouth. WC = Woopen Creek, LM = Little Mulgrave River, BhC = Behana Creek and BbC = Babinda Creek.

Table 5.8. Characteristics of reference groups. Species included represent those species recorded from 30% or more of sites within each reference group. Species present in 50% or more of sites within each group are denoted by bold type.

Group	A	B	C	D	E	F	G
N	12	5	11	7	8	14	3
Mean S	16.3 ± 1.1	8.8 ± 0.6	15.7 ± 1.3	10.3 ± 7.9	7.9 ± 1.0	8.6 ± 0.9	2.3 ± 0.3
Total S	46	21	38	22	18	21	3
S_{50}	14	6	15	9	8	6	2
S_{50}/mean S	0.86	0.68	0.96	0.87	1.01	0.7	0.87
S_{50}/Total S	0.30	0.23	0.39	0.41	0.44	0.29	0.67
	H. compressa	***P. signifer***	***M. splendida***	***P. signifer***	***P. signifer***	***P. signifer***	***M. adspersa***
	A. reinhardtii	***H. compressa***	***A. reinhardtii***	***T. tandanus***	***A. reinhardtii***	***A. reinhardtii***	***A. reinhardtii***
	M. splendida	***C. rhombosomoides***	***Glossogobius sp. 1***	***A. reinhardtii***	***Glossogobius sp. 4***	***Glossogobius sp. 4***	*C.stercusmuscarum*
	G. margaritacea	***O. aruensis***	***G. aprion***	***C. rhombosomoides***	***M. splendida***	***M. splendida***	
	G. aprion	***A. reinhardtii***	***A.acritosus***	***M. adspersa***	***H. tulliensis***	***H. tulliensis***	
	K. rupestris	**Glossogobius sp. 1**	***H. compressa***	***M. splendida***	***T. tandanus***	***Aw. acritosus***	
	Glossogobius sp. 1	*Mug. notospilus*	***K. rupestris***	***Aw. acritosus***	***C. stercusmuscarum***	*C. stercusmuscarum*	
	N. robusta	*Mo adspersa*	***T. tandanus***	***Glossogobius sp. 4***	***N. robusta***	*M. adspersa*	
	H. fuliginosus	*E. fusca*	***P. signifer***	***H. compressa***	*Aw. acritosus*	*T. tandanus*	
	P. signifer	*G. margaritacea*	***H. fuliginosus***	**Glossogobius sp. 1**	*Glossogobius sp. 1*	*C. rhombosomoides*	
	B. gyrinoides	*Oph. gutterale*	*Neo. ater*	*O. aruensis*	*K. rupestris*	*N. robusta*	
	N.ater	*Neo. ater*	***M. adspersa***	*K. rupestris*		*K. rupestris*	
	L. calcarifer	*L. calcarifer*	***B. gyrinoides***	*H. fuliginosus*			
	G. filamentosus	*T. chatareus*	***No. robusta***				
	C. rhombosomoides	*L. argentimaculatus*	***Oph. gutterale***				
	E. melanosoma		*C. stercusmuscarum*				
	Mes. argenteus		*G. margeritacea*				
	S. argus		*Am. agrammus*				
	H. tulliensis		*M. maccullochi*				
	T. tandanus		*R. bikolanus*				
	Aw. acritosus		*H. tulliensis*				
	E. fusca		*A. obscura*				
	L. argentimaculatus						
	Mu. cephalus						
	Meg. cyprinoides						

Figure 5.17 presents the results of an ordination based on the presence/absence of species of reference sites as defined by their *a priori* allocated group membership. Sites located in Group A (DISTM <20 km, E<20 m.a.s.l.) are located negatively on axis 2, whereas sites within group F (ELEV>100 m.a.s.l. and 43<DISTM<63 km) are located positively on this axis. Sites located at very high elevation (>500 m.a.s.l.) (group G) are located high on axis 1 as are, to a lesser extent, high elevation sites close to the river mouth (group B). These sites are differentiated from group G sites on axis 3. Species exhibiting significant correlations with site position in ordination space tended to fall into two groups. The first contained species highly negatively correlated with axis 2 scores and which, with the exception of *Hephaestus fuliginosus* (Hf), are all estuarine dependent species that do not penetrate far upstream (Pusey *et al.* 2004b). Species positively correlated with axis 2 scores and/or correlated with axis 3 scores comprise a mixture of species with estuarine larval phases that penetrate far upstream (i.e. *Glossogobius* sp 1 and sp 4) or entirely freshwater species that appear to be assorting along habitat gradients associated with both or either upstream distance and elevation.

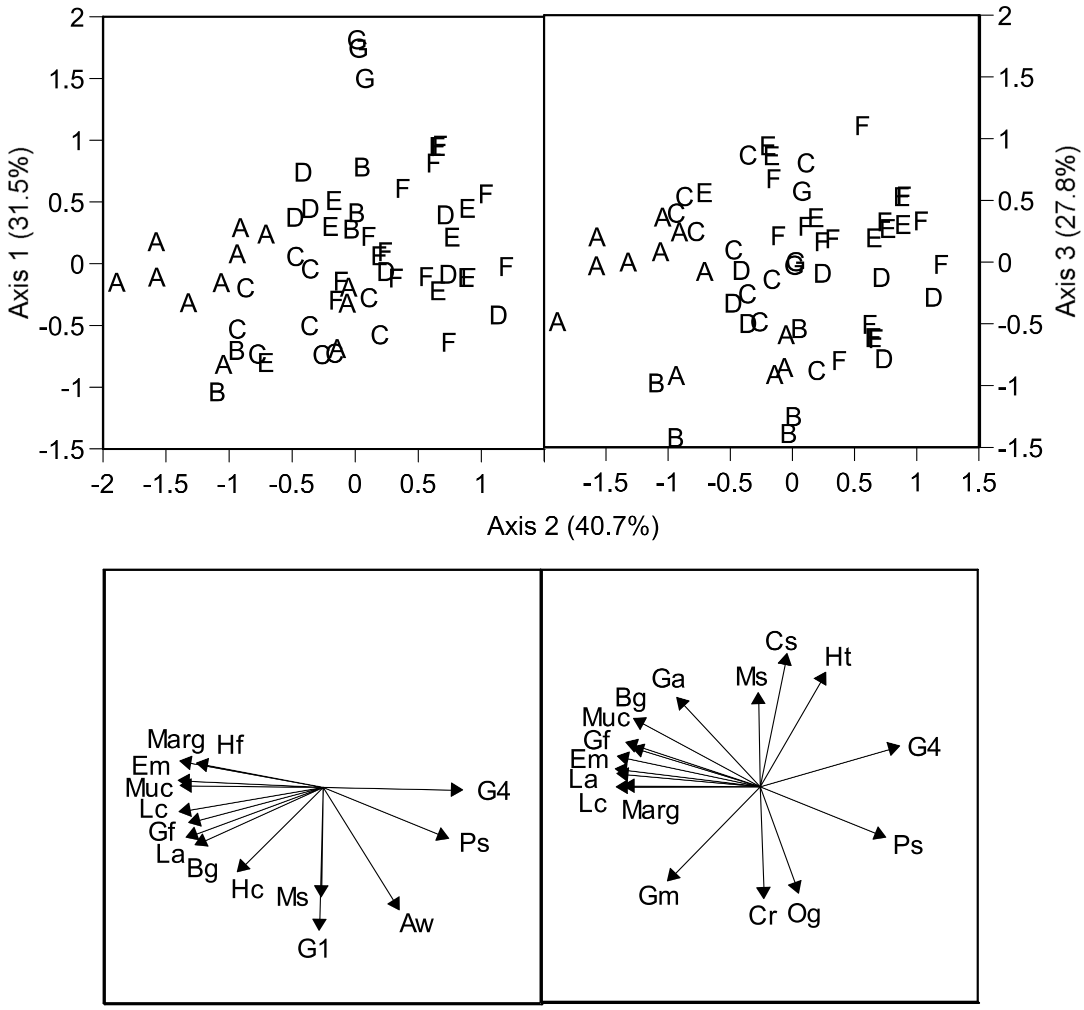

Figure 5.17. Spatial variation in fish assemblage structure within reference sites in the Russell/Mulgrave River. The upper plots represent the position of sites within site groups across ordination space defined by variation in assemblage species composition. Sites are coded according to the reference site group to which they belong (see Fig. 5.3). The lower plots represent significant correlations between species presence/absence and ordination space. Full species names are given in Table 5.2.

It is important to note that despite a general trend for site groups to be sequentially arrayed along axis 2, complete separation between groups did not occur and that groups tended to grade from one into another. This is perhaps not unsurprising given the distribution of sites across the riverine landscape depicted in Figure 5.17 and because we essentially forced an arbitrary group structure on what is continuously varying data. Moreover, species appear to be assorting along gradients in habitat associated with DISTM and ELEV and it is unreasonable to expect that this assortment is in accord with a "two-state" condition (either/or). Nonetheless, the array of sites depicted in Figure 5.17 suggests that our group structure has some biological basis.

The median O/E_{50} of the distribution shown in Figure 5.18 was 0.667, indicating that typically, two-thirds of expected species were observed within individual reference sites. In contrast, the median O/E_{30} was 0.545, indicating that almost half of the entire reference sample contained fewer than about half of the expected species. Clearly, substantial between-site variation in assemblage composition occurred within some groups and this is evident by the lack of well defined clustering of sites within groups in Figure 5.17. We compared mean O/E scores for each of the reference groups (Fig. 5.19) by ANOVA. Mean O/E_{50} and O/E_{30} scores did not differ significantly between groups ($F_{6,53}$ = 2.199 and 2.237, $0.05<p<0.1$, respectively) and therefore the level of deviation away from the expected assemblage composition was common to all groups.

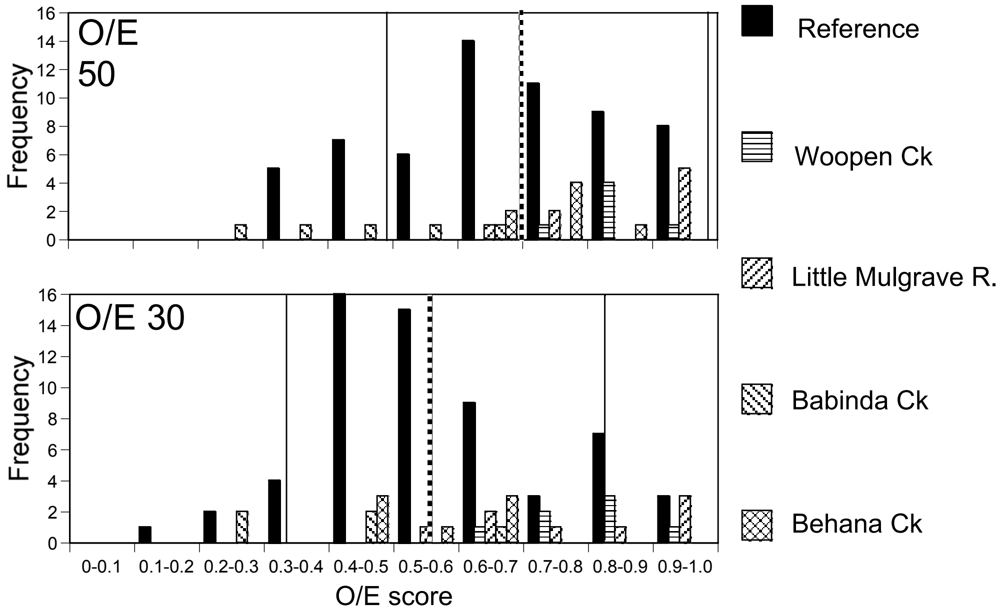

Figure 5.18. O/E scores for the reference sites and each of the test drainages. The heavy broken line and light broken lines represent the median, and the 10th and 90th %ile, respectively, for the reference group. Note that the x-axis is categorical (i.e. different O/E classes with a width of 0.2) for the distribution of test and reference scores but is continuously varying for mean, 10th and 90th %ile values.

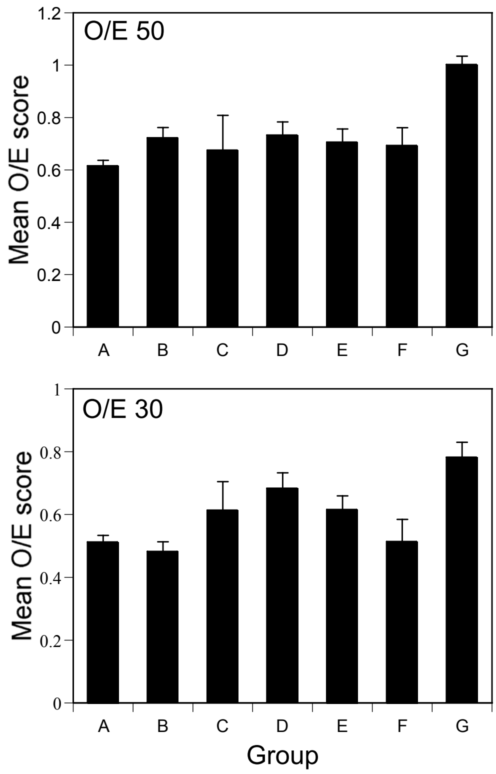

Figure 5.19. Mean O/E scores for each of the reference groups. Error bars are standard error. Sample sizes for each group are given in Table 7.8.

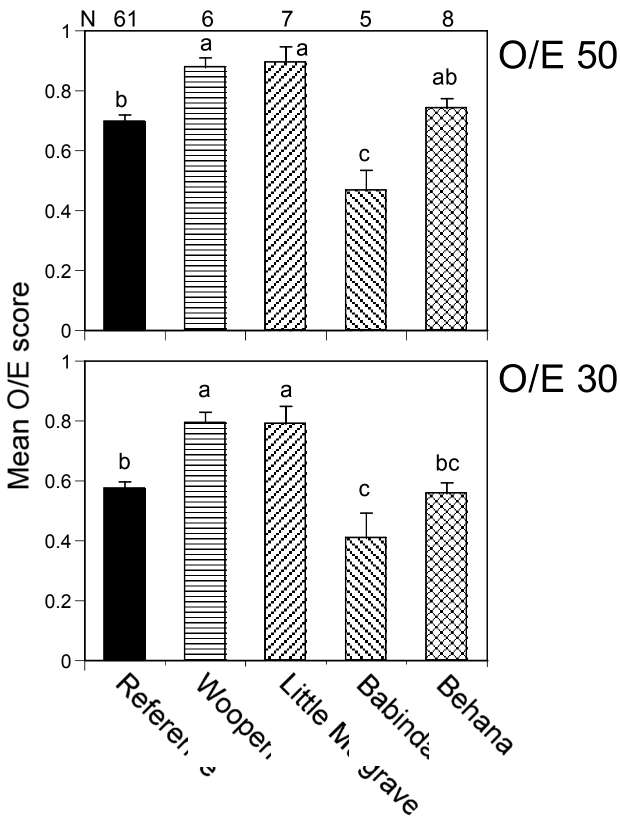

Figure 5.20. Mean O/E scores for the reference sites and each of the four test streams. Significant between group mean differences are indicated by lower case letters above each bar.

O/E scores were then computed for each test site. In general, O/E_{50} and O/E_{30} scores were greater in the test sites and drainages than in the reference sites (Fig. 5.20). Only three sites, all in Babinda Creek, fell outside of the 10th %ile of reference O/E_{50} values, indicating that these sites contained significantly different assemblages from that expected from our model. Similarly, when O/E_{30} criteria were used, two of these three sites were found to be significantly different from that expected because they lacked some species. Importantly, some sites within Woopen Creek, Behana Creek and Little Mulgrave River were found to contain more species than expected.

Analysis of variance revealed that significant between-drainage (reference and the four test streams) differences in mean O/E_{50} scores ($F_{4,85}$ = 6.835, p<0.001) and O/E_{30} scores ($F_{4,85}$ = 6.489, p<0.001) were detectable. Woopen Creek and the Little Mulgrave River had higher average O/E_{50} scores than the reference sites and Babinda Creek, while Babinda Creek had lower scores than Behana Creek and the reference sites (Fig. 5.20). A similar pattern was observed for O/E_{30} scores with the exception that scores for Babinda and Behana Creeks were not significantly different. Babinda Creek was notable by virtue of the significantly depressed O/E score irrespective of the criteria used (O/E_{50} or O/E_{30}).

Longitudinal changes in O/E scores were not apparent for either of the two small sub-catchments (Woopen Creek and Little Mulgrave River) but were prominent in Babinda and Behana Creeks (Fig. 5.21). In both, decreasing O/E scores and hence increasing deviation away from the expected assemblage composition occurred as one moved downstream; however, only the three most downstream sites were so different from expected that O/E_{50} scores were less than the 10th %ile. It could be argued that the low O/E_{50} scores arose because no appropriate reference group was available for the most downstream sites within these drainages and they therefore had to be compared with another, perhaps less appropriate, reference group (in this case group A). However, most sites for which this was the case (four of five) were from Behana Creek not Babinda Creek.

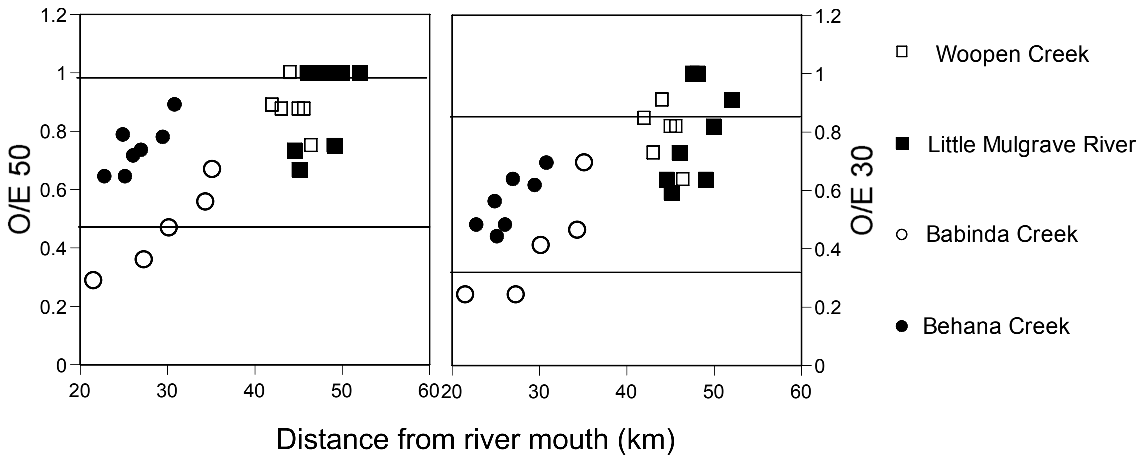

Figure 5.21. Longitudinal changes in O/E scores within each drainage. Broken lines represent the 10th and 90th %ile scores of the reference group.

5.3.9 Associations between O/E and O/P scores and environmental variables

O/E_{50} scores were negatively correlated (r = -0.390, p<0.05) with PC1 scores from the Principal Components Analysis of environmental and land use variables. This component described many aspects of riparian cover and condition, the presence of in-stream weeds and average water velocity. No significant associations between O/E_{30} and O/P scores and this component were detected (Table 5.9). In contrast, PC2 scores, which were associated with position in the catchment and associated changes in habitat structure such as substrata composition, were significantly correlated with O/E_{50}, O/E_{30} and O/P scores. That is, upstream sites contained fish assemblages more like that predicted than did downstream sites (but also note that assemblages within Woopen Creek became progressively more dissimilar to that predicted with distance upstream with respect to species richness). Finally, O/E_{50} and O/E_{30} were negatively correlated with PC3 scores (i.e. depth and macrophyte abundance). Deeper sites contained assemblages different from that predicted. The most downstream sites on Babinda Creek were both deep and swiftly flowing and were distinguished by low O/E and O/P scores (Figs. 5.5 and 5.22).

Table 5.9. Significant correlations between O/E and O/P scores with Principal Components derived from a PCA of environmental and land-use variables. Only those components having significant correlations with O/E and O/P scores are shown. * = p<0.05, ** = p<0.01.

Variable	O/E_{50}	O/E_{30}	O/P
PC1	-0.390*		
PC2	0.445*	0.575**	0.429*
PC3	-0.462*	-0.422*	

We attempted to decompose the relationship between O/E_{50} scores and PC1 by examining which variables loaded on this component were most strongly correlated with predicted scores. O/E_{50} scores were significantly positively correlated with riparian cover but no significant relationship was observed between this variable and O/E_{30} or O/P scores. However, as described previously, sites within Woopen Creek had anomalously high O/P and O/E scores (see Figs 5.13 and 5.14) irrespective of the extent of riparian cover. If these sites are excluded from the analysis, O/E_{50}, O/E_{30} and O/P values are all significantly correlated with riparian cover (r = 0.802, p<0.001; r = 0.690, p<0.001; and r = 0.451, p<0.05, respectively). No significant (p<0.05) associations between the abundance of emergent vegetation and predicted scores (O/E_{50}, O/E_{30} and O/P) were noted. Again, sites within Woopen Creek, despite being among the most heavily weed-infested, were distinguished by high scores (Fig 5.19). When Woopen Creek sites were excluded from the analysis, both O/E_{50} (r = 0.614, p<0.01) and O/E_{30} (r = 0.550, p<0.05) scores, but not O/P scores (r = 0.224, p>0.05), were significantly negatively correlated with the extent of bank-associated weed growth (Fig. 5.20), that is, as weed growth increased stream fish assemblages became less like those predicted.

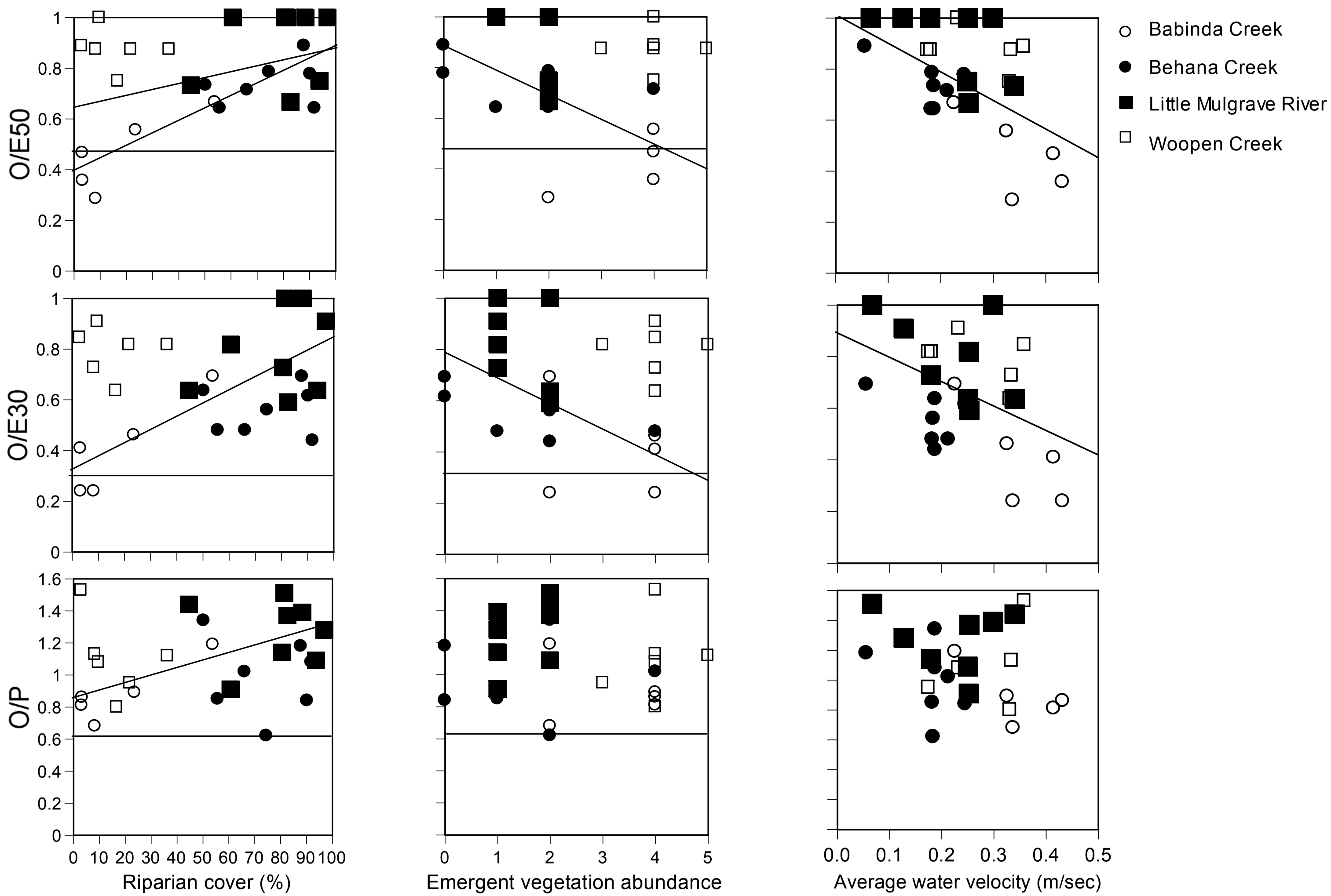

Figure 5.22. Relationships between O/E$_{50}$, O/E$_{30}$ and O/P scores with riparian cover, the abundance of emergent vegetation and average water velocity. The horizontal broken line indicates the 10th %ile for the reference set for each of the respective metrics. The unbroken oblique line represents significant relationships between respective test site scores and environmental variables and the oblique broken line represents signifiant relationships when sites within Woopen Creek were excluded.

We have previously demonstrated that weed infestation tends to increase average water velocities within the stream channel. Increasing water velocity was negatively correlated with O/E_{50} scores (r = -0.552, p<0.01) and O/E_{30} scores (r = -0.451, p<0.05) but not O/P scores (r = -0.239, p>0.05). When average water velocity, riparian cover and emergent vegetation abundance were used in a multiple regression analysis containing all sub-catchments (i.e. Woopen Creek was not excluded), only water velocity was identified as a significant predictor of O/E scores. Clearly, sites within Woopen Creek behave differently with respect to the influence of riparian cover, water velocity and emergent vegetation on O/E and O/P scores. This sub-catchment contained proportionally much greater numbers of the Pacific blue eye *P. signifer* than did Behana and Babinda Creeks and the Little Mulgrave River and we decided to examine relationships between *P. signifer* abundance and environmental characteristics. This species was more abundant in sites located distant from the river mouth (r = 0.472, p<0.05) although this relationship appears driven primarily because the Woopen Creek sites are distant from the mouth (Fig. 5.23). *Pseudomugil signifer* was also less abundant in sites with deep water and this relationship was significant when all sites were considered (r = 0.555, p<0.01). However, it is clear that sites within Babinda Creek contained few *P. signifer* irrespective of water depth. We therefore re-analysed the data with Babinda Creek sites excluded from the analysis. Depth was still a significant influence in this analysis (r = -0.575, p<0.01). *Pseudomugil signifer* was more abundant in sites with elevated mean water velocity but only when Babinda Creek sites were exlcuded from the analysis (r = 0.449, p<0.05). Increased abundance of *P. signifer* was observed in sites with increased para grass or total emergent vegetation when all sites were considered and when Babinda Creek sites were excluded (r = 0.474, 0.388, 0.645 and 0.565; p<0.05, 0.05, 0.01 and 0.01, respectively). Decreased canopy cover was negatively correlated with abundance but only when Babinda Creek sites were excluded.

We then undertook a multiple regression analysis of spatial variation in abundance of *P. signifer* using the variables displayed in Figure 5.21. Mean depth (-ve, r^2 = 0.308) and para grass (+ve, r^2 = 0.204) collectively accounted for over 50% of the variance in *P. signifer* abundance (F = 12.06, p<0.001).

5.3.10 Alien species

Alien species were neither common nor abundant in the four sub-catchments of the Russell/Mulgrave River, with only three species (*X. maculatus, P. reticulata* and *T. mariae*) being recorded and collectively accounted for 0.8% of the total number of fish collected. However, alien contribution to total abundance was as high as 12% in one site (lowermost site in Babinda Creek) and overall, when alien richness was up to three species, abundance was elevated. Alien species communities were richest at low elevation, close to the river mouth and in areas with a high proportion of the catchments devoted to sugar cane production and urbanisation. Similarly, alien species abundance was correlated with these variables (Table 5.10). The results parallel those patterns noted for alien species richness in the ordination analysis above (Fig. 5.10). Alien species contributed most to total fish abundance at deep sites, which were mostly limited to Babinda Creek.

It is noteworthy that the presence and abundance of alien species were negatively associated with both O/E scores although these species were not included in the derivation of these scores. As fish communities became even more dissimilar to those expected, and as a corollary, more indicative of environmental degradation, the number and abundance of alien species increased.

Table 5.10. Significant correlations between alien fish species abundance and richness and habitat characteristics and measures of fish assemblage health (O/E scores). * = p<0.05, ** = p<0.01.

	Alien species richness	Total alien abundance	Proportion of total abundance
Alien species richness			
Total alien abundance	0.696**		
Elevation	-0.680**	-0.390*	
Distm	-0.529**		
O/E$_{50}$	-0.594**	-0.495*	
O/E$_{30}$	-0.610**	-0.424*	
Depth			0.506**
Sugar (proportion of catchment)	0.656**	0.514**	
Urban (proportion of catchment)	0.537**	0.940**	

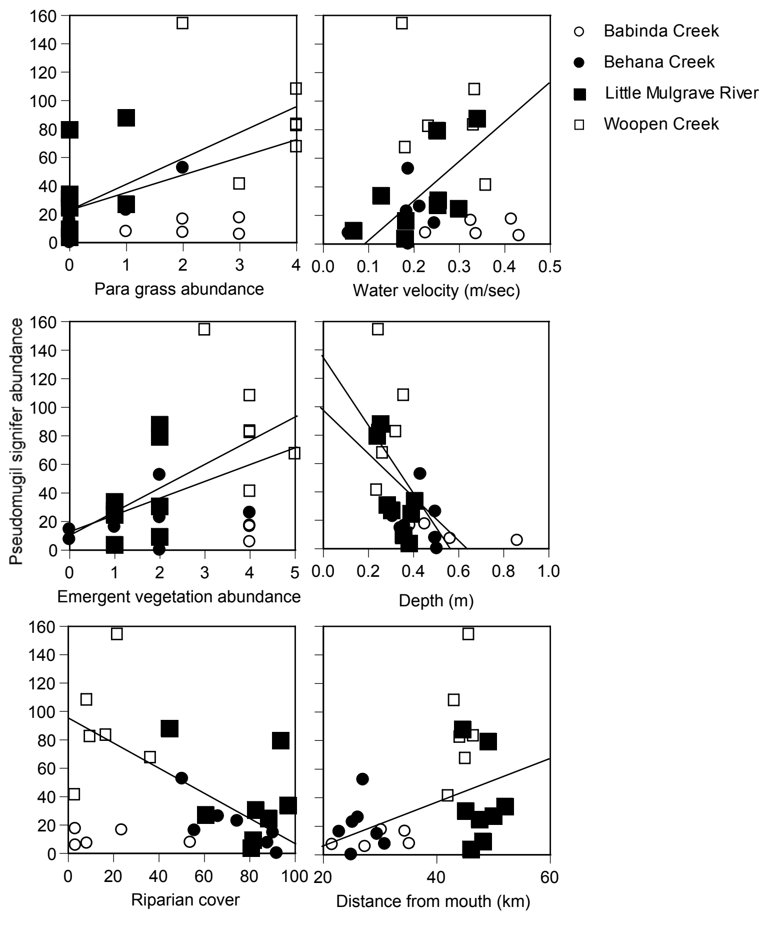

Figure 5.23. Relationships between the abundance of *Pseudomugil signifer* and environmental characteristics of the sites. Solid lines represent significant relationships when all subcatchments were considered and broken lines represent significant relationships when Babinda Creek sites were excluded.

5.4 Discussion

5.4.1 The fish fauna of the study streams

The freshwater fish fauna of the wet tropics region of north-eastern Australia is highly diverse and distinctive in several respects (Pusey and Kennard 1996; Russell *et al.* 1996,b; Unmack 2001; Pusey *et al.* 2004b). A total of 103 native and four alien species has been recorded from fresh waters (<2000 $\mu S.cm^{-1}$) of the region, representing 37 families with almost half (52/107) within six families. During this study of four streams in the Russell/Mulgrave catchments we recorded 30 species (27 native, 3 alien) within 15 families. Fourteen of the native species collected have life histories completed entirely within fresh water, with the remaining 13 species requiring access to estuarine or marine areas for spawning or larval development. The proportion of the total number of species collected that were exclusively freshwater was greatest in the two most upstream sub-catchments: the Little Mulgrave River (60.2%) and Woopen Creek (58.8%); and lowest in the two most downstream sub-catchments: Behana Creek (54.2%) and Babinda Creek (52.6%) closer to tidal influences.

Self-sustaining populations of four alien species – *Oreochromis mossambicus* (Cichlidae), *Tilapia mariae* (Cichlidae), *Poecilia reticulata* (Poeciliidae) and *Xiphophorus maculatus* (Poeciliidae) – occur in the Russell/Mulgrave basin. All but *T. mariae* were recorded in our study streams.

The diversity of the fish fauna in our study area (30 species), its distinctive elements and the low number of alien species suggest that these lowland streams still retain high ecological values, such that protection of stream ecosystems from threatening processes should be a high priority for management in this region.

5.4.2 Patterns of freshwater fish assemblage composition and structure

Major changes in assemblage structure associated with spatial variation in habitat structure and position within the riverine landscape were noted in the present study, paralleling those reported elsewhere for this drainage basin (Pusey *et al.* 1995a; Pusey *et al.* 2000). The overall pattern is one of increasing total species richness with increasing sub-catchment size, decreasing species richness with increasing distance upstream from the river mouth, decreasing proportional representation of species with an estuarine or marine dependence with increasing distance upstream from the river mouth and longitudinal changes in assemblage composition. Between-sub-catchment comparisons of fish assemblage attributes and composition must therefore be mindful of underlying differences with respect to catchment size and position within the landscape, and how these factors might affect the validity of comparisons. For example, Woopen Creek and the Little Mulgrave River contained fewer species than Behana Creek, but this result is not unexpected given that Behana Creek has a larger catchment area and its confluence with the main channel is located closer to the river mouth than are those of Woopen Creek and the Little Mulgrave River. It is especially noteworthy, however, that these two small sub-catchments contained more species than did Babinda Creek. Babinda Creek is not only larger than Woopen Creek and the Little Mulgrave River, but much more of its catchment lies at low elevation close to the river mouth. Accordingly it should contain more species. In addition, the composition of fauna in this sub-catchment should have been be similar to that in Behana Creek, which it closely resembles in terms of size and position within the riverine landscape, but it was not. Furthermore, Behana Creek contained significantly more species at the site level than was observed in Babinda Creek. Babinda and Behana Creeks also differed with respect to the abundance and presence of many species of fish that have an estuarine or marine larval interval and that must migrate upstream as juveniles. Species such as *K. rupestris*, *B.*

gyrinoides, E. fusca, G. margaritacea and *R. bikolanus* were either absent from or occurred in lower abundances in Babinda Creek compared with Behana Creek. Also absent from the lower reaches of Babinda Creek were adult and juvenile *Tandanus* sp. and *N. ater.* These plotosid species are frequently associated with bank-related habitat features (Pusey *et al.* 2004b) and are probably absent from the lower Babinda Creek because such features are either smothered by invasive weeds or are no longer present because of denudation of the riparian zone.

Differences in species richness and assemblage composition suggest that Babinda Creek contains fish assemblages substantially different from those occurring in the remaining three sub-catchments. In addition, the average number of fish collected per site was significantly lower in Babinda Creek than elsewhere ($p<0.10$). Note that significant differences were limited to this large alpha value (rather than $p<0.05$) because abundance levels were comparatively high in the most upstream sites within this drainage and depression of abundance as well as species richness was mostly limited to the most downstream sites within this drainage. It is also apparent from the ordination analysis that the most downstream sites on Babinda Creek contained the most differentiated fish assemblages. For example, the two most downstream sites separate from all other sites in ordination space. Similarly, it was the three most downstream sites in Babinda Creek that were identified as being different from expected in terms of O/E_{50} scores. Clearly, the downstream sites on Babinda Creek were different from expected.

5.4.3 Natural patterns in spatial variation in stream habitat and patterns of human disturbance

The physical nature of stream habitat is governed primarily by the interactions of stream flow, gravity and channel gradient. The typical pattern, and the one observed within the Russell/Mulgrave drainage system, is for headwater streams to be small, high gradient, shallow, and with a substratum dominated by coarse particles. Conversely, larger streams located in the lowlands are wide, deep, of low gradient and characterised by fine particles. As noted above, such predictable changes are important in determining the number and type of freshwater fish species found in particular locations within a catchment (Pusey *et al.* 2000). In the present case, there is a strong spatial relationship between position in the catchment and trends in the amount of the catchment devoted to urban and agricultural activities. Thus, any changes in fish assemblage composition and structure that are associated with human land use are potentially confounded because such gradients in land use are also correlated with a variety of variables important in determining fish assemblage composition. It is important to note that, in general, the amount of catchment area devoted to agriculture and urban development is small, as over 80% of the entire catchments of Babinda, Behana and Woopen Creeks, and the Little Mulgrave River are still covered by relatively intact native forest; however, most of the intact vegetation is restricted to the upper catchment, which was not investigated in this study.

One feature of the stream environment that may be subject to human-induced degradation but for which the severity of change from the natural state was not related to position in the catchment, is the riparian zone (see results of Principal Components Analysis). Damage to the riparian zone that resulted in reductions in the extent of canopy cover over the stream channel gave rise to a proliferation of riparian weeds such as para grass and Singapore daisy, both of which are immersion tolerant and can therefore grow out from the stream bank into the wetted channel (Chapter 3). This study clearly showed that in such cases, this proliferation of weeds had confined most of the stream flow to a narrow channel characterised by elevated water velocities. In Woopen Creek, elevated flows were correlated with the removal of fine sediment such that the sediment was dominated by cobbles and gravel; however, there was no stream incision as the remaining particles were too large to transport. In contrast, where weeds were very prolific in Babinda Creek, elevated stream flows appeared to be associated with stream incision owing to the fine nature of the substratum, and stream depth was greater overall in the lower reaches of Babinda Creek than in equivalent sections of Behana Creek. The overall change in habitat structure was one of increasing depth and water velocity and decreasing diversity in terms of both.

It is notable that riparian cover was the strongest correlate of stream water velocity and when combined with gradient, riparian cover explained 60% of the observed spatial variation in this parameter. Isolation of the stream bank and any associated structures such as undercut banks, root masses and woody debris was common to both Woopen Creek and Babinda Creek. Also common to both creeks was the creation of large areas of stationary water confined within the weed beds. Such habitats tend to be of poor water quality, principally because of depressed dissolved oxygen levels, and are unfavourable habitats for all but a few fish species, many of which are alien (Arthington *et al.*1983; Pusey and Arthington 2003; Kennard *et al.* 2005). Increased aquatic weed growth was correlated with increased abundance of alien fish in the present study. Native species that appear to be favoured by this novel habitat include some wetland species such as the catfish, *Porochilus rendahli* and the hardyhead, *Craterocephalus stercusmuscarum* (Pusey *et al.* 2004b; Kennard *et al.* 2005; 2006b).

5.4.4 Predictive models of species richness and assemblage contribution

Models such as that used here for predicting species richness and assemblage composition at locations in the absence of human disturbance are constrained by a number of factors. First, the suite of reference sites upon which the models are based should ideally be entirely free of any disturbance and represent the pristine state. This is unlikely to have been the case in the present study for a number of reasons. The Russell/Mulgrave catchment is small (<3000km^2) and there are few streams downstream of the World Heritage Area that are not disturbed in some manner (see Russell *et al.* 1996), particularly those that are analogous to the two largest sub-catchments in the present study. We attempted to minimise this effect by excluding sites that were grossly disturbed but, nonetheless, the suite of reference sites used encompassed a range of disturbance levels.

Second, reference sites should ideally be sampled in the same manner as test sites. This was not the case here as information on reference sites was drawn from four separate studies employing dissimilar sampling protocols despite the main sampling method being electrofishing. Moreover, the spatial scale of examination varied among studies. Thus, data used to characterise reference sites was derived from single or double pass electrofishing over more than 100 m of stream (i.e. encompassing multiple different hydraulic units) (e.g., Pusey and Kennard 1996; Russell *et al.* 1996) as well as multiple pass electrofishing plus supplementary seine netting in single hydraulic units (e.g., Pusey *et al.* 1998, 2000). Such differences are very important with respect to estimating species richness and assemblage composition (Pusey *et al.* 1998; Kennard *et al.* 2006c).

Third, reference sites should ideally be sampled at roughly the same time of year as test sites to minimise the extent to which differences in antecedent conditions might influence assemblage composition (Kennard *et al.* 2006a,b). Reference sites used in the present study were all sampled during the 1990s and in some cases (e.g., Pusey *et al.* 1995) a period of 13 years separated the sampling of reference and test sites. For example, wet season flows in 2002 and 2003, prior to the study, were very much lower than the long-term average, whereas March 2004 flows were twice that of the long term average for this month (T. Rayner, *pers. comm.*). The extent to which these events may have influenced the distribution of fish species is unknown; however, we noted at the time of sampling (July 2005) that several species (e.g., *G. margaritacea, E. fusca* and *H. compressa*) were collected at locations much further upstream than ever previously recorded. Such temporal variation, although unusual for the wet tropics region (Pusey *unpublished data*) has substantial capacity to introduce error into any predictive models.

Fourth, models may require a combination of landscape-scale and local habitat-scale variables, that are unlikely to be affected by land use and other stresses, as potential predictors in order to be effective (Pusey *et al.* 2000). Given the diversity of information used here, this luxury was not available and only two variables, both at the landscape scale, were available for use. That is not to say that these variables were not useful – for example, distance from the river mouth and elevation were able to explain 59.4% of the spatial variation in species richness. However, a substantial amount of variation in species richness remains unexplained. These factors place some caveats on our use of predictive models and the extent to which they allow us to discern impacts. Nonetheless, both models (species richness and species composition) provide insights into the way in which fish communities respond to anthropogenic disturbance.

None of the sites examined contained statistically (i.e., < 10^{th} %ile) fewer species than predicted (Figs 5.14 and 5.15) irrespective of the level of disturbance to the immediate or surrounding environment (riparian zone and floodplain, respectively). However, two sites – the most downstream site on Babinda Creek and the penultimate downstream site on Behana Creek – contained fewer than two-thirds of the species predicted. The Babinda Creek site was structurally monotonous, lacked a riparian zone of any description (riparian score of 11.4), was swiftly flowing and was heavily infested with alien weeds such as para grass. The Behana Creek site, in contrast, was distinguished by a near-natural riparian zone (riparian score of 92.5) and contained very few alien weeds. It was, however, structurally very monotonous being a single long run, with a sand-dominated substratum, no in-stream cover and without any bank undercutting. This example illustrates that there are natural causes for low species richness at individual locations and that the causes for such depression in diversity (i.e., local habitat) are not accounted for in our model.

Despite a failure to identify sites with significantly depressed species richness, comparison of O/P scores did reveal how species richness changed within each sub-catchment. No significant change in O/P scores with position in the catchment was noted for the two minimally disturbed sub-catchments but significant associations were noted for Babinda Creek and Woopen Creek. In Babinda Creek, fewer species were observed than predicted (i.e., O/P<1) as one moved downstream away from relatively undisturbed headwater reaches with minimal human land use and intact riparian zones, to downstream reaches with poor riparian integrity and active use of surrounding floodplain for sugar production. In contrast, more species than predicted were recorded with increasing distance downstream in Woopen Creek (i.e., counter to the disturbance gradient). However, detection of a significant relationship with distance appears to have arisen mostly because of the very high species richness (O/P = 1.55) recorded from the most downstream site. Without this one point, there is little suggestion of a significant association of species richness with site position. This site was located only 50 m upstream of the confluence of Woopen Creek and the Russell River. Spatial proximity of small streams to larger rivers is a strong influence on species richness (Gorman 1986) as the larger river provides a source of colonists not usually observed in small streams. The lower site on Woopen Creek was rich in goby species (4 spp.) and also contained *H. compressa* and *G. aprion.*

A downstream change in O/E_{50} scores in Babinda Creek indicated that assemblage composition changed with increasing loss of riparian integrity and increasing agricultural land use. A similar response was not observed in Woopen Creek even though this stream had relatively poor riparian integrity. A possible explanation is that changes in habitat structure wrought by loss of riparian cover, encroachment of para grass into the stream channel and channelling of flow differed between the two streams according to differences in gradient and susceptibility to streambed erosion. In the higher-gradient Woopen Creek, the stream bed was effectively armoured by the large particle size of the substratum (rocks and cobbles). In contrast, Babinda Creek was poorly armoured and as a consequence channelling had caused the stream bed to cut down. The resultant changes of increased depth and increased water velocity create habitat unsuitable to many stream fish (see Pusey *et al.* 2004b). In contrast, the elevated flow and coarse substratum present in Woopen Creek were not too dissimilar from upstream riffle reaches and created habitat suitable for riffle species, *P. signifier* in particular. This species was much more abundant in Woopen Creek than elsewhere and its elevated abundance was the main cause of the significant between-sub-catchment differences in assemblage evenness.

A large proportion of the fish species present in fresh waters of the Russell/Mulgrave River need access to estuarine/marine areas for successful larval rearing. The young of these species need to migrate back upstream to freshwater reaches in order to grow into adulthood. As a consequence, locations close to the river mouth contain such species at a higher proportion of the total species richness and of total abundance than do locations distant from the river mouth. This pattern is especially evident in the ordination plots where the most downstream sites of Woopen Creek and the Little Mulgrave River, and of Behana Creek as a whole, are located to the left, corresponding to locations close to the river mouth, and are distinguished by high contributions to total richness and abundances by estuarine/marine dependent species. Babinda Creek sites, in contrast, are located more to the right and do not fall out along a gradient of position within the riverine landscape. These sites are without a significant contribution to richness and abundance by estuarine/marine dependent species.

5.4.5 Alien fish and their relationship to anthropogenic disturbance

Kennard *et al.* (2005) found that the presence and abundance of alien species were useful indicators of reduced stream health and the present study has clearly identified that some sites with prolific weed growth resulting from reduced canopy cover contained significantly more alien species and individuals than similar sites with good cover. However, other sites with intact riparian gallery forest were occasionally found to contain at least one species of alien fish also, although in such cases abundance was always low. In the present study, alien fish were typically few in number, comprising only 0.5% of the total number of fish collected. At this large scale of examination (i.e., basin), alien species richness and abundance suggest that the fish assemblages are in good condition. However, when the scale of investigation was reduced, differences between sub-catchments and between sites within sub-catchments become apparent. For example, Babinda Creek contained four times as many alien fish as the remaining three sub-catchments. Moreover, the abundance of alien fish increased downstream and with increasing levels of degradation in Babinda Creek. Furthermore, the abundance of alien fish was negatively associated with O/E_{50} scores, suggesting that as anthropogenic factors cause a decline in the suitability of habitat for native species, alien fish were more likely to occur.

When Kennard *et al.* (2005) demonstrated the value of including alien species in fish-based biotic assessment of stream health in south-eastern Queensland they were working in a region with more alien species than observed in the wet tropics. We suggest that the number and abundance of alien species is likely to increase over time in wet tropics streams, given the large pool of alien species in the region (Pusey *et al.* 2004b, 2006). Thus their inclusion and utility as a measure of stream degradation will be even more important in the future and warrant further investigation.

5.4.6 Surrounding land use or riparian degradation as source of disturbance for stream fish?

Agriculture is typically concentrated in lowlands and floodplain valleys, and this is certainly the case in the Russell/Mulgrave basin. The proportion of total catchment area devoted to different human land use was correlated with many aspects of stream habitat that changed along longitudinal gradients. In contrast, changes in riparian integrity were independent of these gradients in stream structure and land use.

The results of the present study indicate that even in the presence of intense agricultural development streams remained 'healthy', as indicated by the observed levels of freshwater fish species richness and assemblage structure compared with the reference state, providing that riparian gallery forests remained in good condition and an adequate buffer was maintained between adjacent cane lands and the stream channel. Behana and Babinda Creeks are the best examples of this pattern. Both have equivalent proportions of their catchments under sugar production but the latter has a depauperate fish community, of low abundance, missing a significant proportion of the natural species assemblage and with more alien fish. The major difference between these catchments is the condition and integrity of the riparian forests along their watercourses. The major impact of a reduction in riparian integrity appears to be a loss of shade and facilitation of introduced immersion tolerant weedy plants (e.g., para grass and Singapore daisy), which then have a range of adverse effects on stream habitat structure, fish assemblage composition and aquatic food webs (Arthington *et al.* 1983, 1997; Bunn *et al.* 1997; Pusey and Arthington 2003; Pusey *et al.* 2004b).

5.4.7 Conclusions and Recommendations

The relationships uncovered in this study, using analyses of factors affecting observed versus expected fish assemblage structure, all point to the value of using fish as indicators of stream degradation resulting from catchment land use and riparian degradation. Fish assemblages in wet tropics streams were particularly responsive to the effects of degraded riparian systems on stream habitat structure, especially aspects of habitat (e.g. velocity) related to the presence and abundance of aquatic macrophytes, including alien species such as para grass and Singapore Daisy (see also Chapter 3). The presence and abundance of alien fish species were also correlated with altered habitat conditions, and were most prevalent and abundant in catchments with a high proportion of land use devoted to sugar cane production and urbanisation. Our major findings reinforce those reported in studies of the utility of fish as indictors of stream health in south-eastern Queensland (Kennard *et al.* 2005, 2006a.b) and elsewhere.

This study demonstrates that modifications to the riparian zone of wet tropics streams can have major implications for the maintenance of their ecological health. Of particular concern in the wet tropics region is that introduced ponded pasture grasses such as para grass and other alien weeds are encouraged by the altered light environment and favourable temperature and water regimes (Bunn *et al.* 1997; Pusey and Arthington 2003). The riparian zone also helps to stabilise bank-associated structures such as undercuts whilst simultaneously providing complexity to the aquatic habitat in the form of. root masses, woody debris and leaf litter. In addition, the fruits of riparian trees and the insects that feed in and on riparian foliage are important to aquatic food webs, particularly in the wet tropics region (Pusey *et al.* 1995b; Bunn *et al.* 1997). Clearly, the riparian zone is very important in maintaining the health of these stream ecosystems (Fig. 5.1).

Accordingly we propose that the next steps in advancing towards a routine monitoring program for stream ecosystem health in the wet tropics should be to explore the use of various methods and tools designed to evaluate the condition of riparian vegetation systems.

This investigation should evaluate the utility of a range of condition metrics (see Werren and Arthington 2002 for a review of approaches), and progress towards the development of rapid assessment methods based on remote sensing techniques, ground-truthed against actual riparian condition. Following this, we suggest that relationships between remotely sensed and ground measures of riparian condition and fish assemblage structure be further explored in a wider range of tropical catchments. Linked to this, we recommend further work on the factors and processes that underpin the observed effects of riparian degradation and aquatic macrophyte proliferation on stream fish assemblages, including effects on alien species. These process studies should include examination of food web structure and how riparian degradation may alter sources of carbon and food web dynamics, and the effects of riparian modification on fish habitat structure, movement requirements and life history processes.

It is essential that the observed relationships between stress factors associated with land use and biotic response be understood and quantified in order to design riparian restoration works and justify investment in riparian rehabilitation.

A final research theme should be to establish thresholds of ecological response to land-use stress, such that the degree of modification of riparian vegetation, and other factors, that endanger stream health can be identified before the stream ecosystem reaches a critical level of deterioration. Procedures for establishment of such thresholds are outlined in Arthington *et al.* (2006). In essence they involve exploration of the response of ecosystem health indicators to gradients of land-use stress, that is, gradients of water quality, riparian condition, flow regulation and alien species, either separately or in combination. The development of quantitative stressor-response relationships should be the major advance in the next phase of stream ecosystem health assessment in the wet tropics and other northern river systems.

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Chapter 6

Summary and Synthesis: Integrated Protocols for Monitoring the Ecosystem Health of Australian Wet Tropics Streams.

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6.1 Introduction

This chapter summarises the findings of the individual chapters of the River Health component of the *Catchment to Reef* program, compares their approaches and outcomes, and evaluates their utility for a river health monitoring program. It presents a suite of indicators suitable for assessment of stream ecosystem health, based on the research findings. This synthesis is preliminary and focuses on streams in the central wet tropics (Russel-Mulgrave catchments). Testing of the conclusions is planned elsewhere in the wet tropics, across other Great Barrier Reef (GBR) catchments and in other habitats (highly seasonal systems, lowland wetlands) as part of the Marine and Tropical Science Research Facility (MTSRF) research program, which will also develop reporting mechanisms that will be readily accessible to all stakeholders. The MTSRF program builds on, extends and will conclude the *Catchment to Reef* research reported here.

This component of the *Catchment to Reef* program was designed to elucidate appropriate indicators of the ecological health of streams and rivers in the wet tropics. It benefited from (i) the study team's extended prior experience of wet tropics streams through research programs at James Cook University and Griffith University, supported by several funding bodies including the Rainforest CRC, ARC and Land and Water Australia; (ii) various prior models for examining the influence of human activities on stream health; (iii) the opportunity to compare essentially similar catchments with different land-use and management practices; (iv) the ready cooperation and interest of landholders and community members; and (v) the diverse range of skills of the study team.

6.2 Approach

Following a review of monitoring methods developed elsewhere, several models were possible candidates for application in monitoring the ecosystem health of wet tropics streams. All approaches required some means of comparing test sites with reference sites, and included the RIVPACS/AusRivas system (Wright 1995; Simpson and Norris 2000) based on macroinvertebrate community composition; the SIGNAL system (Chessman 1995, 2003) based on sensitivity-weighted scores for macroinvertebrates; and various methods that compared observed with expected presence of species, derived from models based on site characteristics and extensive prior information on the distribution of species (e.g. fish – Kennard 2006a,b). The DIBM3 program in south-eastern Queensland (*Design and Implementation of Baseline Monitoring*; Smith and Storey 2001) presented such an approach that was both contemporary and based in the same broad geographical region (eastern Australia), albeit in a different climatic zone. Most approaches use some form of 'observed *vs.* expected' (O/E) relationship, in which an O/E score can be used to assess how much a biotic community found at a specific location differs from what is predicted by the model (based on reference sites). An O/E score of 1.0 indicates that all species expected to occur at a given location were observed there, while a score of 0.5 means that half of the species expected to occur were missing from that location.

The choice of study components ('ecosystem response variables') was based on pragmatic criteria: the components of stream systems known to be sensitive to human impacts; components we know most about; and components that lend themselves to relatively straightforward sampling, processing and data analysis. The last point was important because the intent was to develop monitoring protocols that could be applied regularly and cost-effectively by persons or groups with a range of scientific skills: from government agencies to community groups (always with appropriate scientific back-up for design, analysis and interpretation). Our suite of response components was: physical/geomorphic characteristics of the stream, water quality, riparian integrity, aquatic macrophytes, macroinvertebrates and fish. Water quality was the focus of a separate program within *Catchment to Reef*, but it was important to include it in the evaluation of all components as an indicator of both current condition and catchment influences. Other components were also considered but discarded: for example, microbial and micro-algal community composition were at the more difficult end of the spectrum of biota in terms of sample processing and community acceptance. The DIBM3 program used a similar suite of components in south-eastern Queensland, with the additional inclusion of stream metabolism, a measure of overall ecosystem function. However, stream metabolism was not included in the *Catchment to Reef* program because it involved specialist equipment and expertise and was regarded as too involved for a program suitable for multiple users.

DIBM3 provided a partial template for development of this program. However, we recognised the opportunity of a 'paired catchment' comparison in the Russell-Mulgrave catchment that was not available to DIBM3. Further, it was not assumed that the DIBM3 approach and indicators could be simply bolted on to the wet tropics, given the differences in the environments, and our understanding of differences in tropical systems, such as levels of diversity, flood frequency, rates of key process, etc. (Pearson *et al.* 2003; Connolly and Pearson 2004; Pearson 2005; Connolly *et al.* 2007). Moreover, while extensive data sets were available for invertebrates in upland streams in the wet tropics, comparable data for floodplain streams was sparse, and so a robust predictive model for the invertebrates was not available (one was developed in this program). Lack of appropriate prior information was also an issue for the use of aquatic plants as indicators of stream ecosystem health in the tropics. For freshwater fish, on the other hand, extensive data were available for floodplain streams (see Pusey *et al.* 2004). Our study design, therefore, allowed for application of the DIBM3 approach where appropriate, but deviated somewhat, in that it involved comparisons of sites between paired catchments. The use of both approaches facilitated an understanding of the ecological processes influencing distributions of biota in the streams, and enabled a comparison of the methodologies used to detect the influences of land-use disturbance on these ecosystem components.

Both the DIBM3 and paired-catchment approaches (which are not mutually exclusive) depend on identification of key physico-chemical factors associated with land-use disturbance, to allow association of responses to influences. Unfortunately, in many cases such associations are confounded by natural gradients. For example, while we predict that agricultural contaminants will increase in concentration in a downstream direction as the proportion of the catchment under agriculture increases, we also know that the number of fish species increases as we move from headwaters to the river mouth as a result of increased habitat diversity and availability, and greater access to estuarine fish (e.g., Hortle and Pearson 1990; Pusey *et al*. 1995), independently of agricultural contaminants. Meanwhile, there is a natural decline of invertebrate diversity downstream, independent of any human impacts. Without appropriate reference sites or (preferably) streams, unravelling causes and effects and differentiating them from these natural gradients is almost impossible. A dearth of appropriate reference sites in floodplain systems (because there are few sites that are free from human impact) can cause problems in model development; however, if we sample along natural gradients with sufficient intensity, it may be possible to partition out the effect of the natural gradient and identify impacts. In this way, what is not possible to differentiate site-by-site may become possible stream-by-stream.

The case study on which much of this report is based allowed us to consider these issues across a range of response variables, while simultaneously inferring some strong cause-effect relationships that pointed to appropriate candidates for indicators of stream ecosystem health. We were mindful that there are a number of misconceptions about indicators, and that care is needed in attributing effects to causes without appropriate evidence. Thus, absence of some key fish species suggests that conditions are not appropriate for that species; but 'condition' can mean a large range of things, such as fish access to the site (e.g., no barramundi above waterfalls), time of year (with regard to seasonal migrants), temperature, presence of toxicants, or absence of suitable feeding, breeding or predator-avoidance habitat (Pusey *et al.*– Chapter 5). Further, some species are not good indicators of system health: for example, the platypus is an iconic species but one that is not greatly sensitive to some types of poor water quality, such as organic enrichment (which may actually enhance food availability for the platypus). Platypus presence is, therefore, a good indicator of the status of the platypus, but not necessarily of stream health. On the other hand, a simple measure like integrity of the riparian zone can be a very good predictor of stream condition (other things being equal – there are exceptions to all generalisations as indicated below) and therefore a good indicator of overall health, because the riparian zone has major influence on stream processes (Pusey and Arthington 2003). In this case, although riparian integrity is less the indicator and more the proximal cause of the stream's condition, it is still a very useful measure once the links between riparian condition and its effects are elucidated (though it too cannot stand alone, as indicated below).

The probable links between cause and effect are mostly determined by correlation of variables measured in the field. This approach can become circular, a circle that may be broken by manipulative experiments of the type typically done to assess toxicity of particular contaminants in particular situations. Examples include assessment of the effects of heavy metals on selected species in association with mine-site wastes (in the tropics there has been extensive work of this nature at the Ecological Research Institute of the Supervising Scientist in the Northern Territory). There are few studies of the direct effects of more prosaic but widespread contaminants, but studies on impacts of hypoxia and ammonia on fish and invertebrates in the Queensland tropics are providing some headway in this challenging arena (Pearson and Connolly 2000; Pearson *et al.* 2003; Økelsrud and Pearson 2007; Connolly *et al.* 2004, 2007; Flint 2007). Greater research effort in this area would help put real values into our understanding of thresholds, and into sensitivity weightings of species (as in SIGNAL – Chessman 1995, 2003), which currently appear not to be appropriate for tropical taxa (Connolly *et al.*, Chapter 4). It might also be noted that such research can inform us of species whose presence (a more powerful measure than absence) signifies impacts: for example, this has been shown for a species of midge larva in Babinda Creek, (Pearson and Penridge 1987).

Our approach involved the survey of four streams in the Mulgrave and Russell catchments that differed in levels of degradation and land management. Land management in the Mulgrave catchment approaches current best management practice, whereas the Russell catchment suffers many common problems such as cleared riparian vegetation and the proliferation of weeds. We sampled up to 40 sites, which were distributed at approximately 1 to 1.5 km intervals along each stream, from the base of the foothills to the confluence with the Mulgrave or Russell rivers. The data from these samples were then subjected to a variety of statistical analyses to develop and/or test our models and explore cause-and-effect relationships. The main conclusions from this work are summarised below.

6.3 Characteristics of the study streams and catchments

The case study focussed on lowland sections of streams in the Russell and Mulgrave catchments, characterised by good perennial flows and short-lived floods in the wet season, forested uplands, and extensive agriculture on the floodplain. The stream sediments have strong gradients in particle size, from bedrock, boulders and cobbles upstream to finer gravels and sands downstream. Behana and Babinda Creeks narrow downstream rather than widen, in contrast to the classic model of streams widening with distance downstream, probably a result of them over-topping their banks at the base of the mountains during floods so that the full stream discharge is not contained in the channel during high flows. Land use on the floodplains of Behana and Babinda Creeks was dominated by sugarcane production, with smaller areas of cane in the catchments of the Little Mulgrave River and Woopen Creek, where other land-uses covered similar areas.

6.4 Channel characteristics and water quality

The overall physical condition of the streams (bank structure, riparian integrity, etc.) generally decreased downstream in all streams, with Woopen and Babinda Creeks being in worse condition than the Little Mulgrave River and Behana Creek. There were distinct differences in channel integrity among streams. Geomorphic conditions of the banks were generally poor in Babinda Creek, as a consequence of poor riparian structure. Mass bank failures were evident along many reaches of the stream. Where both banks had dense stands of para grass (*Urochloa mutica*) or Singapore daisy (*Sphagneticola trilobata*) (both introduced invasive weeds), the flow had become channelised and velocities had increased. This appears to have caused incision in the lower reaches of Babinda Creek, creating a uniformly deep, featureless, sandy channel, with swift flows. In contrast, the lower reaches of Behana Creek were shallow, meandering from side to side within the banks, and sand bars were present. Although there were some stands of para grass in Behana Creek, the edge habitats were varied and included tree roots and other complex structure important to fish. In Babinda Creek, this habitat were almost completely overgrown, consisting of inundated stands of Singapore daisy or para grass, and habitat diversity was generally low.

Water quality in all study streams was generally good during base flows, although nitrate plus nitrite (NO_x, the main constituents of nitrogen fertilisers) and total phosphorus concentrations exceeded the 2006 Queensland Water Quality Guidelines in all study streams. Nitrogen concentration, particularly NO_x, increased substantially with distance downstream, and Babinda Creek had consistently higher NO_x concentrations than Behana Creek. Differences in the patterns of NO_x concentrations between streams can be explained largely by differences in the proportion of land-use types in the catchments, with greater concentrations at sites with greater areas of agriculture in the catchment. This relationship makes it possible to compare streams and thus to directly compare the effects of different land-use practices and riparian vegetation cover. For example, Behana Creek, which had greater riparian vegetation cover along the stream length and application of best practice land management, had lower concentrations of NO_x than the other three study streams.

Concentrations of phosphorus were variable within and between streams, and concentrations of total phosphorus exceeded Queensland Water Quality Guidelines at nearly all sites in all four study streams as a result of high concentrations of particulate phosphorus and filterable organic phosphorus. Concentrations of filterable reactive phosphorus only exceeded these guidelines consistently at sites in the lower reaches of Woopen Creek.

The longitudinal gradient in nutrients and the effect of rainfall demonstrate that it is impractical to compare the nutrient concentrations between sites and attribute a cause without considering other factors such as adjacent land-use management and extent of riparian vegetation. The comparison between Behana and Babinda Creeks represented a large-scale test of the effectiveness of riparian vegetation and management approaches in reducing agricultural contamination, taking into account the relative amount of agriculture in each catchment. Land management in Behana Creek approaches current best management practice with regard to protecting river ecosystem health and reducing the transport of agricultural contaminants to the Great Barrier Reef lagoon. This was reflected in lower nutrient concentration in Behana Creek, but further work is needed to show that this observation was not due to differences in rainfall and hydrology. Nevertheless, it was clear that a substantial change in nutrient management would be needed even in the Behana catchment if the instream water quality measures were to come close to the Queensland Water Quality Guidelines.

Tracking the influence of agriculture along each stream and accounting for differences in the relative proportion and distribution of land-use cover was a powerful means of examining these effects on a large scale. It seems very likely that such a detailed approach to stream descriptions will be vital in disentangling potential influences on the ecological integrity of the streams. This approach would clearly benefit any future health assessment of streams in the region and elsewhere.

6.5 Macrophytes

Assemblages of aquatic macrophytes (non-microscopic plants) of Australian lotic ecosystems have not been well described (Mackay *et al.* 2003). Consequently, the responses of aquatic macrophytes to anthropogenic disturbance of river catchments are not well known except in terms of gross assemblage changes such as infestation by alien species. This study used macrophyte assemblage composition and simple assemblage metrics to investigate whether aquatic macrophyte assemblages of wet tropics streams could be used as reliable indicators of catchment land-use, riparian condition and water quality.

Aquatic macrophytes were surveyed at 34 sites with regard to spatial variation, cover, species richness, native *vs.* alien species, submerged plants, emergent plants, and grasses. Forty-four taxa were recorded from the study area. Grasses, sedges and mosses were the most frequently occurring taxa. Key alien species infesting stream banks were para grass and Singapore daisy.

The most consistent macrophyte association found for the wet tropics region was the bryophyte *Cladopus queenslandicus* assemblage that occurred in headwater sites located at the base of the mountains, where substrata were mostly rocky, the riparian zone was less disturbed and the canopy cover was good. Macrophyte assemblages of most lowland sites were dominated by emergent vascular species. Emergent assemblages occurring in the Little Mulgrave River were characterised by a variety of taxa but only ferns (Pteridophytes) appear to have any utility as bioindicators. Ferns were present in many of the sites in the Little Mulgrave River and a small proportion of the headwater sites, and typically were absent from relatively disturbed sites with greater proportion of anthropogenic land uses in the upstream catchment area, lower riparian condition and lower riparian cover. Introduced para grass and Singapore daisy were particularly abundant where there was poor riparian canopy cover.

The study identified relationships between riparian disturbance and macrophyte cover in edge habitats, as well as the diversity and cover of alien species and grasses. Macrophyte cover in edge habitats is potentially the most useful of these indicators. In contrast, the use of macrophytes as indicators of catchment land use and water quality appears limited. Riparian condition and riparian canopy cover were more important drivers of macrophyte assemblage structure than land use in the study streams. Maintenance of riparian zones would therefore seem to be the soundest way to maintain intact aquatic macrophyte assemblages in wet tropics streams, provided that other environmental factors remain more or less undisturbed (e.g., flow regime, water quality, grazing pressure).

6.6 Macroinvertebrates

The macroinvertebrate fauna of wet tropics streams is very diverse compared with streams globally (Pearson *et al.* 1986; Vinson and Hawkins 2003; Boulton *et al.* 2007) and has high conservation value. It has previously been shown to have utility in indicating major impacts to stream ecosystem condition with regard to sugar mill effluent (Pearson and Penridge 1987).

In this study, 118 macroinvertebrate taxa were collected (not all were identified to species level). A strong upstream-downstream gradient in composition and richness was apparent for all four streams, and it was clear that substratum particle size was a strong determinant of this gradient. Sites were distinguishable along the stream gradient by the number of taxa, but there were also clear changes in composition and presence of taxa associated with particular sediment sizes. The strength of the longitudinal gradient, and the strong association between the substratum particle sizes and the macroinvertebrate assemblage composition, provided predictability that facilitated comparisons between streams. This relationship will be useful in developing predictive models to use in monitoring programs.

The use of the mean sediment particle size as a covariate in analysis of covariance was a robust way of detecting differences between streams. For example, the relationship between taxon richness and mean sediment size was similar in both Behana and Babinda Creeks, but the number of taxa was significantly lower in Babinda Creek sites by about 20-25%. If only a few sites had been compared, and site selection had not been carefully stratified by sediment particle size, such differences would probably not have been apparent. These results highlight how an understanding of the natural gradients is vital to prevent inappropriate comparisons and conclusions.

Differences in the macroinvertebrate assemblages between streams were attributable to the differences in riparian vegetation cover, which affected the amount of coarse particulate organic matter (CPOM) available in the streams. When substratum sediment size was accounted for, there was a significant difference in the number of invertebrate taxa at sites with or without riparian vegetation with the effect driven by the availability of CPOM in the stream.

None of the water quality parameters correlated with macroinvertebrate distributions, despite there being high concentrations of agricultural nutrients in all streams surveyed. For example, the concentrations of dissolved inorganic nitrogen generally increased with the proportion of the area of catchment used for agriculture. However, it was not clear what effect this enrichment had on the macroinvertebrate assemblages. Because other water quality parameters were generally unremarkable, with high flows and reasonably cool water temperatures, and there was no obvious increase in growth of algae, enrichment is likely to be having only minor impacts in these streams. Although apparently high densities of some taxa may be indicative of nutrient enrichment, it is impossible to compare between streams because nutrient levels in all streams were enriched.

Significant differences in the macroinvertebrate assemblages between streams were detected using the number of species, the number of families, or the number of PET (Plecoptera, Ephemeroptera and Trichoptera) species collected at a site. Significantly, however, the popular index, SIGNAL, recommended in the DIBM3 for use in the Ecosystem Health Monitoring Program in south-eastern Queensland, was unable to detect differences between the macroinvertebrate assemblages in Behana and Babinda Creeks. Our results show that there is a trade-off between the number of sites sampled and the level of detail necessary for each site. Thus, indices such as the number of families or PET can be used to detect differences if several sites are sampled in each stream. However, if only a few sites are sampled, or if a reference model is used, then species richness is the measure that will have the best likelihood of detecting differences between the streams. Similarly, the ability to detect differences between Behana and Babinda Creeks is a trade-off between the number of sites sampled and the sampling effort, measured by number of individuals collected at a site.

Therefore, if taxonomic expertise is not available, it may be beneficial to sample more sites but limit identification to specific groups or to the family level. If a higher level of taxonomic expertise is available, savings can be made by surveying fewer sites. However, in both situations, expertise is required to design the appropriate sampling scheme and to decide on the number and locations of sites. Appropriate design is based on prior knowledge or a pilot study in the region to assess variability, and includes the need to stratify sample locations or account for gradients in macroinvertebrate distributions.

Macroinvertebrate assemblages proved to be powerful indicators of stream ecosystem health. They provided a tool that could detect differences between streams that differed in condition, with strong statistical support, and so provided defensible evidence of impacts and strong inference regarding causal factors. This information will now improve the design of future ecosystem health monitoring protocols and guide management actions to remediate some of the impacts that are degrading wet tropics streams.

6.7 Fish

The freshwater fish fauna of the wet tropics is diverse and distinctive in several respects, with a total of 103 native and four alien species recorded from fresh waters of the region, representing 37 families with almost half (52/107) within six families. During this study of the Russell-Mulgrave catchments we recorded 30 species (27 native, 3 alien) from 15 families. Fourteen of the native species collected have life histories completed entirely within fresh water, with the remaining 13 species requiring access to saline waters for spawning or larval development.

This study found strong changes in freshwater fish assemblages associated with variation in habitat structure and position in the catchment. Species richness increased with sub-catchment size and decreased with increasing distance upstream from the river mouth. The proportional representation of species dependent on estuarine or marine influence also decreased with increasing distance upstream from the river mouth. Between-stream comparisons of fish assemblages must therefore recognise differences in catchment size and position within the landscape, and how these factors might affect the comparisons. Any investigation of changes in fish assemblage structure associated with different land-uses and catchment health must take these natural distribution and species richness patterns into account, as noted above for macroinvertebrates.

Woopen Creek and the Little Mulgrave River contained more species than the more degraded Babinda Creek, while the composition of the fauna in Babinda Creek was markedly different from that in Behana Creek, which it closely resembles in terms of size and position within the riverine landscape. Behana Creek also contained significantly more species site-by-site than were observed in Babinda Creek. The average number of fish collected per site was significantly lower in Babinda Creek than in the other streams, paralleling the results from the invertebrate samples. Babinda and Behana Creeks also differed with respect to the presence and abundance of fish with an estuarine or marine larval interval and which must migrate upstream as juveniles. Several species (e.g., jungle perch, catfishes) were either absent from Babinda Creek or occurred in lower abundances than in Behana Creek. The catfishes are probably absent from the lower Babinda Creek because key bank-related habitat features (Pusey *et al.* 2004) are either smothered by invasive weeds or are no longer present because of denudation of the riparian zone.

Differences in species richness and assemblage composition indicated that Babinda Creek differed substantially from the other three streams. The fish assemblages at downstream sites in Babinda Creek were different from expected, a trend that corresponded with a strong physical gradient, from relatively undisturbed headwater reaches to downstream reaches with poor riparian integrity and active use of surrounding floodplain for agriculture. The downstream change in observed/expected scores in Babinda Creek showed that assemblage composition changed with increasing loss of riparian integrity and increasing agricultural land use.

Alien species were not common in the four study streams. Only three species (the platy, *Xiphophorus maculatus,* the guppy, *Poecilia reticulata* and the cichlid *Tilapia mariae*) were recorded and they collectively accounted for only 0.8% of the total number of fish collected. However, alien contribution to total abundance reached 12% at the most downstream site in Babinda Creek, and overall, when alien richness was up to three species, abundance was also elevated. The greatest number of alien species occurred at low elevation, close to the river mouth and in areas with a high proportion of the catchment devoted to agriculture and urbanisation. The abundance of alien species was correlated with these forms of catchment land use, suggesting that as anthropogenic factors reduced the suitability of habitat for native species, the number and abundance of alien species increased.

The results of this study indicate that, even in the presence of intense agricultural development, streams remained 'healthy' (as indicated by the observed levels of freshwater fish species richness and assemblage structure compared with the reference state) provided that riparian gallery forests remained in good condition and an adequate buffer was maintained between adjacent sugarcane lands and the stream channel. Behana and Babinda Creeks were the best examples of this pattern. Both have equivalent proportions of their catchments under sugar production but the latter has a depauperate fish community, of low abundance, missing a significant proportion of the natural species assemblage and with more alien fish. The major difference between these streams is the condition and integrity of their riparian vegetation, mainly related to loss of shade and facilitation of introduced immersion-tolerant weedy plants (e.g., para grass and Singapore daisy).

The relationships uncovered in this study, using analyses of factors affecting observed *vs*. expected fish assemblage structure, all point to the value of using fish as indicators of stream degradation resulting from catchment land use and riparian degradation. Fish assemblages in these wet tropics streams were particularly responsive to the effects of degraded riparian systems on stream habitat structure, especially aspects of habitat (e.g. velocity) related to the presence and abundance of aquatic macrophytes, including alien species such as para grass and Singapore Daisy (see also Chapter 3). The presence and abundance of alien fish species were also correlated with altered habitat conditions, and were most prevalent and abundant in catchments with a high proportion of land-use devoted to sugarcane production and urbanisation. Our major findings reinforce those reported in studies of the utility of fish as indictors of stream ecosystem health in south-eastern Queensland (Kennard *et al.* 2005, 2006a,b) and elsewhere.

6.8 Status of the test streams

The components of this case study show strong parallels, indicating that the observed impacts had a general effect across the spectrum of biophysical variables. It was clear that, even in the presence of intense agricultural development, some streams could be 'healthy' (e.g., Behana Creek) as demonstrated by all biological indicators, providing that riparian vegetation remained in good condition and an adequate buffer between adjacent agricultural land and the stream channel was maintained. The major impact of a reduction in riparian integrity appeared to be a loss of shade, a loss of detrital input and facilitation of alien immersion-tolerant weeds, which then had a range of negative impacts on stream habitat structure (particularly bank-associated habitat structure) and aquatic food webs. Changes in habitat structure also appeared to inhibit the upstream migration of species with a marine or estuarine interval in their life history. Therefore, the study indicated that there are major stream health benefits to be gained from land management approaches that are sensitive to environmental values.

The difference between the nutrient concentrations in Behana and Babinda Creeks is particularly interesting given the different riparian cover and land-use management practices in their catchments. This comparison potentially represents a large-scale test of the effectiveness of riparian vegetation and land-use management practices on reducing agricultural contamination on wet tropics streams. It is apparent that while riparian vegetation had a strong influence on the biodiversity in these streams, it had a limited effect on contamination by dissolved inorganic nitrogen from on-farm fertilisers when a large proportion of the catchment was used for agriculture. This is significant given the goals of the Great Barrier Reef Protection Plan and because riparian rehabilitation is touted as a key management tool in reducing contaminant loads entering streams. Thus, while riparian restoration is vital for aquatic and terrestrial habitat values, its role in substantial contaminant stripping in wet tropics streams remains to be demonstrated.

As a case study, these results are to some extent specific to the focus streams. It is probable that streams of different character will respond differently to the potential stressors. For example, effects of elevated nutrient concentrations on photosynthesis and hypoxia are much more prevalent in slow-flowing streams and riverine waterholes in the wet tropics (Pearson *et al.* 2003) and the dry tropics (unpublished data), and case studies in such places are required to generate and test appropriate models. In such cases local factors may have greater influence than catchment-scale effects, and riparian condition may have greater influence on water quality that is apparent for perennial streams in the wet tropics. Nevertheless, similar driving factors are important to biodiversity values, including the integrity of the riparian vegetation and the prevalence of habitat modification by weeds; and notwithstanding differences between systems, our approach in this case study is applicable in these other situations.

6.9 Comparison of methods

Good correspondence between the study components raises the question of redundancy: can we achieve our monitoring aims by simply measuring one or two ecosystem response variables and so simplify the process? For example, if the riparian vegetation is so important, could it be used as a surrogate indicator for stream condition? While poor riparian condition is a good predictor of poor ecosystem health (as in the case study), good riparian condition does not necessarily indicate good stream health, as other factors can be important. For example, in a study of stream health in the Mackay region of the central Queensland coast, Reliance Creek was found to have intact riparian vegetation but a degraded invertebrate fauna resulting from organic inputs from a mill and resultant hypoxia (Pearson and Penridge 1992). Good riparian condition is clearly a requirement for healthy streams, but not the only requirement. Under circumstances where there might be a suite of stressors, the biota has long been recognised as the best integrator of environmental conditions, such that long-term presence of a species indicates that conditions are suitable for that species. But note the differences between examples above: the platypus appears to be tolerant of organic pollution, the larvae of a species of *Chironomus* midge thrive on it, while many invertebrates and fish avoid it. Furthermore, species may be considered important in their own right and therefore worthy of monitoring to assure managers of their status. Iconic species are clear contenders (e.g., platypus, barramundi, jungle perch), but most managers these days would hope for a healthy biological community, which includes all components. Only those components can reliably indicate their own population or community 'health'.

To test the suggestion of redundancy or surrogacy we might consider which elements could be dropped from a river health monitoring program:

- The physical description of the stream environment was necessary to classify streams so that we could compare like with like (size, catchment characteristics, flow regime, etc.), underpin the models (e.g., sediment size) and highlight issues potentially important in interpretation of results (e.g., flow, depth, bank and riparian integrity). Physical description, apart from detailed geomorphic profiling, is simple and rapid to perform.

- Water quality and other environmental data are vital for characterising stream conditions and pin-pointing probable sources of problems. Basic information on temperature, light environment, conductivity, pH, dissolved oxygen and clarity are easy to measure using appropriate field instruments. However, they are not always measured well. Dissolved oxygen and pH, for example, can cycle substantially over a 24-hr period, so when to sample becomes a major issue (addressed in the accompanying Water Quality report). Nevertheless, when measured properly, these water quality variables can be important indicators of aquatic ecosystem health. Other factors, such as dissolved nutrients, require careful collection, including careful planning with regard to timing and placement of samples, storage and laboratory analysis, and can be expensive unless appropriately targeted. However, understanding of nutrient status of streams is clearly useful, although in the case study nutrients were not directly implicated in affecting other components (nitrate and phosphate are not usually toxic, but they do promote growth of algae and invasive weeds, especially in slow-flowing situations).
- Riparian and instream macrophytes are important in their own right from a biodiversity perspective, but also have an important bearing on conditions experienced by the fauna. They variously provide or alter habitats and substrata and contribute to productivity. Their presence can indicate health, while the presence of invasive weeds in large quantities indicates the opposite. Aquatic plant cover and approximate diversity measures can be done rapidly, but species-level monitoring requires appropriate taxonomic expertise.
- Invertebrates are the most widely used monitors of river health in Australia and globally. Their diversity is such that different species can respond to environmental factors in different ways, so the overall invertebrate community composition can be a good and subtle indicator of conditions (e.g., gradual switching from various chironomid species to *Chironomus* in deteriorating conditions (Pearson and Penridge 1987)). A simple measure – species or family richness – gave clear and significant signals in this case study, and confirmed the value of invertebrates in monitoring. Invertebrates are easy and cheap to sample effectively, and are abundant throughout the longitudinal river profile. Sample processing in the laboratory can be prolonged and expensive, but more rapid techniques are now available (e.g., we recommend subsampling from large composite samples). Species-level studies require taxonomic expertise and so can be expensive but, as we have shown, family-level identifications are effective, even if less powerful than species-level. Skills in identification to family level are achievable quite readily by interested individuals. However, as with all measures, expertise in sampling design, data analysis and interpretation is required.
- Fish are of direct interest to a broad spectrum of the community, are key components of stream ecosystems and, like invertebrates, are good indicators of stream conditions (albeit at a larger scale because of their mobility). Scientific sampling of fish can be involved and relatively expensive, depending on frequency (determined by the questions being addressed, see below), but laboratory processing (except for follow-up of difficult species identifications and specific studies of diet, breeding, etc.) is not required if the personnel have appropriate taxonomic expertise. Thus most of the samples can be returned to the water alive.

A composite approach to health monitoring is therefore required. This not a novel finding: for example, in south-eastern Queensland, DIBM3 resulted in the combination of five components which, apart from the inclusion of stream metabolism, were similar to those used here. However, more components in a monitoring program mean greater complexity and expense, so application of those components needs to be judicious and cost-effective. Our recommended measures are indicated below.

6.10 Monitoring protocols

Our monitoring manual will be produced as part of the MTSRF program. It will deal with monitoring as a 'question-driven' activity, rather than as a data-amassing exercise, and will include methods for appropriate analysis and interpretation to meet particular needs. By 'question-driven' we mean that it is important to be clear *why* monitoring is being undertaken (e.g., is it for decadal, annual or monthly reporting of condition; is it to examine riverine and catchment condition; is it to determine quantities of contaminants being delivered to the GBR etc.?). This consideration then drives other questions: *what*, *how*, *when* and *how often* to sample and *who* to plan, administer and undertake the exercise (including sampling, sample analysis, data analysis and interpretation) are all crucial issues that will be addressed in the manual.

As indicated above, depending on the questions being addressed, and subject to an appropriate study design (multiple sites *vs.* appropriate models) and methods, our suite of recommended variables to describe/measure in streams of the wet tropics is:

- flow regime of the stream;
- physical condition of the stream sites including: current velocity; bank stability; channel form; width; depth; and sediment characteristics, including particle size and amount of detritus;
- major water quality characteristics, including maximum and minimum values (measured through repeated 24-hr cycles) of temperature, conductivity, pH, dissolved oxygen, clarity, suspended solids, hardness, nitrate, phosphate;
- riparian condition (vegetation structure, weediness, canopy cover);
- aquatic macrophyte cover;
- species richness of aquatic macrophytes;
- proportion of aquatic macrophyte species that are alien;
- species richness of invertebrates ('species' here meaning taxa at highest level of resolution possible);
- family richness of invertebrates;
- fish species richness and assemblage composition;
- number and proportion of alien fish species;
- proportion of fish abundance due to alien species.

These same variables will form the basis of monitoring programs of rivers and wetlands of different character, although the study designs will need to be modified to incorporate flow regime characteristics, and physical-chemical gradients in slow-flowing, intermittent and non-linear systems, such as floodplain lagoons. Methods for different types of system (e.g. wet tropics *vs.* dry tropics) will be tested in the MTSRF program.

6.11 References

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