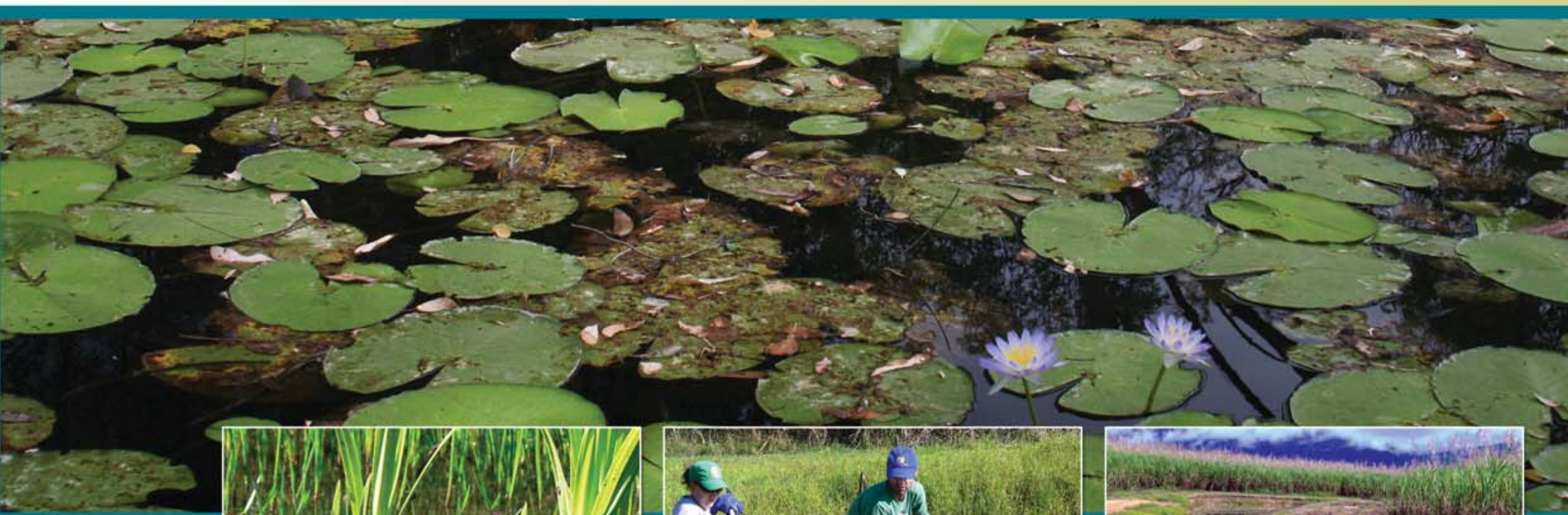
Marine and Tropical Sciences Research Facility

Ecosystem Health of Wetlands of the Great Barrier Reef Catchment: Tully-Murray Floodplain Case Study



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with contributions by J. Wallace, F. Karim and M. Ellison

Australian Government
Department of Sustainability, Environment, Water, Population and Communities

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Reef & Rainforest RESEARCH CENTRE

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Acronyms and Abbreviations

ACTFR .... Australian Centre for Tropical Freshwater Research
ANOSIM .... Analysis of similarity
ANZECC .... Australian and New Zealand Environment and Conservation Council
ARI .... Average Recurrence Interval
CPUE .... Catch per unit effort
CRC................... Cooperative Research Centre
CSIRO .... Commonwealth Scientific and Industrial Research Organisation
DSS .... Decision Support System
GBR .... Great Barrier Reef
GIS .... Geographic Information System
GradDip .... Graduate Diploma
GU .... Griffith University
JCU .... James Cook University
LCMS .... Liquid chromatography mass spectrometry
LOR .... Limit of reporting
MAppSci .... (Degree of) Master of Applied Science
MRPP .... Multi-response Permutation Procedure
MTSRF .... Marine and Tropical Sciences Research Facility
NGO(s) .... Non-Government Organisation(s)
nMDS .... Non-metric multidimensional scaling
PC1 .... Principal Component (PC1, PC2, PC3, etc.)
PCA .... Principal Components Analysis
PhD .... (Degree of) Doctor of Philosophy
RIS .... Research Investment Strategy
WTWHA .... Wet Tropics World Heritage Area

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Summary

S1: Background

- This project aimed to measure and understand the ecological processes, linkages and interdependencies that govern the biodiversity, physical condition, ecological health, temporal trends and resilience of rivers and floodplain wetlands in catchments of the Great Barrier Reef (GBR).
- The project builds on research in the freshwater ecosystems of the Wet Tropics over the past twenty years, and on the former joint *Catchment to Reef* program[1] of the Rainforest CRC and CRC Reef Research Centre.
- This research can provide the basis for cost-effective biophysical monitoring tools to help sustain these habitats and their dependent environmental assets, and will serve as a benchmark for monitoring and other studies (e.g. effects of climate change).
- Below we summarise key findings, followed by a point-by-point outline of the main research findings.

S2: Summary of key findings

- The most serious factors affecting health in Wet Tropics streams and wetlands are changes to habitats (flow modification, loss of riparian vegetation, weed invasion, infrastructure).
- Water-quality effects of agriculture are evident in streams and lagoons, particularly in systems that are not flushed throughout the year. In perennial streams (e.g. Russell-Mulgrave) and lagoons (e.g. Tully-Murray), dilution ameliorates impacts.
- The diverse aquatic invertebrate assemblages of Wet Tropics waterways are sensitive to habitat and water quality changes; they are good indicators of local ecological conditions.
- Wet tropics waterways provide essential habitat, including nursery habitat, for unique fish assemblages, which are very good indicators of the physical characteristics, hydrological connectivity and ecological condition of sub-catchments and floodplain lagoons.
- The presence of alien fishes is a strong indicator of disturbance in the broader landscape, and an early warning indicator of the potential for further disturbance from increasing numbers of individuals and species (e.g. tilapia).
- The alien ponded-pasture grass *Hymenachne amplexicaulis* and other alien plants that now occupy Wet Tropics waterways are avoided by most native fish species.
- Surrogate measures such as the Cassowary Coast Regional Council's (previously, Cardwell Shire's) Floodplain Project Values and Threats scores can provide rapid assessment of waterway ecosystem health.
- Connectivity pathways of streams and wetlands to estuaries, coasts and the Reef are an essential feature of Wet Tropics waterways, and can be compromised by infrastructure, flow regulation, weeds and water quality barriers.
- Ecological condition of floodplain lagoons may serve as a powerful indicator of climate change, because they are vulnerable to sea-level rise and hydrologic alterations, especially loss of flood pulses, dry-season base flow and connectivity between rivers and wetlands.

[1] http://www.rrrc.org.au/catchment-to-reef/index.html

- Hydrological connectivity between individual lagoons and the stream network is vital for maintenance of fish assemblages and normal waterway function.
- The special nature of the Tully-Murray wetlands as a unique assemblage of Wet Tropics habitats, with functional links to the GBR lagoon, needs to be specially recognised.

S3: Key findings of MTSRF Project 3.7.3

S3.1: Streams of the Wet Tropics – the *Catchment to Reef* program

- The *Catchment to Reef* program focused on streams of the Wet Tropics and was reported extensively elsewhere; its main findings are summarised here.
- Ambient rather than event-based water quality is of greatest importance to the ecology of Wet Tropics rivers and wetlands, in contrast to the flood events that are typically most important for contaminant exports.
- The most serious factors affecting health in Wet Tropics streams and wetlands are changes to habitats (flow modification, loss of riparian vegetation, weed invasion, infrastructure).
- Climate change is a looming issue for streams and wetlands; its effects have not been studied nor adequately elucidated but could involve sea level rise into freshwater habitats and altered hydrological regimes (e.g. reduced dry-season base flows).
- In some systems (e.g. the Herbert), organic effluents from cane fields and point sources cause fish kills and a major decrease in biodiversity as a result of oxygen depletion.
- Deposition of fine sediments derived from agricultural lands and other sources reduce biodiversity in streams.
- Dry tropics systems are impacted by similar influences which vary according to land use, flow regimes, habitat alterations, etc.
- Many Dry Tropics rivers contract to isolated lagoons in the dry season, which provide refugia for the biota and develop their own character depending on their lithology, riparian vegetation, cattle access, etc.
- Flow regime has a strong influence on aquatic refugia and on fish assemblages and populations in streams and rivers.
- The gradient of flow regime from mid Wet Tropics to Dry Tropics is very clearly reflected by biodiversity, with even the smaller rivers of the Wet Tropics supporting many more fish species than the large Dry Tropics systems of the GBR catchment.
- Damage to riverbanks and their vegetation by feral animals, farming, etc. can severely disturb habitats and biodiversity.
- The benefits of riparian vegetation to normal ecosystem function include: habitat and habitat corridors for terrestrial animals and plants; habitat for semi-aquatic animals; shade; filtration mechanisms; organic inputs; bank stability; instream habitat via roots and snags; basking sites for reptiles; and breeding and roosting sites for many partly aquatic species, ranging from insects to birds.
- The dynamics of oxygen and pH are complex and have major impacts on stream and wetland fauna; natural oxygen status can best be achieved by maintaining normal flow regimes and riparian zones; by controlling weed growth; by preventing the input of nutrients; and by removing blockages to flow.
- The tropical Australian invertebrate and fish fauna appear resistant to low dissolved oxygen status; however, their tolerance thresholds can be breached, as evidenced by the occasional fish kills that occur in floodplain waterways.

- Invertebrate diversity and fish diversity provide strong indicators of stream ecosystem health at site and subcatchment scales.
- Surrogate measures related to the style of management of land in the catchment, and the quality of the riparian vegetation, can provide rapid assessment of stream health when point sources of impact are not in evidence.

S3.2: Floodplain lagoons of the Wet Tropics – the Tully-Murray study

General points – habitats and water quality

- Many of the above points apply widely across stream and wetland systems in the GBR catchment.
- The Tully-Murray study focused on floodplain lagoon systems as potential biodiversity refugia in highly modified landscapes.
- Wetland systems in the GBR catchment have been greatly reduced in extent as a result of agricultural practice, roads and settlements.
- Nevertheless, floodplain wetlands remain a prominent landscape feature within the Tully-Murray catchment.
- These wetlands are considered very significant from an environmental perspective as they provide habitat for diverse aquatic and riparian biota in addition to potentially improving water quality delivered to rivers and to the GBR lagoon.
- These wetlands are of high value from a fishery perspective.
- There is a complex interplay between flow regime, land use, infrastructure development, habitat, water quality and contamination, weed infestation, alien fish, ecological processes and biodiversity.
- The Tully-Murray wetlands are dominated by a perennial flow and frequent flood regime, in contrast to the much more seasonal systems in the Dry Tropics.
- The Tully-Murray wetlands presented a range of lagoons, varying in size, bathymetry, surrounding vegetation and water quality, producing clear biophysical gradients in a number of parameters.
- However, these gradients were not extreme as there were apparently no pristine sites and none that were severely degraded from a water quality perspective, probably because of the regular flushing of lagoons resulting from the rainfall regime of the region.
- Strong degradation due to weed infestation and loss of riparian vegetation was evident.

Zooplankton and invertebrate assemblages

- Zooplankton was abundant but not diverse, and showed some moderate relationships with the biophysical descriptors of the lagoons, especially size variables, habitat variables (e.g. presence of alien plants) and water quality variables (e.g. transparency, dissolved oxygen and hexazinone concentrations).
- Invertebrate assemblages were diverse, and typical of tropical waterways, especially in the representation of insects and crustaceans.
- Invertebrates were strongly influenced by habitat irrespective of lagoon type or identity.
- Invertebrates showed moderate relationships with biophysical variables such as plant species richness, transparency, pH, dissolved organic nitrogen, dissolved inorganic nitrogen, dissolved oxygen, hexazinone and atrazine.
- Temporal variation in invertebrate assemblages was not strong.

- Overall, invertebrate assemblages displayed no indication of severe stress from changes in water quality, possibly because anthropogenic chemicals and associated water quality factors did not reach threshold levels that would be of concern to the biota of the lagoons.
- This study has shown that invertebrate assemblages can provide useful indications of the health of floodplain lagoons, as follows:
  - summary measures such as invertebrate diversity suggest that the Tully-Murray lagoons are not yet seriously impacted by water quality stressors associated with the surrounding agricultural land use; and
  - agricultural chemicals and associated water quality factors (e.g. dissolved oxygen) have not reached threshold levels that are of major concern to the biota of the lagoons.

Fish assemblages

- The fish fauna of lagoons included 22 species (21 native 1 alien) including six migratory species that require access to estuarine or marine areas for spawning and/or larval development.
- Total number of fish species varied little among lagoons (15-17 species) and mean species richness was marginally more variable with both summary metrics showing no clear relationship to possible anthropogenic stressors in the Tully-Murray system.
- There were strong associations between fish species and microhabitat types within lagoons. Some fish species avoided stands of the alien ponded pasture grass *Hymenachne amplexicaulis* and other alien emergent plants. *Urochloa mutica* (another ponded pasture grass) had less impact on fish assemblages.
- A single alien fish species was collected from nine of the ten Tully-Murray lagoons – the platy (*Xiphophorus maculatus*) – an aquarium escapee often found in Queensland's coastal streams. This species was most abundant in lagoons with more leaf litter.
- Fish showed moderate relationships with water quality variables, including transparency, pH, dissolved organic nitrogen, dissolved inorganic nitrogen, dissolved oxygen, conductivity, diuron, desethyl atrazine and atrazine.
- Overall, fish assemblages displayed no indication of severe stress from changes in water quality, possibly because anthropogenic chemicals and associated water quality factors did not reach threshold levels that would be of concern to fish in these lagoons.
- Coastal floodplain lagoons provide habitat for the early life history stages of a range of freshwater and estuarine fish species, including iconic species (e.g. barramundi), species of conservation significance (spotted blue-eye) and migratory species (e.g. empire gudgeon).
- Recruitment strategies were documented for ~70% of fish species in the Tully-Murray lagoons. Species were placed into one of three groups depending on the timing of their appearance in the lagoons as early life history stages.
- The first group of species, including *Neosilurus ater*, *N. hyrtlli*, *Giurus margaritacea* and *Mogurnda adspersa*, were concentrated in samples from the immediate 'post wet' season.
- The second group of species, including *Hypseleotris* sp. 1; *Ambassis agassizii* and *Denariusa bandata*, was collected in the dry season.
- The third group comprised a suite of fish species with an extended reproductive season that appeared in the lagoons as early life stages in both the wet and dry seasons.
- The extent of hydrological connectivity between floodplain lagoons and the stream network appeared to influence assemblage composition and the population structure of species among lagoons by determining the timing and duration of accessibility for

migratory species to individual lagoons. Small-bodied species are able to move through shallow streams and cane drains at water depths as low as one centimetre.

- The contrasting patterns of fish recruitment highlight the importance of maintaining the natural patterns of low flows followed by seasonal flooding and hydrological connectivity between the main river channels and lagoons on the floodplains.
- This study has shown that fish assemblages can provide robust indications of the health of floodplain lagoons, as follows:
  - summary measures such as fish species richness suggest that the Tully-Murray lagoons are not seriously impacted by water quality stressors associated with the surrounding agricultural land use;
  - agricultural chemicals and associated water quality factors (e.g. dissolved oxygen) have not reached threshold levels that would be of concern to the biota of these floodplain lagoons;
  - fish respond to the types of habitat disturbance typical of these floodplain wetlands, in particular, proliferation of alien ponded pasture grasses;
  - the presence of alien fishes is a strong indicator of initial disturbance by human activities in the broader landscape, and an early warning indicator of the potential for future disturbance from increasing numbers of individuals and species (e.g. tilapia);
  - the presence of early life history stages of several fish species is a revealing indicator of the degree of hydrological connectivity between salt- and freshwater habitats;
  - distorted patterns of fish age structure in lagoons could indicate loss of connectivity due to barriers or poor habitat/water quality in connecting channels, or altered flooding patterns due to human water use/abstraction and/or climate change; and
  - the ecological condition of floodplain lagoons may serve as a powerful indicator of climate change, because they are vulnerable to sea levels rise as well as hydrologic alterations, especially loss of seasonal flood pulses and connectivity between rivers and floodplain wetlands.

Conclusions

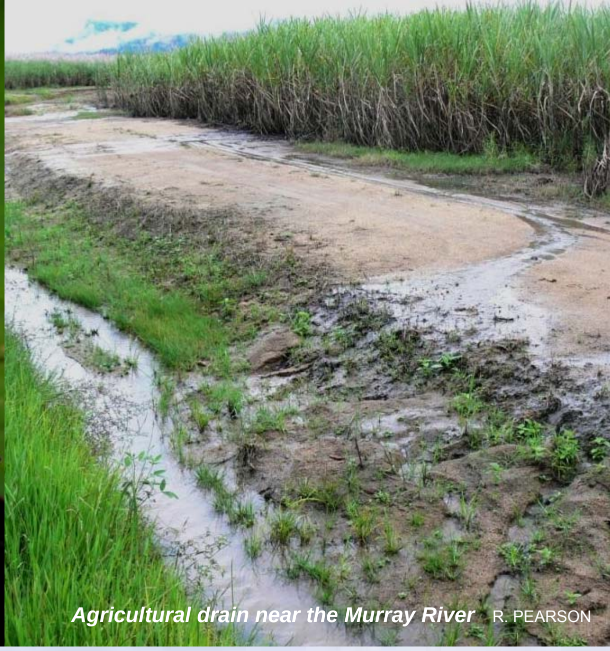
- Our results suggest that much of the aquatic fauna in the wetlands of the Tully-Murray floodplain is in good condition and resistant to the immense changes in surrounding land use.
- Results from this study provide a benchmark against which future improvement or deterioration in wetland condition can be evaluated.
- The Cassowary Coast Regional Council study, indicating that broad assessments can provide a fair reflection of ecological conditions, especially in relation to the fish fauna.
- Threshold of concern were difficult to identify from this research because of the multiplicity of variables, some of which acted in concert and some of which did not.
- However, clear responses of the aquatic biota to many of the variables that we measured indicates that some composite ecosystem health threshold has been crossed.
- Each lagoon links to other systems by the network of natural waterways and drains, and each links to its immediate surroundings and catchment through direct inputs of materials, and through surface and groundwater flow, which together determine the ecological character of the lagoon.
- Past research on catchments of the GBR has mostly focused on delivery of contaminants and sediments to marine systems.

- Little attention has been paid to the high biodiversity and iconic biota of these wetland systems in their own right, or to their potential utility in retaining contaminants through a variety of processes, especially during non-flood periods.
- Little attention has been paid to the importance of GBR catchments to migratory species; the extent of this connectivity between catchment waterways and the reef lagoon needs to be quantified. Management should pay due regard to the entire 'Greater Barrier Reef' ecosystem, as highlighted in the *Catchment to Reef* program.
- Floodplain wetlands are important because there are so few such habitats in the GBR catchment, and because they are remnants of probably much more extensive wetland systems in immediate post-Pleistocene times.
- The special nature of the Tully-Murray wetlands as a unique assemblage of Wet Tropics habitats, with functional links to the GBR lagoon, therefore needs to be specially recognised.
- Monitoring protocols are discussed with regard to explicit questions and benchmarks.
- Monitoring is best achieved via a suite of variables, especially flow, habitat and water quality variables, invertebrate and fish diversity, and occurrence of alien fish and plant species; these variables need to be investigated over spatial and temporal scales that are appropriate to the questions being addressed. Study design and sampling protocols are established in this report.
- Similar protocols apply in Dry Tropics systems, subject to modifications appropriate to the different flow regimes, as evidenced by PhD projects associated with this MTSRF project, and past research.
- A suite of future research needs is identified from this project and as a result of a stakeholder workshop held in May 2010.
- A key recommendation is that wetland research needs to be extended geographically to further validate indicators of wetland health in the broader Wet Tropics and the Dry Tropics; other wetland types also need to be investigated.
- A second key factor identified by the stakeholder workshop was the need to fully understand the dynamics and ecological roles of connectivity, both across catchments and between catchments and marine systems.

S3.3: Postgraduate projects associated with MTSRF Project 3.7.3

- 'Indicators of stream ecosystem health in Mackay-Whitsunday streams', K. Leonard, MAppSci, JCU, completed 2008.
- 'Dynamics of phytoplankton and water quality in Dry Tropics waterways', C. Preite, PhD, JCU, completed 2009.
- 'Scoping indicators of ecosystem health in floodplain lagoons', M. Ellison, Grad. Dip., JCU, completed 2009.
- 'Ecosystem dynamics in Dry Tropics waterways', M. Blanchette, PhD, JCU, completion 2011.
- 'Influence of flow seasonality on the recruitment ecology of riverine fishes from lowland Wet Tropics rivers', P. Godfrey, PhD, GU, completion 2011.
- 'Conservation planning for Wet Tropics waterways', S. Januchowski, PhD, JCU, completion 2011.


Nymphaea R. PEARSON


Agricultural drain near the Murray River R. PEARSON


Choked agricultural drain R. PEARSON


Boongaray Lagoon P. GODFREY


Kyambul Lagoon P. GODFREY

1. Introduction

1.1 Background

The Marine and Tropical Sciences Research Facility (MTSRF) is a $40 million component of the Australian Government's Commonwealth Environment Research Facilities program, managed by the Department of the Environment, Water, Heritage and the Arts. Through a consortium of fifteen research agencies, involving around three hundred scientists, the MTSRF aimed to deliver scientific solutions for the problems facing North Queensland's key environmental assets: the Great Barrier Reef (GBR) and its catchments, tropical rainforests including the Wet Tropics World Heritage Area, and Torres Strait.

In order to achieve its objectives, the MTSRF planned, funded and coordinated the highest quality, inter-disciplinary research for public good to:

- ensure the protection, conservation, sustainable use and management of the Great Barrier Reef (GBR) and its catchments, tropical rainforests including the Wet Tropics World Heritage Area (WTWHA), and Torres Strait;
- foster an understanding of the interactions of North Queensland's natural environment with the social and economic aspects of North Queensland's communities;
- support the adoption of science-based knowledge in policies and practices for ecologically sustainable management; and
- facilitate capacity building for sustainable environmental management research, in partnership with the community, environmental managers, research institutions, and industry and policy makers.

Research funding has been guided by the MTSRF Research Investment Strategy (RIS), which represents the collective view of scientific experts, business and industry leaders, government agencies, NGOs and community groups. The RIS provides a framework for the development of a suit of collaborative, multidisciplinary research projects that address aspects of priority environmental issues in North Queensland. The complete MTSRF research program (Figure 1.1) comprised five themes, with Theme 3 devoted to 'Halting and Reversing the Decline of Water Quality'.

High quality water is vital to the continuing health of the GBR, its catchments and the Wet Tropics rainforests, and for the communities, industries and ecosystems that are reliant on these natural systems. Research within Theme 3 has focused on improving understanding of how freshwater, estuarine and marine ecosystems are interlinked and influenced by terrestrial processes, land use and land management practices in the GBR catchment. The concept of a land-water continuum (see Vannote *et al.* 1980) – the *Catchment to Reef* continuum – underlies the structure of MTSRF Program 7 and informs the projects that constitute this research program.

1.1.1 MTSRF Program 7: Halting and Reversing the Decline of Water Quality

Program 7 is built around identifying robust indicators for water quality in freshwater, estuarine and marine ecosystems that will enable the development of tools to improve water quality monitoring and determine pollutant thresholds of potential concern for key ecosystems. Pollutants of particular concern are sediments, nutrients and herbicides. Improved understanding and quantification of the ecological outcomes of pollution processes and other stressors (e.g. altered river flow regimes, loss of riparian functions, and loss of habitat) are central to the objectives of Program 7 and its component projects. This program aims to develop predictive tools incorporating the likely impacts of changes in land use, management and climate on the water flow and water quality regimes and ecological dynamics in the wetlands and floodplains of catchments adjacent to the GBR. The overarching aim of projects within this program is to identify and introduce sustainable environmental targets and associated land-use and land-management practices for the catchment of the GBR.

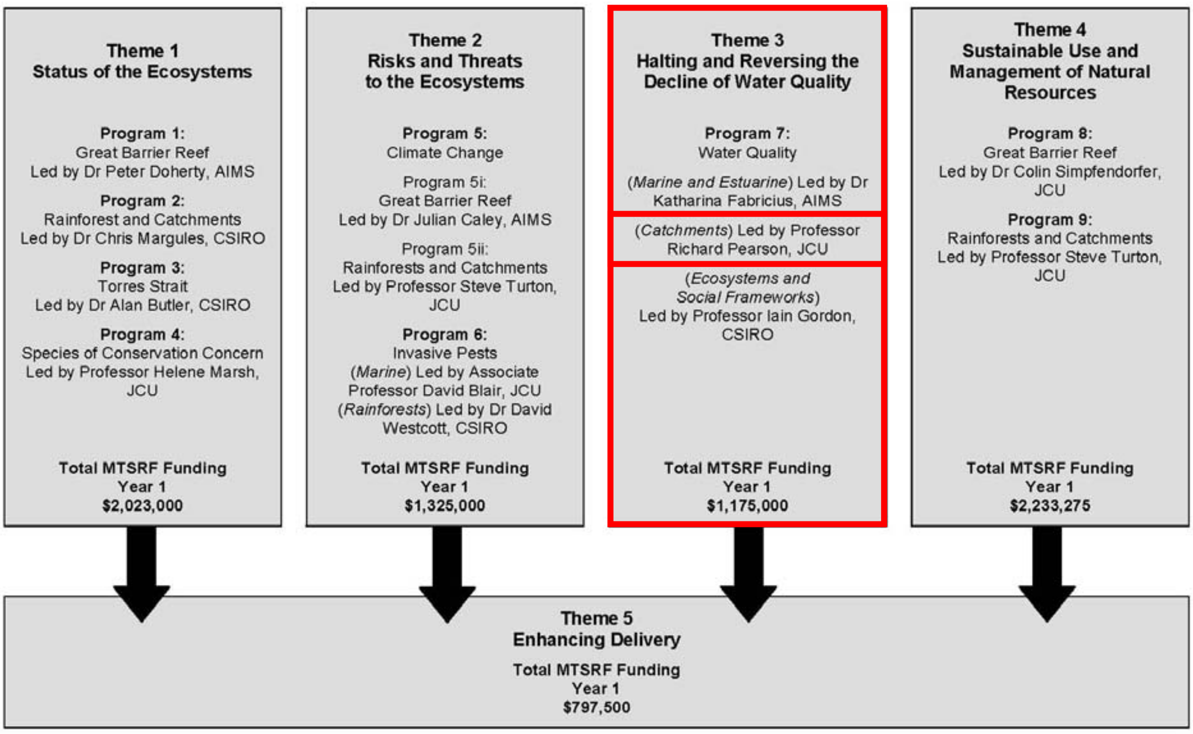


Figure 1.1: The complete MTSRF Research Program structure, showing the location of the Water Quality Program (Program 7).

1.1.2 MTSRF Project 3.7.3: Freshwater indicators and thresholds of concern

This project within Program 7 (Theme 3) sought to measure and understand the ecological processes, linkages and interdependencies that govern the condition, health, trend and resilience of rivers and their floodplain wetlands in GBR catchments, as a basis for development of cost-effective biophysical monitoring tools to help sustain these habitats and their dependent environmental assets. The project builds on research in the freshwater ecosystems of the Wet Tropics over the past twenty years, and on the former highly successful *Catchment to Reef* program (Arthington and Pearson 2007), which was completed as part of Project 3.7.3. While most past freshwater research has been focused on streams and rivers (e.g. Pusey and Kennard 1996; Pearson *et al.* 1986; Rosser and Pearson 1995; Pearson and Connolly 2000; Cheshire *et al.* 2005; Arthington and Pearson 2007; Bastian *et al.* 2008; Rayner *et al.* 2008), MTSRF research in freshwater systems has shifted to investigations of floodplain wetlands and their connectivity to rivers, as these form a distinctive and vitally important continuum of habitats and ecosystems in both the Wet and Dry Tropics.

Objectives of MTSRF Project 3.7.3 were to:

(a) Conduct field, laboratory and desktop research to develop physical, chemical and ecological indicators of freshwater ecosystem health in the Wet and Dry Tropics as part of an Integrated Water Quality Report Card that meets end-user needs and objectives. [*NB: the Report Card project did not eventuate in the MTSRF program.*]

(b) Identify thresholds of potential concern relating to land use, water quality, riparian condition, habitat and food-web structure in freshwater ecosystems of the Wet and Dry Tropics.

(c) Develop an interactive Web database documenting the distribution and ecological requirements of freshwater biota in the Wet and Dry Tropics, to assist river health assessments and inform a range of end users. [*This part of the project was not funded, but was initiated; it may be completed separately, at least for fish*.]

(d) Train new researchers via PhD programs that will be integral to the identification and testing of efficient and effective freshwater condition indicators in the Wet and Dry Tropics.

(e) Provide monitoring methods, manuals and guidelines of relevance to a range of skills and end users.

1.1.3 Project methodology

Objective (a): Conduct field and laboratory research to develop physical, chemical and ecological indicators of freshwater ecosystem health in the Wet and Dry Tropics as part of Integrated Water Quality Reporting that meets end-user needs and objectives

[*NB: Funding was not provided for work in the Dry Tropics, apart from support for a PhD project*]

Project 3.7.3 set out to develop conceptual biophysical models based on previous research to identify: (i) appropriate indicators of wetland health, and (ii) probable thresholds of concern, in terms of contaminant concentrations, ecological processes and biodiversity. Initial workshops explored methods of freshwater ecosystem health monitoring using biophysical indicators and protocols. The research team reviewed knowledge of spatial/temporal scales of indicator response to disturbance in rivers and wetlands, and identified possible spatial and temporal thresholds of concern in Wet Tropics river and floodplain systems using results from the literature and the *Catchment to Reef* river health monitoring program (Arthington and Pearson 2007), and in relation to land use, in-stream and riparian habitat and water quality, flow regime, hydrological connectivity and biodiversity. The project has workshopped conceptual models and freshwater indicators with end users to ensure that they can provide the means for various user groups to monitor and interpret pressures of particular relevance to their interests and concerns in tropical waterways (Pearson *et al.* 2010). These pressures are likely to include patterns and types of land use, general water quality and contaminants, hydrological regime, channel and habitat structure, riparian vegetation condition and alien species of plants and fish.

Climate change may intensify many of these pressures on freshwater ecosystems and their assets, and for this reason significant attention has been paid to the potential influence of hydrological changes that may reduce connectivity between rivers and floodplain wetlands. This has been achieved through close collaboration between Project 3.7.3 (this project) and Project 3.7.4 (‘Wetlands and floodplains: connectivity and hydro-ecological function’), as agreed during early consultations and formulation of the MTSRF RIS with stakeholders and agencies. The Tully-Murray catchment was selected as the focus of this collaboration between researchers from James Cook University, Griffith University and CSIRO. Floodplain wetlands are a prominent ecosystem type in the lowland catchments of the Tully and Murray Rivers, they support significant biodiversity, they provide habitat for iconic species of fish and wildlife, and they are vulnerable to changes in hydrological connectivity with the main channels of the Tully and Murray Rivers. Moreover, they provide a complex of wetlands unparalleled in the Wet Tropics, probably representing more extensive systems prior to development for agriculture and, more distantly, when sea levels were lower.

Objective (b): Identify thresholds of potential concern relating to land use, water quality, riparian condition, habitat and food web structure in the Wet and Dry Tropics

To provide theoretical and practical support to the research on thresholds outlined under Objective (a), Project 3.7.3 has continued to assess existing approaches to determining and representing thresholds of potential concern, such as water quality guidelines (e.g. ANZECC Guidelines), benchmarking methods applied in Queensland Water Resource Plans (Brizga *et al.* 2002), and other approaches used globally (e.g. in South Africa).

Project 3.7.3 set out to measure both spatial and temporal variability of biophysical indicators in floodplain wetlands of the Tully-Murray catchment along natural environmental gradients (e.g. proximity to coastal waterways and hence the potential for saline intrusions and/or fish movements between coastal and freshwater systems) and gradients of disturbance, including gradients of land use, water chemistry and habitat quality, riparian disturbance, wetland hydrology and connectivity. The major field study in the Tully-Murray catchment was designed such that stressor-response relationships along gradients of disturbance (supported by data from laboratory trials and the literature) would help to identify 'thresholds of concern', i.e. points along each disturbance gradient where ecological changes of scientific or management concern become apparent. Early recognition of such thresholds can guide management actions to alleviate the associated stressors, or may signal environmental factors that could be restored to a more natural condition to improve the health of the aquatic ecosystem (Biggs and Rogers 2003; Arthington *et al.* 2006; Poff *et al.* 2010)

Objective (c): Develop an interactive Web database documenting the distribution and ecological requirements of freshwater biota in the Wet and Dry Tropics, to assist river health assessments and inform a range of end users

It was intended that results of this project and the previous *Catchment to Reef* program would be incorporated into an interactive atlas that would be available to all interested parties. The rudiments of the atlas were developed, but funding was not available to develop it fully. Nevertheless, work has continued independently on the development of a freshwater fish atlas building on work done in the *Catchment to Reef* program; and the MTSRF has progressed a reef-based atlas[2] that could yet include catchment data.

Objective (d): Train new researchers via PhD programs that will be integral to the identification and testing of efficient and effective freshwater condition indicators in the Wet and Dry Tropics

The intention was that, subject to supplementary funding/co-investment, Project 3.7.3 would progress towards testing relationships, indicators and models in the Dry Tropics, following a similar research plan to that followed in the Wet Tropics. Funding through MTSRF or external partners was not available to properly pursue this aim; however, some headway has been made through the affiliated program of postgraduate research. MTSRF has funded some of this PhD research. Projects have been crafted to test organism-level indicators of fish, invertebrate and plant health, as well as ecological processes, by measuring response levels, rates and times, and the efficacy of measurement using field surveys and laboratory work. Supervision has been shared between JCU and GU. The following projects are completed or in progress:

- 'Indicators of stream ecosystem health in Mackay-Whitsunday streams', K. Leonard, MAppSci, JCU, completed 2008.

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[2] http://e-atlas.org.au/

- 'Dynamics of phytoplankton and water quality in Dry Tropics waterways', C. Preite, PhD, JCU, completed 2009.
- 'Scoping indicators of ecosystem health in floodplain lagoons', M. Ellison, Grad. Dip., JCU, completed 2009.
- 'Ecosystem dynamics in Dry Tropics waterways', M. Blanchette, PhD, JCU, completion 2011.
- 'Influence of flow seasonality on the recruitment ecology of riverine fishes from lowland Wet Tropics rivers', Paul Godfrey, PhD, GU, completion 2011.
- 'Conservation planning for Wet Tropics waterways', S. Januchowski, PhD, JCU, completion 2011.

Objective (e): Provide monitoring methods, manuals and guidelines of relevance to a range of skills and end users

Consultation with interested parties has guided the development of the types of outputs needed. Most recently, the stakeholder workshop held towards the end of the program was very clear about the types of products required (Pearson *et al.* 2010). The focus was more on system understanding than monitoring methods, although it is obvious that appropriate system monitoring requires proper understanding. Two types of output were recommended: (i) peer-reviewed journal papers to establish research results in the scientific literature; and (ii) very concise summaries of research results and their bearing on management, designed to be accessible to a wide range of readers. While the research can well support development of monitoring protocols, this report focuses more on these two recommendations.

1.2 Project 3.7.3 context

Projected outcomes are expected to be: (i) enhanced understanding of ecosystem health and its assessment in waterways of the Wet Tropics and, to a lesser extent, the Dry Tropics; (ii) improved knowledge in the community of the importance of ecosystem health in Wet and Dry Tropics waterways; and (iii) cost-effective biophysical indicators and tools for monitoring the condition of floodplain wetlands, applicable also to tracking the effects and benefits of catchment management and restoration activities, such as containment of runoff form developed lands, riparian restoration, environmental flows and restoration of floodplain connectivity.

1.2.1 Outcomes of the *Catchment to Reef* program

Our understanding of the ecosystem health of GBR waterways has been greatly enhanced by recent reports generated through the CRC *Catchment to Reef* Program and the MTSRF Program on Wet Tropics streams (e.g. Arthington and Pearson 2007; Pearson and Stork 2007; Connolly *et al.* 2007a 2007b; Pusey *et al.* 2007b) and floodplain waterways (Pearson *et al.* 2003), and on the riverine waterholes and floodplains of the Dry Tropics (e.g. Perna and Burrows 2005; Preite 2009; Blanchette pers. comm.). The *Catchment to Reef* project, which focused on Wet Tropics streams, commenced under the joint Rainforest and Reef CRC program, and was completed in Year 1 of the MTSRF program. A substantial report was produced (Arthington and Pearson 2007) so only a summary is presented here.

Summary of the impacts of degraded water quality on GBR catchment and instream ecosystems

- Ambient or chronic water quality is of greatest importance to the ecology of the rivers and wetlands, as opposed to the short-term events that appear to drive water quality in coastal waters. Flood events are typically most important with regard to contaminant exports, while the intervening periods are more important to the ecology of waterways.
- The most serious factors currently affecting health in Wet Tropics streams and wetlands are changes to habitats, including flow modification, invasion by alien weeds and loss of riparian vegetation, which can cause major changes to waterway morphology, habitat complexity, food availability, gas exchange with the atmosphere and, therefore, biodiversity.
- Organic effluents have been shown to cause fish kills and a major decrease in biodiversity as a result of oxygen depletion; and deposition of fine sediments derived from agriculture and other sources reduce biodiversity in streams.
- Dry tropics streams and wetlands are impacted by similar influences but, because of a dominance of cattle grazing, are generally more exposed to issues related to sedimentation. Many Dry Tropics rivers cease to flow in the dry season, contracting to isolated lagoons, which provide refugia for the biota. These lagoons develop their own character depending on their lithology, riparian vegetation, cattle access, etc.
- Flow regime has a high influence on fish populations. For example, P. Godfrey has demonstrated (unpub. data) that the relationship between the structure and dynamics of the larval fish assemblage in Wet Tropics rivers and the variability of the habitat and its condition are shaped primarily by the flow regime.
- The gradient of flow regime from the Wet Tropics to the Dry Tropics is very clearly reflected by their biodiversity, with even the smaller perennial rivers of the Wet Tropics supporting many more fish species than the large Dry Tropics systems of the GBR catchment.
- The benefits of riparian vegetation to normal ecosystem function are now well documented (e.g. Pusey and Arthington 2003) and include: habitat and habitat corridors for terrestrial animals and plants; habitat for semi-aquatic animals; shade; filtration mechanisms; organic inputs; bank stability; instream habitat via roots and snags; basking sites for reptiles; and breeding and roosting sites for many semi-aquatic species, ranging from insects to birds.
- The dynamics of oxygen and pH in catchment waterways are complex and dependent on a range of natural and human-influenced variables (Pearson *et al.* 2003). Natural oxygen status can best be achieved by maintaining normal flow regimes and riparian zones, by curtailing weed growth, by preventing the input of nutrients, and by removing blockages to flow.
- While the tropical Australian invertebrate and fish fauna appear extremely resilient to low dissolved oxygen status (Pearson *et al*. 2003; Connolly *et al*. 2004), their tolerance thresholds can be breached, as evidenced by the occasional fish kills that occur in floodplain waterways.
- Prolonged high sediment levels reduce diversity and abundance of stream biota such as invertebrates (Connolly and Pearson 2007) and fishes (Hortle and Pearson 1990).

1.2.2 Floodplain wetlands in the GBR catchment

Floodplains and wetlands are important physical and biological components and links in the aquatic continuum, providing unique and essential habitat and connectivity for specialist and wide-ranging biota. Floodplains support high biodiversity and provide highly valued ecological goods and services – water supplies, artisanal and commercial fisheries, aquaculture, floodplain recession agriculture and pastoral animal husbandry (Postel and Richter 2003; Welcomme *et al.* 2006; Sala *et al.* 2008). In addition, various regulating and cultural services (flood control, sediment/nutrient trapping, recreation) are important to human welfare and livelihoods in many parts of the world (Millennium Ecosystem Assessment 2005; Rodriguez *et al.*; 2007). These rich resources and opportunities have attracted human use and occupancy of floodplains, often to the detriment of the very processes that generate ecosystem benefits (Welcomme *et al.* 2006). Floodplain rivers are particularly vulnerable to flow regime alteration, loss of connectivity and other stresses associated with human occupancy and use of catchment resources (Tockner *et al.* 2008).

In the GBR catchments, floodplain wetlands provide vital habitat for iconic species such as fish and water birds, and they may perform an important water retention and filtering function (Hogan and Graham 1994, Vallance and Hogan 2004). However, they are typically badly managed, highly impacted and severely depleted (Tait, undated; Arthington *et al.* 1997; Pusey *et al.* 2004; 2007a). One estimate suggests that ~75% of such wetlands in GBR catchments have been lost to agricultural and other developments (Queensland Government 2003), and although the scale of this loss has since been questioned (M. Ronan, pers. comm.), it is clear that substantial modification and loss of habitat has occurred over the last 100 years or so. Some evidence suggests that the wetlands of the Wet Tropics lowlands may still be in good ecological condition, with the possibility that the many man-made drainage channels (e.g. cane drains) provide habitat for a range of aquatic species, and may compensate for the loss or degradation of natural wetland habitats in these developed catchments. However, previous work on the Herbert floodplain has indicated severe impacts of agriculture on the waterways (Pearson *et al.* 2003).

Proper management of floodplain wetlands depends on understanding the biophysical relationships, dependencies and hydrological connectivity inherent in these systems (Paterson and Whitfield 2000; Arthington *et al.* 2005; 2007; 2010; Naiman *et al.* 2008). It is particularly important to develop a better and more predictive capacity to quantitatively link changes in land use, land management, water management and even climate change to freshwater and wetland water quality and ecological health.

1.2.3 Water quality

Water quality and the ecological impacts of degraded water quality in GBR freshwater systems are complex issues. Water quality variables do not act alone and the actual impact of a contaminant on ecological systems and biota is sometimes difficult to predict. For example, hypoxia that can result from nutrient enrichment (cultural eutrophication) might not occur if the water is also very turbid and hence likely to limit phytoplankton growth (Pearson and Connolly 2000). 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 and wetlands that act as delivery pathways. On the other hand, smaller but chronic inputs of contaminants will 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, but 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.

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 (Downes *et al.* 2002).

Monitoring water quality at a catchment scale is also a complex issue. Values for water quality variables are affected by many factors, all inter-related to a greater or lesser extent. Figure 1.2 summarises some major variables and interactions affecting ambient water quality in Australian tropical fresh waters. The diagram is not exhaustive in its coverage, but serves to demonstrate the inter-related nature of water quality processes and measures and their influence on the ecological system. It also indicates that these processes occur as part of a continuum from the catchment to the sea, determined by the flow regime of the system. In perennial streams and rivers the inputs and outputs of surface waters are continuous, whereas in floodplain lagoons and riverine waterholes in the Dry Tropics in particular, surface inputs and downstream outputs may be local or non-existent.

While event-focused monitoring aims to identify the types and qualities of materials being transported to the marine environment during a short residence time in the stream, ambient monitoring is concerned with the quality of water that is resident in the stream or wetland for extended periods of low or zero flow. The short-term nature of large events means that water quality during the event is less of an issue in inland waters, where the power of floods in removing plants and animals and the connectivity they create between waterways and wetlands are the dominant considerations. The long-term exposure of the biota to water quality characteristics during low flow periods means that water quality, at a local scale, becomes a predominant issue in Dry Tropics streams and wetlands, and in some of the smaller and/or more peripheral streams and wetlands of the Wet Tropics.

1.2.4 Habitat quality

One of the major structuring influences in ecological systems is the nature of the habitat (Southwood 1977; Townsend and Hildrew 1994). As was shown in the *Catchment to Reef* program, a change in gradient in a stream can cause major change in substratum: typically, as we move down a catchment, stream substrata follow a distinct gradient from coarse to fine with corresponding changes in the biota (Arthington and Pearson 2007; Connolly *et al*. 2007a, 2007b). For example, rooted vegetation is more able to establish in siltier environments; invertebrates range from clingers and swimmers in upper reaches to burrowers in fine substrata downstream; and fish species maybe variously adapted to stony riffles, slow deep pools, beds of vegetation, etc.

Human influences can substantially alter habitats, and thereby affect the biota and ecosystem health. Most drastic is complete removal of habitats, for example by filling wetlands. Other major effects include flow modification (by dams, weirs, and water harvesting), removal of riparian vegetation, and all the roles it plays, sand dredging, sedimentation via land and bank erosion, invasion by alien plants (e.g. para grass), reduction of connectivity caused by weed masses, culverts, flow regulation structures, etc.

In determining ecosystem health it is therefore necessary to examine the separate and collective effects of habitat and water quality variables (as well as non-habitat-related effects of alien plants and animals, such as tilapia). In some cases, cleaning up of water quality may have only small influence on ecosystem health if the habitat is severely damaged.

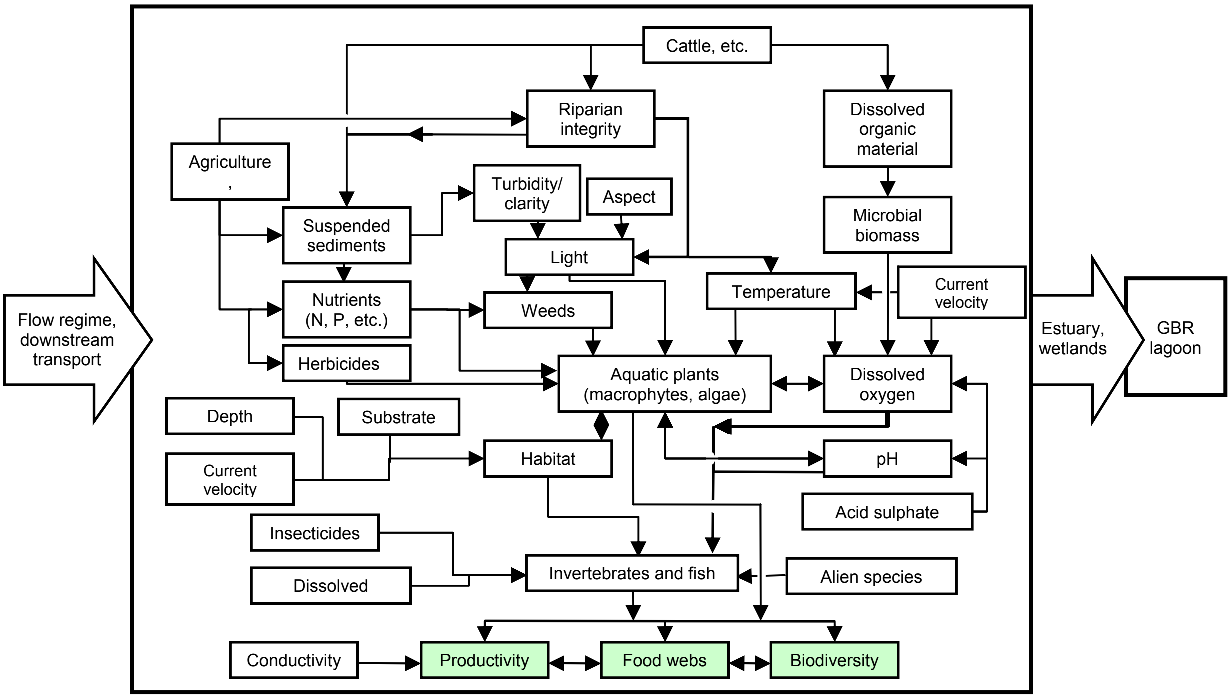

Figure 1.2: Biophysical interactions affecting ambient water quality in tropical agricultural landscapes. The large box represents typical processes and interactions in a stream reach, a discrete waterhole or a habitat within them. The large arrows represent flow-related connectivity. The shaded boxes represent different types of ecological response. Connectivity with the terrestrial landscape is implicit in some of the smaller boxes. Not all factors or interactions can be shown – for example, riparian integrity has a number of influences on ecological responses that are not indicated here; and there is no indication of the influences of urban infrastructure, mines, etc. [R. Pearson; from Brodie *et al.* 2009]

1.2.5 River and wetland health

Concerns for the ecological condition or 'health' of rivers and wetlands have been at the forefront of recent conservation and environmental movements across Australia and in other countries (Bunn and Arthington 2002; Dudgeon *et al.* 2006; Sala *et al.* 2008). 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; Bunn *et al.* 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 resist or 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 (Baron *et al.* 2002; Downes *et al.* 2002).

Thus, the concept of waterway health considers not only the structural integrity of river and wetland ecosystems, but also functional aspects such as the resilience of the system (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 (Figure 1.3). 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 *et al.* 1999; Arthington *et al.* 2007).

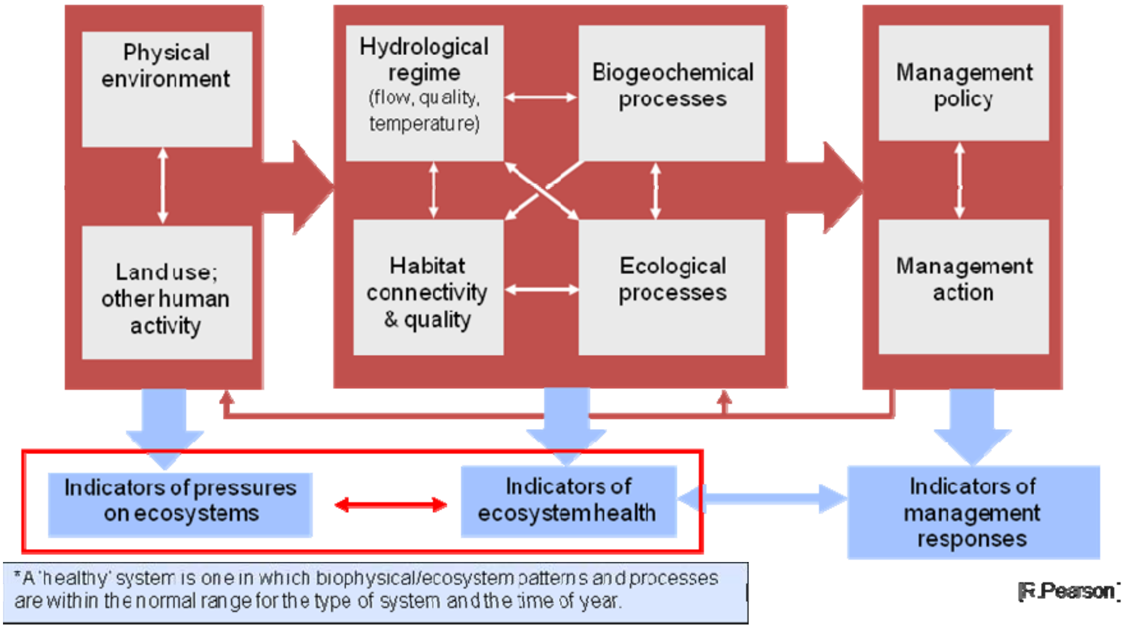

Figure 1.3: Conceptual model of a freshwater system and the links between the different drivers and processes.

1.2.6 Approaches to freshwater ecosystem health assessment

Several multivariate techniques comparing assemblages across sites have been used to assess stream health, 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 1996; 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; Usseglio-Polatera *et al.* 2000). Habitat and water quality are also evaluated using a series of measures and indices (Barbour *et al.* 1999; ANZECC and ARMCANZ 2000) and ecosystem processes such as photosynthesis and respiration are receiving some interest (Bunn *et al.* 1999; Bunn and Davies 2000).

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 (e.g. Pearson and Penridge 1987). 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 in the Wet Tropics reported by Arthington and Pearson (2007) and Pearson and Stork (2007).

1.2.7 Freshwater indicators

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.* 2007b; Kennard *et al.* 2006a, 2006b). Therefore, contemporary assessments of river health incorporate both physico-chemical measures and measures of ecological integrity. Habitat integrity, riparian condition and levels of weed infestation can all have a major bearing on aquatic ecosystem health, as shown in previous research on Wet Tropics streams (Werren and Arthington 2002; Arthington and Pearson 2007; Connolly *et al.* 2007a, 2007b; Mackay *et al.* 2010), and this may be the case for coastal wetlands in the GBR catchment, such as the Tully-Murray wetlands.

Different indicators can be expected to reveal different aspects of stream health but a comprehensive monitoring protocol does require the incorporation of a suite of 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 pressure and known disturbances. Research detailing the chain of influence from land use to stream ecosystem response, via the responses of individual ecosystem components, is required to understand how these influences operate and to underpin the development of monitoring tools and guidelines appropriate to Wet Tropics streams and wetlands.

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, invertebrates, fish and frogs. In addition, indicators of ecosystem processes such as benthic metabolism have been developed and applied in streams and floodplain wetlands (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).

Potential indicators of wetland ecosystem health were sought from a list of candidate metrics established during prior research in the Wet Tropics and broader region, in particular the *Catchment to Reef* program (Arthington and Pearson 2007). Potential indicators include:

- flow regime of the waterway;
- connectivity between rivers and floodplain wetlands;
- physical condition and habitat structure of sites;
- major water quality characteristics;
- riparian condition (vegetation structure, canopy cover, weediness);
- aquatic macrophyte cover and species richness;
- proportion of aquatic macrophyte species that are alien;
- species and/or family richness of invertebrates;
- species richness and assemblage composition of fishes;
- number and proportion of alien fish species; and
- proportional contribution of alien species to total fish abundance.

1.2.8 Floodplain wetlands

Very little is known about the hydrological dynamics of floodplain wetland systems in the GBR catchment, and how hydrological dynamics and physical connectivity influence aquatic habitats, water quality, biological diversity and ecosystem processes (Hogan and Graham 1994; Vallance and Hogan 2004; Perna and Burrows 2005; Pusey *et al.* 2007b; Rayner *et al.* 2008). Proper management will depend on understanding the biophysical relationships and connectivity spatial/temporal patterns in these systems. It is particularly important to develop a better and more predictive capacity to quantitatively link changes in land use, land management, water management or climate change to freshwater and marine ecosystem health (Junk *et al.* 1989; Junk and Wantzen 2004; Robbins *et al.* 2005). Conceptually this is mediated through the hydrological regime, where both the flow and quality of water can affect habitat and the biogeochemical and ecological processes in rivers, wetlands and estuaries (see Figure 1.3).

A particular challenge lies in the quantification of the hydrological and related ecological processes that relate to wetland ecosystem health, so Project 3.7.3 has developed a strong collaboration with MTSRF Project 3.7.4 (‘Wetlands and floodplains: connectivity and hydro-ecological function'). Project 3.7.4 is focused on the floodplain hydrological regime, aiming to develop a core floodplain hydrological model to quantify two important aspects of hydro-ecological functioning, (i) sources, sinks and transport of sediments and nutrients across floodplains, and (ii) connectivity of wetland systems within floodplains. The twin development of conceptual models of the ecological dynamics of these systems and how these interact with hydrological processes was strategically designed to improve our capability to predict the impacts of changes in land use, land and water management and climate on the flow and water quality regimes and ecological dynamics in the wetlands and floodplains of catchments adjacent to the GBR.

Given these challenges, project leaders of the MTSRF Projects 3.7.3 and 3.7.4 held a joint workshop on hydro-ecological modelling within their first year of activity (April 19-20, 2007), with about thirty hydrological and ecological experts attending (Wallace *et al*. 2007). The workshop identified three major freshwater habitats other than streams and rivers that occur on floodplains: (i) remnant lagoons, (ii) distributaries, and (iii) flooded areas. These habitats are not in a natural state (due to clearing, drainage, irrigation, fertilizer and other chemical inputs). The following text presents a summary of our knowledge of wetland function in the Wet and Dry Tropics as presented at the workshop.

Wet Tropics

In the floodplain and freshwater wetland zone, over-bank floods move out onto the floodplain or back to the channel, or they may flow out to the ocean via distributary channels, or groundwater flow may connect to and feed 'wonky holes' (off-shore benthic inflows of fresh water). Off-stream habitats do not necessarily link back into the river via connectivity pathways; instead, water may flow towards the coast via distributaries after it overtops river banks, depending on bank elevation, slopes and levees, etc. During these processes of water movement, physical and water quality barriers to the movements of biota may be created.

Isolated pools on the floodplain may or may not flood when the river overtops, depending on flood discharge and the spatial scale of flood. Wetland pools are left after floods, and these may be sustained by rainfall, inflow from small catchments and groundwater. Distributaries may be fed also by small local catchments. The retention time of water in these floodplain areas has changed significantly due to changes in connectivity, related to land use/land cover change and construction on the flood plain, and this must have flow-on effects on water quality, habitat suitability, ecological patterns and processes.

The drivers of primary productivity and trophic structure in floodplain water bodies are important to their capacity to sustain aquatic biota during non-flood periods. These processes are expected to differ between wet and dry periods. What are the roles of terrestrial organic matter and carbon sources in floodplain water bodies? Are they sustained by local productivity from algae (e.g. benthic algae, phytoplankton) and/or aquatic plants? A key question for both MTSRF projects is the role of floodplain wetlands as habitat for aquatic biota. Are these water bodies the main nursery areas for fish, prawns, etc. in the Wet Tropics? Such areas are important in Northern Territory and Gulf rivers. Are they 'sinks' in many cases – particularly when adverse conditions lead to fish kills or when lagoons dry out? This may not be so common in the Wet Tropics but remains a possibility depending upon patterns and pathways of water movement, retention times and groundwater influences.

With regard to life-history and recruitment processes, the workshop discussed how the timing of spawning and/or recruitment relate to flood timing and extent and hence the availability of floodplain habitats. It was suggested that at present we really have little or no idea what is happening in these habitats, including the effect of surrounding land uses and land management on their ecology. It was remarked that the Tully River may experience three to four overbank flood events per year. Is this more frequent than in other wet tropical rivers? If so, what are the ecological consequences of several flood events that may or may not inundate and connect the river and the full array of floodplain wetlands? Some may receive one flood event, some all flood events, yet others, no flood events in some years. In the Tully it is thought that groundwater sustains floodplain wetlands during the dry season, with overland flow 'topping up' in the wet season – we need to confirm this and other aspects of water regimes and their influence on the river channel and floodplain wetlands.

Dry Tropics

Floods (and so over-bank flows) do not happen every year in the Dry Tropics, so replenishment of floodplain wetlands and connectivity between them is unpredictable. Wetlands therefore may be either intermittent, or permanent if they receive sufficient groundwater input. Flood waters typically move through a series of lagoons and distributary channels, reaching the coastal environment via distributaries that discharge through saline wetlands (e.g. Barratta Creek on the Burdekin floodplain). However, in some floodplains, wetlands connect in both directions – filling from and emptying back into the main river system (e.g. in the Queensland Fitzroy River).

Antecedent flood conditions therefore have important influences on productivity, connectivity and ecosystem dynamics. In the floodplains, which have been extensively developed for agriculture, loss of wetlands has led to a reduction of water retention, with likely flow-on effects on ecological patterns and processes. Development of floodplains has led to new barriers to connectivity, such as drop boards that control water levels, weed infestations, and patches of poor water quality.

When well connected, wetlands provide important fish habitat – for example, for juvenile barramundi. It is unclear whether fish actively seek out these habitats, or whether they use them opportunistically. It is also unclear to what extent wetlands act as sinks for organic matter, including fish.

Dry tropical systems have different rates of change in processes from those in the Wet Tropics: for example, the length of time between wet and dry periods varies enormously – in the Dry Tropics the interval may be up to 5 years. Productivity in riverine lagoons is strongly influenced by local features such as size and bathymetry of the water body, and landscape processes immediately contiguous with the water hole.

Sources of productivity include algae in the shallows, extensive macrophyte beds in deeper water and phytoplankton in open waters. Inputs from the riparian vegetation can be important in smaller water holes, although litter from the most abundant riparian trees (e.g. *Melaleuca* species) breaks down very slowly. Many fish take advantage of terrestrial fruit and insects, derived from the riparian vegetation (Pusey *et al.* 2004).

Various studies are available on invertebrates and fishes of rivers of the floodplains of the drier zones, such as Pusey *et al.* (1998) on Burdekin River fish and Pearson *et al.* (1991) on the invertebrates and fish of the Burdekin floodplain.

The following knowledge gaps were identified:

1. A major gap exists in our ability to quantify flow regime and water quality impacts on ecological processes. Some empirical approaches exist to relate flow regimes to freshwater ecology (e.g. habitat simulation models); however, these presently take no account of water quality. A major effort is therefore needed to explore the effects of acute and chronic expose of important freshwater biota to key pollutants.

2. Within the floodplain, river channel ecology is less well understood than it is in upper catchment streams. Some of the concepts developed in upper catchment streams therefore need tested on the floodplain. This testing will inevitably involve the collection of new freshwater ecological data in floodplain river channels. For example, the life history of fish, their recruitment processes and floodplain habitats and how these relate to the timing, size and duration of floods will need to be explored.
3. The role of terrestrial organic matter in freshwater productivity is currently poorly understood except in forest streams of the Wet Tropics (e.g. Bastian *et al.* 2008). It will therefore be necessary to include studies of carbon and detritus fluxes in rivers and floodplains as well as the bioavailability of the various carbon and nutrient forms.
4. Much more information on the role of wetlands is needed to clarify their potential role as pollutant filters and freshwater habitats. For example, to predict the sediment and nutrient filtering capacity of wetlands, their capacity and the residence time of water in them needs to be known. More data is required on the filtering capacity of rehabilitated and artificial wetlands as these are seen as potential ways of cleaning up polluted water before it enters the marine environment. The biological role of wetlands also needs further study: for example, are they a sink or source of organic matter? Do they provide juvenile habitats for fish?
5. Much of the above study would focus on the collection and interpretation of contemporary hydrological and ecological data. There is, however, a need also to look at the longer term hydro-geological background, in order to separate anthropogenically induced change from natural change. Progress in this area could be made by characterising historical wet and dry periods using long-term river flow records and proxy data such as sediment cores and coral growth rings.
6. The role of groundwater in riverine and wetland hydrology needs to be taken into account as does its role in sustaining low-flow habitats.
7. Estuarine ecology is poorly understood in tropical systems. It would be useful to establish a project to review the literature to focus future studies in this area. Ultimately, it will also be necessary to connect estuarine models with floodplain models, so that the final fluxes of water and contaminants to the ocean can be specified.

1.3 Structure of this report

In the following technical summary we present a brief description of the Tully-Murray wetlands and the rationale and approach of this study. We then summarise the results of our investigations: habitat and water quality; zooplankton; invertebrates; fish; and connectivity. We conclude by discussing the results in relation to the aims of the study.

Barrett's Lagoon R. PEARSON

Barrett's Lagoon R. PEARSON

2. Methods

2.1 Study area (contributed by Dr J. Wallace)

The Tully and Murray (hereafter the Tully-Murray) catchments are located towards the centre of the Queensland Wet Tropics bioregion (Figure 2.1). The topography of the catchments varies from steep rainforest-covered mountains in the west, to a low-relief floodplain mostly developed for agriculture in the centre, with further remnant forest near the coast. The combined area of the Tully-Murray catchment is 2,072 km^2, of which forty percent (832 km^2) is floodplain (Karim and Wallace 2008). The agricultural areas are largely confined to the floodplain with sugar cane occupying the largest area (36,700 ha), followed by grazing (14,900 ha), forestry (10,300 ha) and bananas (7,900 ha) (see Armour *et al.* 2009).

The mean annual rainfall is 3,316 mm, but this can vary between 2,000 and 4,000 mm, depending on the location in the catchment. There is strong seasonality in rainfall (Figure 2.2a), with most (80%) occurring during the wet season from December to May. The Tully and Murray Rivers are the two main gauged waterways in this area that discharge into the GBR lagoon. Gauged mean daily discharge and stage height data for the Tully River at Euramo (station 113006A, Figure 2.1) and the Murray River at Upper Murray (station 114001A, Figure 2.1) were obtained from the Queensland Department of Environment and Resource Management (DERM).

The Tully-Murray floodplain in inundated frequently, with the river going overbank three to four times a year on average (Wallace *et al.* 2009b). Figure 2.3 shows a time series of flow at the Euramo gauge on the Tully River and this confirms that this river floods virtually every year and sometimes as many as ten times in a year. As the Tully-Murray floodplain is very flat and the rivers are quite close, water from the two rivers often merges during a flood forming one large inundated area.

Flooding normally only happens in the wet season with the largest flood flows occurring in January (Figure 2.2b), when several km^3 of water can inundate the floodplain (Wallace *et al.* 2009b). However, the number of floods reaches a maximum later in the year around March (Figure 2.2c), coincident with the maximum monthly rainfall (Figure 2.2a). At this time there is an average of nearly one flood per month, whereas the frequency is about half of this in January and April. There is an average of ten days each year when freshwater biota can exploit the opportunities associated with floodplain inundation.

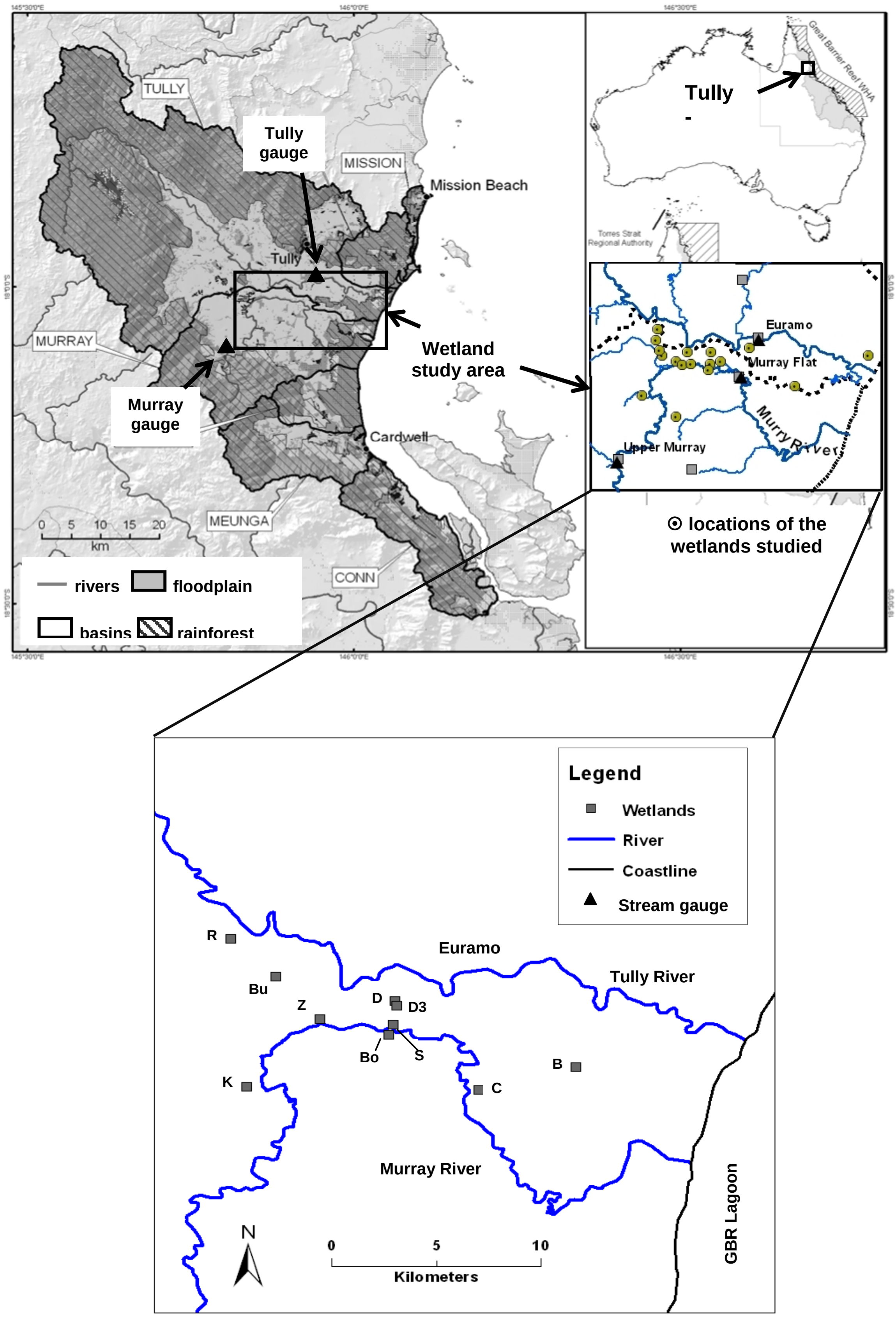

Figure 2.1: Schematic map of the Tully-Murray catchment in northern Queensland showing the locations of the floodplain wetlands and the gauging stations on the Tully and Murray Rivers. Base map reproduced from Kroon (2009) and modified by F. Karim. Lagoons: B: Barrett's; Bo: Boongaray, Bu: Bunta; C: Carroll's; D: Digman's; D3: Digman's No. 3; K: Kyambul; R: Raccanello's; S: Selby's; and Z: Zamora's.

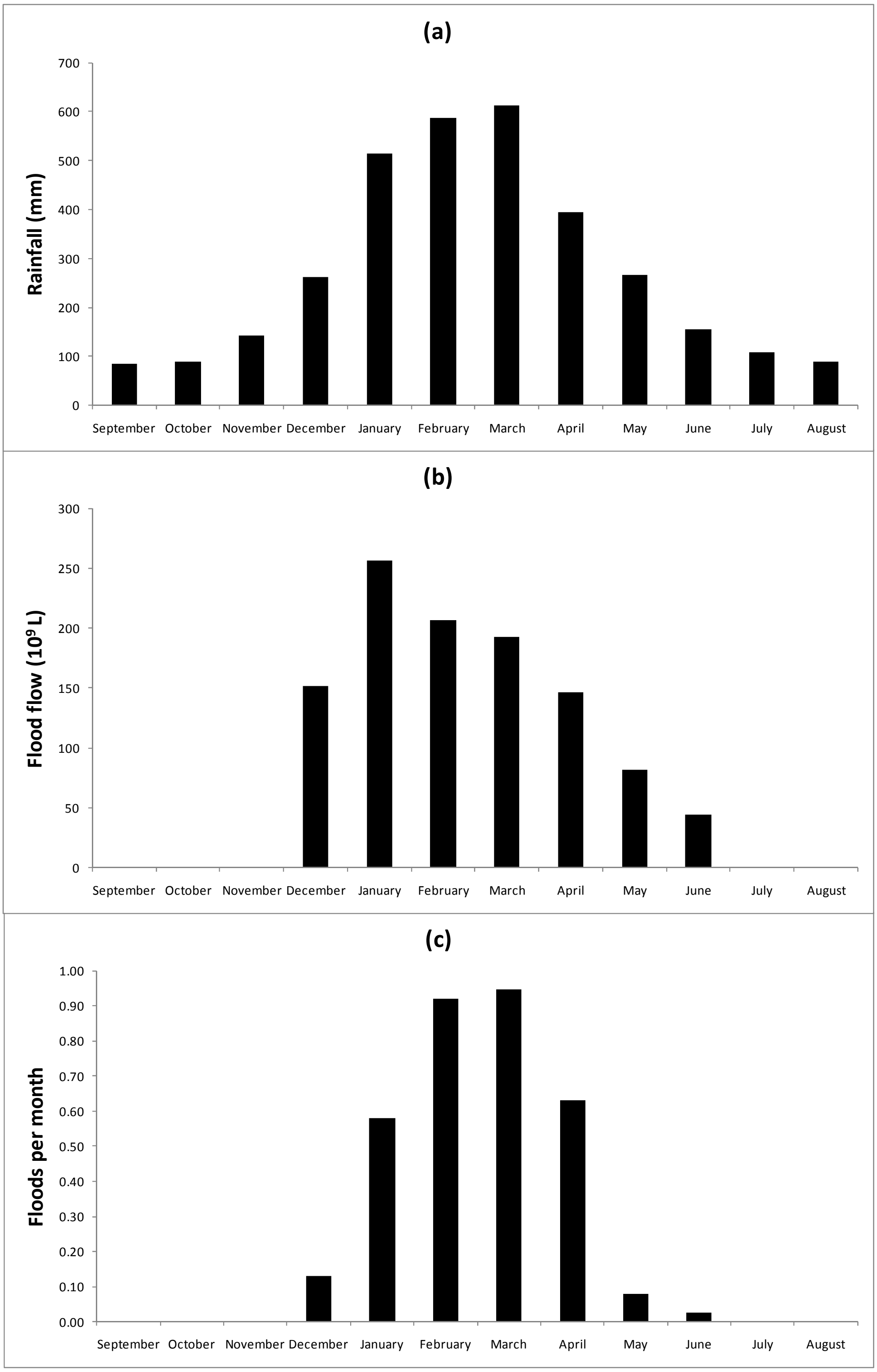

Figure 2.2: Seasonal variations in monthly average (a) rainfall, (b) flood flow and (c) number of floods.

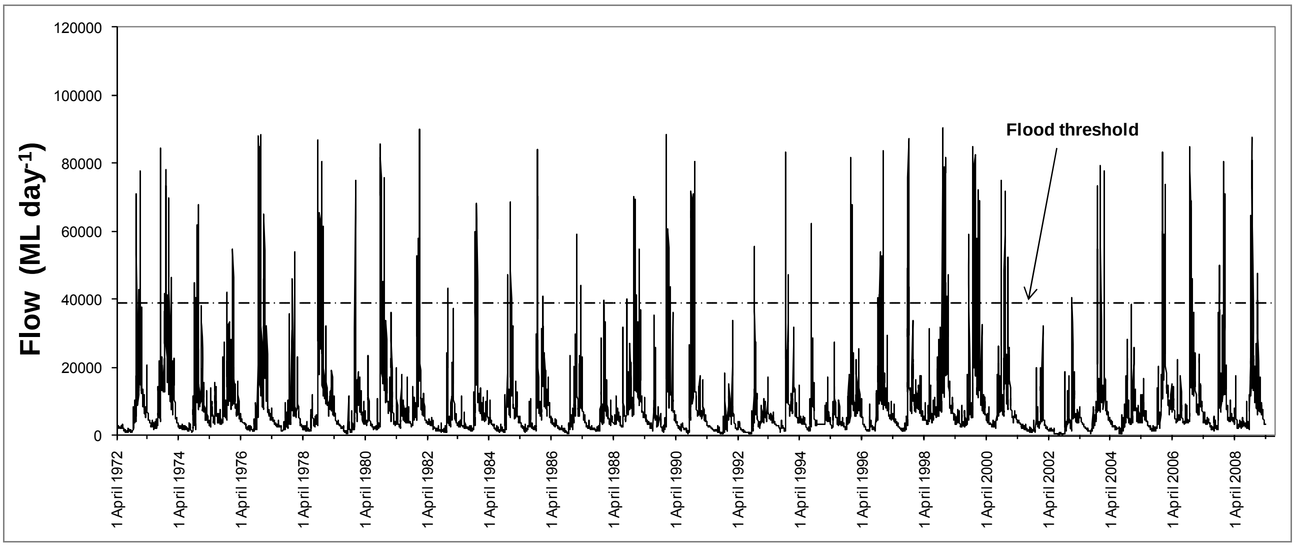

Figure 2.3: Time series of flow at the Euramo gauge on the Tully River showing the frequency with which the flood threshold is exceeded.

2.2 Study approach

Following our study of stream environments (Arthington and Pearson 2007) we focused on floodplain lagoon systems, because of their potential importance as freshwater habitat, as links from catchment to reef lagoon environments, and because of their likely vulnerability to disturbance. Previous work in the Herbert floodplain (Pearson *et al*. 2003) indicated that these systems are very susceptible to water quality issues derived from input of organic material and nutrients from cane fields, leading to excessive growth of invasive weeds and frequent problems with hypoxia.

It was expected that it would be difficult to find good reference sites (sites in pristine condition and comparable with presumed impacted sites) because virtually all the floodplain was developed for agriculture. Therefore, we sought to determine possible gradients in condition, including natural gradients, as we had done for Wet Tropics streams (Arthington and Pearson 2007). However, this was difficult because of the low number of accessible sites available and because of the complexities of the floodplain.

The main field study aimed to measure spatial and temporal variability of biophysical variables in floodplain wetlands of the Tully-Murray catchment along two sets of gradients: (i) natural environmental gradients, including position in the floodplain landscape, wetland size and morphology, hydrology and connectivity, and (ii) gradients of disturbance, including gradients of land use, hydrology and connectivity, water chemistry and habitat quality, riparian disturbance, and alien species.

Although every effort was made to establish strong disturbance gradients, ranging from near pristine to highly degraded lagoons, we were unable to identify simple gradient. Given the lack of high quality reference conditions within either catchment, and the mixture of different disturbances across the two catchments, the field design was therefore treated as a multivariate case study with the potential to reveal some gradients but a greater likelihood of discovering multiple interacting gradients that were not necessarily co-varying. This multivariate study design can nevertheless inform conceptual models and help to validate indicator variables.

2.2.1 Sampling design

The condition of aquatic and riparian habitats, water chemistry and biota were measured in ten floodplain lagoons between May 2008 and May 2009 (Figure 2.1). All ten lagoons were sampled in May and September 2008 and May 2009 to capture the response of biota to the seasonal changes in water chemistry, hydrologic conditions and the extent of connectivity between the stream network and the floodplain lagoons (Table 2.1). In addition, five lagoons were sampled in July 2008 and four in November 2008 to gather a more comprehensive dataset and to enhance our understanding of the temporal dynamics of the biotic assemblage. Figure 2.4 shows the timing of sampling in relation to the hydrograph during 2008 and 2009. Details of the sampling design for water quality, aquatic and riparian habitat and biota in the ten Tully-Murray lagoons is summarised in Table 2.1.

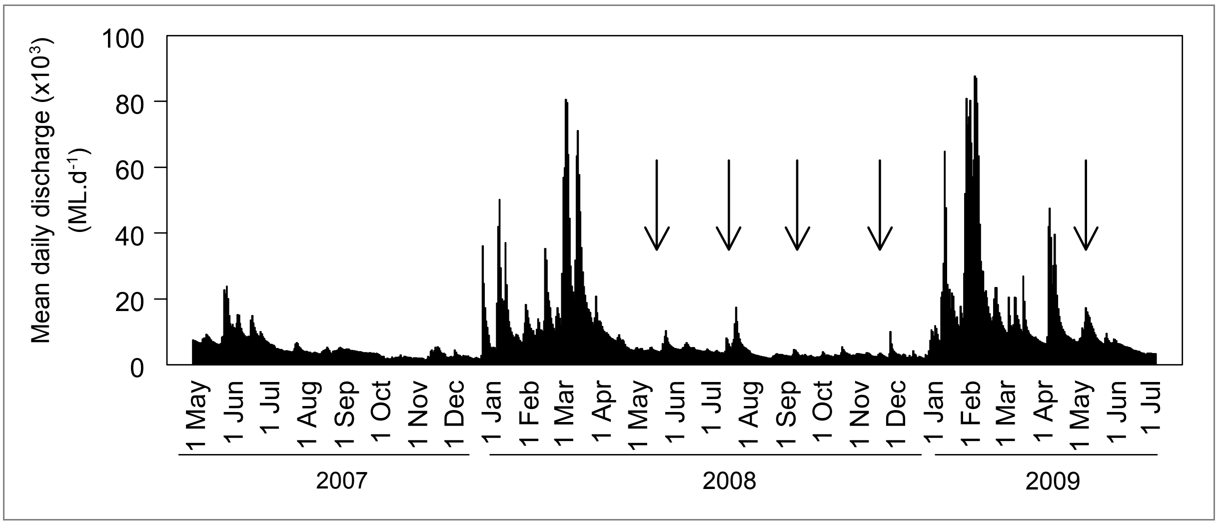

Figure 2.4: Daily discharge of the Tully River (Euarmo gauge) over 2007-2009. Arrows mark the approximate date of each biophysical survey of the floodplain lagoons. Daily flow data was supplied by the Queensland Department of Environment and Resource Management.

Figure 2.5: Views of Digman's Lagoon illustrating changes in water level in (a) September 2008 (dry season), (b) March 2009 (wet season) and (c) May 2009 (immediate 'post' wet season).

Table 2.1: Sampling design for water quality, aquatic and riparian habitat and biota in the ten Tully-Murray lagoons. Numbers indicate the number of transects (vegetation and habitat structure), sites (invertebrates), habitats (fish) and surveys (temporal) across the ten lagoons. Lagoons are ordered from from downstream to upstream locations (channel distance).

				Lagoon									
Sampling dimension	Sampling design component	Method	Habitat	Barrett's	Carroll's	Digman's 3	Digman's	Bunta	Boongaray	Selby's	Zamora's	Raccanello's	Kyambul
Spatial	Water quality	Physico-chemical	Microhabitats and open water	3	2	2	3	3	2	2	2	3	2
		Nutrients	Composite	1	1	1	1	1	1	1	1	1	1
		Herbicides	Composite	1	1	1	1	1	1	1	1	1	1
	Habitat structure and condition		Aquatic	7	4	3	3	5	4	4	4	5	3
			Riparian	7	4	3	3	5	4	4	4	5	3
	Spatial organisation of biotic assemblages	Plankton	Open water	2	2	2	2	2	2	2	2	2	2
		Invertebrates	Microhabitats	3	2	2	3	3	2	3	2	3	3
		Fish	Lagoon margin	3	3	2	2	2	2	3	2	3	2
			Channel-lagoon connection	1	1	2	2	2	2	1	2	1	2
			Microhabitats	3	2	2	3	3	2	3	2	3	3
	Habitat-biotic association	Invertebrates	Leaf litter	1	1	-	1	1	-	1	-	1	1
			Submerged macrophyte	-	-	1	1	1	1	1	1	1	-
		Fish	Leaf litter	-	1	-	1	1	1	1	1	1	1
			Submerged macrophyte	1	-	1	1	1	1	1	1	1	1
			Emergent macrophyte	1	1	1	1	1	1	1	1	1	1
Temporal	Seasonal variation in biotic assemblages	Invertebrates	Microhabitats	3	3	3	3	3	3	3	3	3	3
		Fish	Lagoon margin, channel-lagoon connection and microhabitats	3	3	3	3	3	3	3	3	3	3
	Recruitment dynamics: relationships with flow and connectivity	Fish	Lagoon margin, channel-lagoon connection and microhabitats	5	5	5	5	4	-	-	-	-	-

2.3 Field and laboratory methods

2.3.1 Physico-chemical methods

Physical processes that shape biotic assemblages in floodplain river systems operate at multiple spatial and temporal scales (Junk *et al.* 1989; Tockner *et al.* 2008). The spatial scales that structure biotic communities in floodplain systems include the floodplain landscape, the gross morphology of the entire water body and habitat structure within each water body (Winemiller *et al.* 2000; Arthington *et al.* 2005; Balcombe *et al.* 2006; Balcombe and Arthington 2009). Physical variables were measured from GIS maps or on-site (Table 2.2). Attributes of aquatic habitat were measured within 10 m wide belt transects (hereafter referred to as the habitat transects) that extended across the wetted width of each lagoon, perpendicular to its longest axis. Structural elements were estimated either as a proportion of the total transect area (e.g. cover of aquatic macrophytes) or the number of occurrences (e.g. density of large woody debris) within a transect (Table 2.2). The vegetation was identified to the lowest taxonomic level possible (genus or species). The number of transects per lagoon was proportional to its length, enabling comparisons of cover and density estimates among lagoons of variable dimensions. These transects were fixed and re-visited on subsequent sampling occasions.

Aspects about the nature, extent and integrity of the riparian vegetation were assessed using the assessment protocol of Werren and Arthington (2002). Riparian condition was assessed along both shorelines of each lagoon within the same transects used to estimate the condition of aquatic habitat. 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 regeneration of indigenous species. Each component was scored from 1 (poor) to 5 (very good) providing a maximum score of 25 for each lagoon bank, which were then summed and divided by two to provide an average score for each lagoon.

Variables describing the physico-chemical condition of lagoons, including water temperature (°C), conductivity µs.cm^{-1}), pH and dissolved oxygen (% saturation) were measured with a Hydrolab multi-parameter probe (Quanta). Transparency (m), and by inference turbidity, was estimated with a Secchi disk. Measurements were recorded at four positions (adjacent to lagoon margin 1 m, 4 m and mid-lagoon) from within a selection of the habitat transects (Table 2.2). Readings were taken at one metre depth intervals from the surface to the bottom at each of the four positions. Where possible, measurements were recorded at a similar time of day across all lagoons and on the first and subsequent sampling occasions. Multiple measurements over 24 hours were not possible, limiting the value of our records of dissolved oxygen and pH.

The concentration of water column chlorophyll *a* (a surrogate for algal biomass) and nutrients (nitrogen and phosphorus species) were measured in both open water and adjacent to structural habitat in two of the habitat transects. All samples were from approximately 20 cm below the surface collected in one-litre acid-washed polyethylene bottles, and stored on ice for up to two hours before being filtered through a 0.45 µm Whatman GF/C filter paper to separate algal cells. These filters were stored in aluminium foil at -20°C until being extracted in acetone for one hour and analysed using a spectrophotometer at the Australian Centre for Tropical Freshwater Research, James Cook University, Townsville. The concentration of chlorophyll *a* was determined in the presence of phaeophytin using the formula from Jeffrey and Humphrey (1975) after correction for the homogenate and solvent.

Table 2.2: Environmental variables estimated at each sampling location. For some, there were multiple measurments recorded (e.g. maximum, mean and minimum pH).

Variable	Acronym	Description
Landscape		
Channel (Ch), straight line (SL) distance to river mouth	DIST_RM	Estimate from GIS map
Channel and straight line distance to river	DIST_R	Estimate from GIS map
Lagoon attributes		
Number of incoming channels	Chan	Estimate from GIS map
Surface area	Sf_Area	"
Length	Length	"
Width	Width	"
Perimeter	Perimeter	"
Length-to-width ratio	L-W ratio	"
Maximum depth	Depth	Hydrolab water quality probe
Width-to-depth ratio	W-D ratio	In-situ and GIS map
Riparian condition	Rip. Con.	Standard protocol (Werren and Arthington 2002)
Connectivity metrics	CONN	Hydrologic modelling
Within-lagoon attributes		
Large woody debris	LWD	Estimated visually from multiple habitat transects
Leaf litter cover	LL	"
Total plant species richness	Total_SR	"
Native plant species richness	Native_SR	"
Alien plant species richness	Alien_SR	"
Total plant cover	Total_Cover	"
Native plant cover	Native_Cover	"
Alien plant cover	Alien_Cover	"
Total submerged plant cover	Submerg_Cover	"
Total emergent plant cover	Emerg_Cover	"
Water temperature (˚C)	Temp.	Measured '*in situ*' using a Hydrolab water quality meter
pH (pH units)	pH	"
Conductivity ($\mu S.cm^{-1}$)	Cond.	"
Dissolved (% saturation)	DO	"
Transparency (m)	Trans.	Secchi disk
Water column chlorophyll *a* ($\mu g.l^{-1}$)	Chl a	1 L water samples
Dissolved inorganic nitrogen ($\mu g.l^{-1}$)	DIN	"

Variable	Acronym	Description
Dissolved organic nitrogen (µg.l^{-1})	DON	“
Particulate nitrogen (µg.l^{-1})	PN	“
Total phosphorus (µg.l^{-1})	TP	“
Atrazine (µg.l^{-1})	Atrazine	“
Desethyl Atrazine (µg.l^{-1})	De_Atzine	“
Diuron (µg.l^{-1})	Diuron	“
Hexazinone (µg.l^{-1})	Hex	“
Simazine (µg.l^{-1})	Sim	“

Owing to the cost of chemical analysis, only a single water sample was collected for analysis of Photosystem II herbicides in each lagoon. A composite one-litre water sample comprising approximate 0.25 L of water was collected from the same four positions (and depth) as the nutrient and algal biomass measurements. Water samples were analysed using liquid chromatography mass spectrometry (LCMS) at the National Association of Testing Authorities accredited Queensland Health Scientific Services Laboratory, Brisbane. LCMS is the preferred method for the range of herbicides targeted in this investigation because of the method's lower LOR (limit of reporting) for triazine herbicides.

2.3.2 Zooplankton

Zooplankton tows were confined to open water habitat of the mid channel to minimise the net encountering submerged snags and vegetation. Zooplankton was sampled during daylight hours using a single 0.5-m diameter 0.8-m-long conical net with 80-µm mesh that was attached to a rope and deployed from the rear or of a motorised boat or, in heavily snagged lagoons, was attached to the end of a pole and deployed from the side of the boat. Oblique tows were performed in the upper 1.5 m of all lagoons to ensure the net avoided disturbing the sediment and snags. Four, three-minute, replicate tows were performed in each lagoon at a average speed of 0.45 ms^{-1} thereby filtering approximately 12.3 m^{-3} of water. Samples were sorted, identified as far as possible using available keys and counted under a stereo microscope in a bogorov tray. Results were analysed using pooled data from each site.

2.3.3 Macroinvertebrates

Invertebrates associated with macrophytes and leaf litter (i.e. two of the dominant microhabitats consistently recorded among lagoons) were sampled from within a selection of the habitat transects that also corresponded to the position of fish sampling (Table 2.1). The number of transects sampled per lagoon was proportional to its length enabling comparisons of species richness and assemblage composition among lagoons. Each invertebrate sample was collected using a square frame dip net with a 250 µm mesh. Sampling effort was standardised by the area (0.5m^2) of vegetation or leaf litter swept by the dip net. The material collected in the dip net was washed into a 1 L plastic container, fixed on 70% ethanol and returned to the laboratory for processing. In the laboratory, samples were sorted under a 3x magnifying lamp for 60 minutes each. This process removed most of the animals present. Animals were then identified to the lowest possible taxon from available keys. This approach follows a relatively rapid assessment method, in keeping with the need for cost effectiveness of any monitoring protocol that may be implemented.

2.3.4 Fish

Fish sampling focused on small-bodied fish species and the early life-history of any larger species; this design maximised information return on the structure of lagoon fish assemblages and the recruitment patterns of component species. Targeting the smaller size group also allowed for the choice of sampling gear that would minimise the opportunity for encounters with estuarine crocodiles that occurred across the study area. Furthermore, it was undesirable to sample large fish that may be few in number and vulnerable to disturbance.

Fish assemblages within each lagoon were examined using two sampling methods. Fyke netting provided insight into the dynamics of freshwater fishes that occupy the lagoon margins and data on the movement patterns between natal habitat (i.e. stream network and estuaries) and the floodplain lagoons. Up to three pairs of dual-winged, fine-mesh, fyke nets were set back-to-back around the margin of each lagoon in areas of about 1-2 m depth, with the tips of the wings approximately 5 m apart and the cod end held above the water surface to ensure any air-breathing species (turtles, etc.) captured would not drown. The net pairs were spaced no closer than 50 m to avoid interference between neighbouring net pairs. In addition, pairs of nets were set at the point of connection between the channel network and the lagoon; the number of these settings varied between one and two connection points among the ten lagoons. Fyke nets comprised dual wings, each 2 m long x 1.5 m deep, with five supporting hoops (0.5 m diameter) and a stretched mesh size of 2 mm that enabled a comprehensive size range of smaller-bodied species to be collected. The nets were deployed at midday and retrieved approximately 24 hours later. The duration of sampling was noted for each net for subsequent calculation of catch per unit effort (CPUE), where CPUE represented the abundance values for all species collected per net set for 24 hours, soak time.

Species associated with structural habitat elements, which included some species and developmental groups not well represented in the fyke nets, were sampled with a back-pack electrofishing unit (Smith-Root model 7) and modified anode (15 cm diameter ring). Fishes were sampled in a habitat-specific manner from the three dominant microhabitats (leaf litter and submerged and emergent macrophytes) that occurred most consistently across lagoons. The electrofisher was operated at 300 V, 50-70 Hz frequency and 4 msec pulse width from an aluminium dinghy. The occurrence (presence-absence) and total abundance of individuals associated with each microhabitat was standardised across lagoons by performing an equal number (10 x 30 second ‘on time’) of electrofishing shots within each microhabitat type; this was adequate to represent the diversity of fish species associated within each microhabitat (Figure 2.6).

Habitat replicates were selected at random from across the entire lagoon and not confined to the habitat transects due to the insufficient quantity of some habitat groups within the confined transect space. All fishes collected by fyke netting and electrofishing were placed into a plastic sorting tray and those that could be identified were individually measured (standard length, L_s) and released alive at the point of capture (except alien fishes which were euthanized on site in accordance with the *Queensland Fisheries Act 1994*). In the case of very large catches, a random sub-sample of 100 individuals was preserved in a solution of 10% formalin (37% aqueous solution of formaldehyde, diluted in water) for identification and length measurement in the laboratory. Individuals representing the early developmental stages were also preserved in formalin for subsequent identification in the laboratory. Preserved fish larvae were identified to species and counted using a dissecting microscope in the laboratory. Specimens were categorised according to developmental stage as pre-flexion, flexion or post-flexion larvae or juveniles/adults based on the development of the notochord and the appearance of the adult phenotype (Kelso and Rutherford 1996; Neira *et al.* 1998; Leis and Carson-Ewart 2000). Where the larval stages of species were not represented in the lagoons, fish were separated into size categories to distinguish between cohorts (based on published literature). The standard lengths of all preserved fish larvae were measured to the nearest millimetre using an eyepiece graticule.

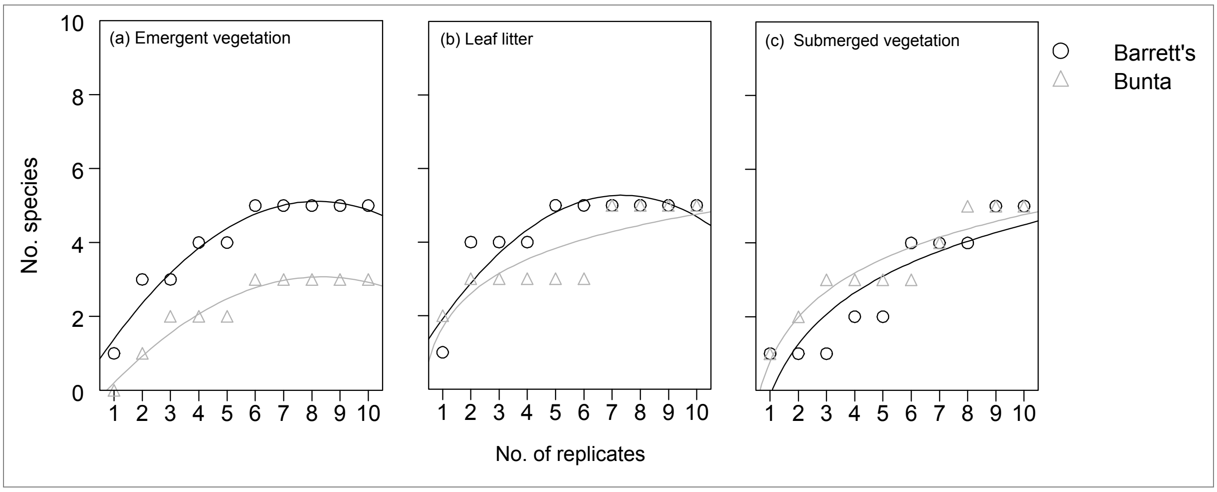

Figure 2.6: Species accumulation curves illustrating the optimum number of 30 second electrofishing replicates in three microhabitats of two floodplain lagoons in the Tully-Murray river system.

2.4 Data analysis

2.4.1 Environmental gradients

We used principal components analysis (PCA) to explore relationships among 48 variables describing position in the landscape, local habitat structure and human land use, to identify dominant gradients of environmental variation (i.e. variables loaded on the same component) in the data set. The data set comprised four variables describing the position of each lagoon in the landscape, seven variables describing gross morphology of the lagoon, one composite metric describing the extent, nature and integrity of the riparian vegetation, six variables describing the structure and condition of the aquatic habitat and 25 variables describing lagoon water chemistry (Table 2.2). All variables were measured on-site or estimated from maps or GIS. The distributions of variables were visualised as pair-wise scatter plots to identify those that were heavily skewed, and where necessary were transformed before all variables were normalised (i.e. subtracting the mean and dividing by the standard deviation) prior to the analysis. The PCA was performed in PRIMER-E© (v 6.1.12). Following ordination, relationships between a selection of habitat variables loaded together on individual principal components were examined using Pearson’s correlation coefficient (α =0.05). The analysis was performed in SPSS 17.0.1 for Windows©.

2.4.2 Hydrologic condition and the extent of connectivity

This section was developed from collaboration between MTSRF Projects 3.7.3 and 3.7.4. See Wallace *et al.* (2010) and Karim *et al.* (2010) (MTSRF Research Report Series: http://www.rrrc.org.au/publications/wetlands_and_floodplains.html).

2.4.3 Univariate comparison of zooplankton, invertebrate and fish assemblage characteristics

A number of measures of assemblage structure were derived using standardised CPUE data from each lagoon (combined fyke net and electrofishing data) collected over the first survey (plankton) (May 2008), all five surveys (invertebrates) (May, July, September and November 2008 and May 2009) and three comprehensive surveys (May and September 2008 and May 2009) (fish).

For the fish samples, the imbalance in the allocation of nets to the ‘lagoon margin’ and the ‘connection point’ (i.e. the point where a lagoon and stream met) habitat required averaging fyke catches from within each habitat prior to pooling catches across habitats to provide a standard unit of fyke net effort for each lagoon (Table 2.1). Similarly, the absence of individual microhabitats within some lagoons required the averaging across all electrofishing replicates within a lagoon when standardising effort for comparison of fish numbers among lagoons Univariate measures included: species richness (total number of species collected per lagoon), evenness (Pielou’s J), total abundance and the proportion of total abundance contributed by exotic species. 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 anthropogenic disturbance on stream fish communities (Karr 1981; Kennard *et al.* 2005, 2006a, 2006b). Analysis of variance was used to compare means for each lagoon.

2.4.4 Multivariate comparison of assemblages and relationships with environmental gradients

The spatial organisation and the temporal variability of the plankton, invertebrate and fish assemblages were analysed using the samples indicated above.

Fish samples

Fish assemblage structure was examined separately over the two scales of investigation (i.e. among the ten lagoons sampled on three occasions and among the five lagoons that were sampled on four or more occasions). All multivariate analyses were performed in PRIMER-E© (v. 6.1.12). The data were transformed to ln (abundance + 1) before constructing sample-by-sample association matrices using the Bray-Curtis dissimilarity index. Differences in species density owing to: (i) lagoon; and (ii) survey identity were analysed in separate one-way analysis of similarity (ANOSIM).

ANOSIM compares rank similarities within *a-priori* defined groups against rank similarities between groups and calculates a statistic, Global *R*, which lies along a scale between +1 and -1 (Clarke and Gorley 2006). Values approaching +1 in this analysis would reveal that samples from within the same lagoon (or surveys) were more similar to one another than any samples from different lagoons (or surveys); an *R* value of 0 occurred if samples from different lagoons were perfectly mixed and bore no relationship to the lagoon identity; and a value of -1 would indicate that the most similar samples all occurred outside the lagoon.

Statistical significance was determined by referring the observed value of *R* to its permutation distribution that is generated from the random reshuffling of the sample labels that occurs 999 times with *R* calculated for each permutation (Clarke 1993). Following Global *R* calculation, pair-wise *R* contrasts were employed to determine where between-group differences lay. The pair-wise *R* value lies between 0 and 1; values close to 1 indicate separation of the two groups; values close to 0 imply little or no segregation.

Analysis of microhabitat use was based on electrofishing data collected over the three comprehensive surveys. Emergent vegetation was separated into the dominant floral species or groups (for rarely occurring taxa) to investigate relationships between fish occurrence/ abundance and the deterioration in habitat diversity from the proliferation of introduced ponded pasture grass (e.g. *Hymenachne amplexicaulis*). Microhabitat groups were pooled across lagoon prior to transformation (presence-absence and ln abundance +1 and construction of the sample-by-sample association matrices using the Bray-Curtis dissimilarity distance measure. Patterns of association between fish assemblages and microhabitat were explored using one-way ANOSIM.

Ordinations constructed using non-metric multidimensional scaling (nMDS) were used to map spatial and temporal relationships among samples in two dimensions. Ordinations were constructed from the sample-by-sample association matrix to illustrate: (i) the pattern of groupings according to lagoon and microhabitat identity; and (ii) the collective nature of temporal trends in the lagoon group. Information on the strength and nature of the correlation of dominant taxa within the nMDS space was added to the ordinations as vectors indicating the direction of greatest increase in the number of the abundance of individuals most highly correlated with the space. The direction of these vectors was determined by correlation of each species on the nMDS space, with the length reflecting the R value for each correlation. No procedure is provided in PRIMER-E© to determine the statistical significance of these correlations.

Associations between fish assemblage structure and explanatory environmental variables (Table 2.2) were explored in the same manner by superimposing the environmental variables onto the biological ordination.

To identify recruitment events in relation to sampling date and hydrologic and connectivity conditions, length-frequency histograms were constructed for *Neosilurus ater*, *Hypseleotris* sp. 1 and *Melanotaenia splendida splendida*. These species capture the diversity of fish recruitment strategies that exists within the lagoon fish assemblage. Length data from both sampling methods were used to construct these plots ensuring a more comprehensive size range of individuals was included for those species that undergo ontogenetic shifts in habitat use. Recruitment patterns were examined only in the five lagoons that were sampled extensively on more than three occasions (Barrett's, Digman's, Selby's, Zamora's and Kyambul lagoons). Differences in the extent of hydrological connectivity between floodplain lagoons and the stream network and its effects on population size structure of migratory fishes was examined by comparing the length-frequency data of *Hypseleotris compressa* (a fish species that recruits into floodplain lagoons via the stream network under channelised flow) across the same five lagoons that represent a gradient of hydrological connectivity.

Plankton and invertebrate samples

Similar analyses were performed on the zooplankton and invertebrate samples using the PCORD® package (McCune and Mefford 2006), using the nMDS procedure as above, and the Multi-response Permutation Procedure (MRPP), which is similar to ANOSIM, described above. The contributions of each taxon to the analysis axes were investigated by linear regression; and the links between environmental variables and the nMDS axes was similarly tested, using the Sigmaplot® package.

Cardwell Values and Threats assessments

A classification of values and threats to wetlands (here termed 'Cardwell Values and Threats assessments'), produced by Cassowary Coast Regional Council Revegetation, Nursery and Pest Control Department (via the then-named Cardwell Shire Floodplain Project (D. Sydes, pers. comm.), was also compared with the invertebrate and fish samples to test for any accord between the ecological data and the independently generated classification. This classification of wetlands in the Tully-Murray used a Decision Support System (DSS), developed under the Great Barrier Reef Coastal Wetlands Protection Programme, to help prioritise investment in wetland protection and management within the catchment. A DSS involves multiple criteria analysis based on a set of alternatives or choices, evaluation criteria, performance values and criteria weighting. There were 22 wetland criteria spread across three classes (values, threats and capacity) (Smith *et al.* 2007). Significant relationships between our analyses and Values and Threats scores were illustrated graphically.

Fish sampling with electro-shocker R. PEARSON

Invertebrate sampling R. PEARSON

3. Results

3.1 Environmental gradients

The main field study aimed to measure spatial and temporal variability of biophysical variables in floodplain wetlands of the Tully-Murray catchment along two sets of gradients: (i) natural environmental gradients, including position in the floodplain landscape, wetland size and morphology, hydrology and connectivity, and (ii) gradients of disturbance, including gradients of land use, hydrology and connectivity, water chemistry and habitat quality, riparian disturbance, and alien species.

Although every effort was made to establish strong disturbance gradients, ranging from near pristine to highly degraded lagoons, we were unable to identify simple gradients. Given the lack of high quality reference conditions within either catchment, and the mixture of different disturbances across the two catchments, the field design was therefore treated as a multivariate case study with the potential to reveal some gradients but a greater likelihood of discovering multiple interacting gradients that were not necessarily co-varying. This multivariate study design can nevertheless inform conceptual models and help to validate indicator variables.

The first five principle components collectively explained approximately 70% of the total observed variation among the 48 environmental variables measured in the floodplain lagoons of the Tully-Murray River system (Table 3.1). Principal component 1 (PC1) accounted for 23.0% of the variance observed and equated to a gradient principally in water chemistry that was related to lagoon position in the catchment (Table 3.2).

Table 3.1: Eigenvalues and proportion of variance accounted for by each of the first five components.

PC	Eigen values	Percentage of variance	Cumulative percentage of variation
1	11.1	23.0	23.0
2	7.01	14.6	37.7
3	6.08	12.7	50.3
4	5.21	10.9	61.2
5	3.4	7.1	68.3

Lagoons arrayed negatively on this component (e.g. Kyambul, Boongaray) were situated more distant from the river mouth (via the channel) and were characterised by water of low transparency (Figure 3.2a). Lagoons situated at greater distance from the river mouths also contained higher concentrations of dissolved oxygen and diuron (Figure 3.2b,c). PC2 (14.6%) represented a gradient in features describing the gross morphology of the lagoon and associated attributes of habitat structure (Table 3.2). Lagoons loaded positively on this axis (e.g. Zamora's and Digman's 3) were short, wide and shallow (i.e. with a small length-to-width and width-to-depth ratios) and contained a high proportion of submerged vegetation cover (Figure 3.3a).

The proportion of total vegetation cover (pooled submerged, floating and emergent growth forms) comprising native taxa was also higher in these lagoons (Figure 3.3b). In contrast, lagoons positioned negatively on PC2 tended to be long and narrow and characterised with a high proportion of emergent vegetation (Figure 3.3b). PC3 (12.7%) equated to a gradient in water chemistry and habitat structure (Table 3.2). Lagoons located negatively on this axis (e.g. Boongaray, Zamora's and Barrett's) contained water with high maximum conductivity and algal biomass. Total vegetation cover (comprising both native and alien forms) as well as the proportion of taxa being of introduced origin was also high in these lagoons.

Table 3.2: Principal components analysis: loadings of individual variables on the first five principal components. Each component explains a different sub-set of the data variance. Similar scores within each component indicate those metrics that vary together (positively or negatively). See text for interpretation of the principal components.

Variable	PC1	PC2	PC3	PC4	PC5
Average transparency (m)	0.258				
Maximum transparency (m)	0.255				
Minimum transparency (m)	0.241				
Alien plant cover	0.158				
Maximum pH	-0.186				
Maximum dissolved oxygen (%sat.)	-0.197				
Channel distance to the Tully or Murray River mouth	-0.200				
Minimum pH	-0.201				
Minimum conductivity ($\mu s.cm^{-1}$)	-0.207				
Average conductivity ($\mu s.cm^{-1}$)	-0.209				
Average pH	-0.217				
Diuron ($\mu g.l^{-1}$)	-0.221				
Average dissolved oxygen (%sat.)	-0.237				
Native plant cover		0.253			
Submerged plant cover		0.237			
Width-to-depth ratio		0.182			
Log (average dissolved inorganic nitrogen) ($\mu g.l^{-1}$)		-0.126			
Emergent plant cover		-0.162			
Log (average particulate nitrogen) ($\mu g.l^{-1}$)		-0.173			
Maximum depth		-0.187			
No. incoming drains/channels		-0.204			
Average total phosphorous ($\mu g.l^{-1}$)		-0.220			
Hexazinone ($\mu g.l^{-1}$)		-0.223			
Log (Surface Area)		-0.232			
Log (lagoon length)		-0.250			
Perimeter		-0.256			
Length-to-width ratio		-0.285			
Minimum dissolved oxygen (%sat.)			0.216		
Log (average dissolved organic nitrogen) ($\mu g.l^{-1}$)			-0.128		
Alien plant species richness			-0.142		
Total plant cover			-0.220		

Variable	PC1	PC2	PC3	PC4	PC5
Log (average chlorophyll *a*) (µg.l^{-1})			-0.237		
Maximum conductivity (µs.cm^{-1})			-0.243		
Alien plant cover				0.211	
Straight line distance to Tully or Murray River mouth				-0.155	
Log (straight line distance to the Tully or Murray river)				-0.243	
Lagoon width				-0.262	
Log (channel distance to the Tully or Murray river)				-0.297	
Native plant species richness					0.329
Plant species richness					0.323
Large woody debris					0.277
Atrazine (µg.l^{-1})					0.271
Riparian condition					0.258
Desethyl Atrazine (µg/L) (µg.l^{-1})					0.256
Average water temperature					0.256
Maximum water temperature					0.245
Leaf litter cover					0.237
Minimum water temperature					0.235
Simazine (µg.l^{-1})					0.101

PC4 (10.9%) represented a gradient related to lagoon position on the floodplain and corresponding features summarising the gross lagoon morphology and habitat structure (Table 3.2). Those lagoons positioned negatively on this axis (e.g. Bunta, Barrett's and Raccanello's) were located further from either the Tully or Murray rivers (via the channel and straight line) and were of greater absolute width.

PC5 (7.1%) represented a gradient in the riparian condition and the associated changes to in-channel habitat structure and water chemistry (Table 3.2). The margins of those lagoons arrayed positively on PC5 (Barrett's, Digman's) were covered by riparian vegetation providing a relatively high load of large woody debris and packs of leaf litter. Water temperature in lagoons with an intact riparian zone appeared lower (although the trend was not significant) (Figure 3.1). A suite of herbicides was also detected in these lagoons (Table 3.2).

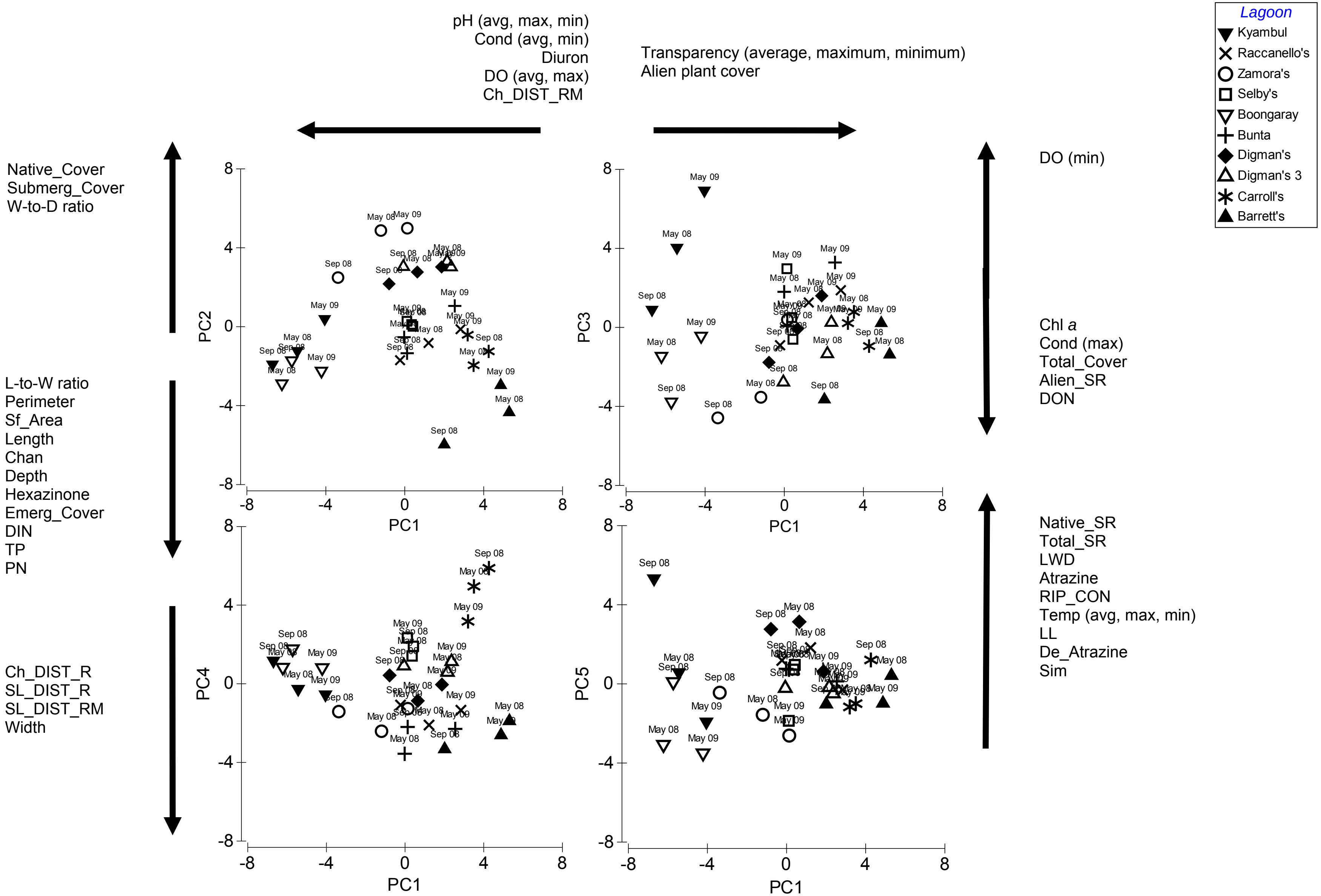

Figure 3.1: Location of floodplain lagoons of the Tully-Murray system within the space as defined by the first five principal components. Survey dates are aligned with symbols. Refer to Table 2.2 for explanation of the abbreviation of environmental variables.

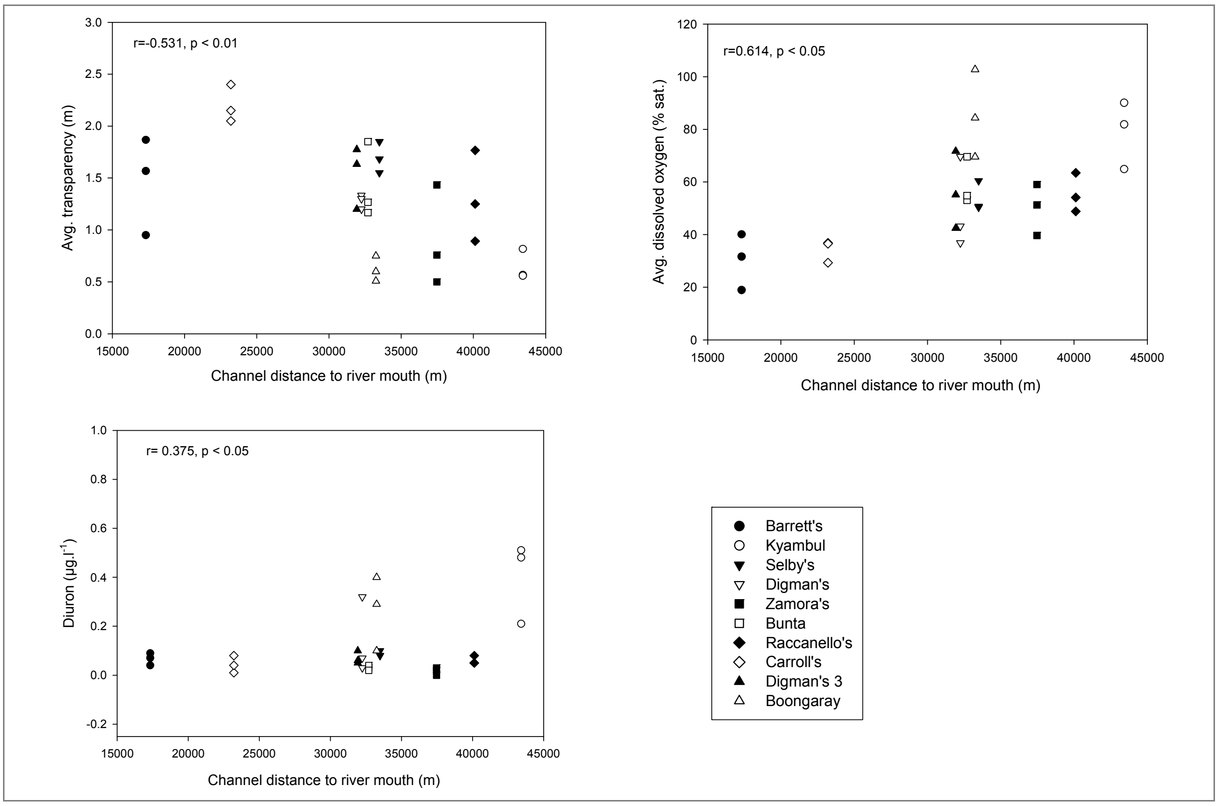

Figure 3.2: Longitudinal variation in a suite of water chemistry variables measured in ten floodplain lagoons of the Tully-Murray River system.

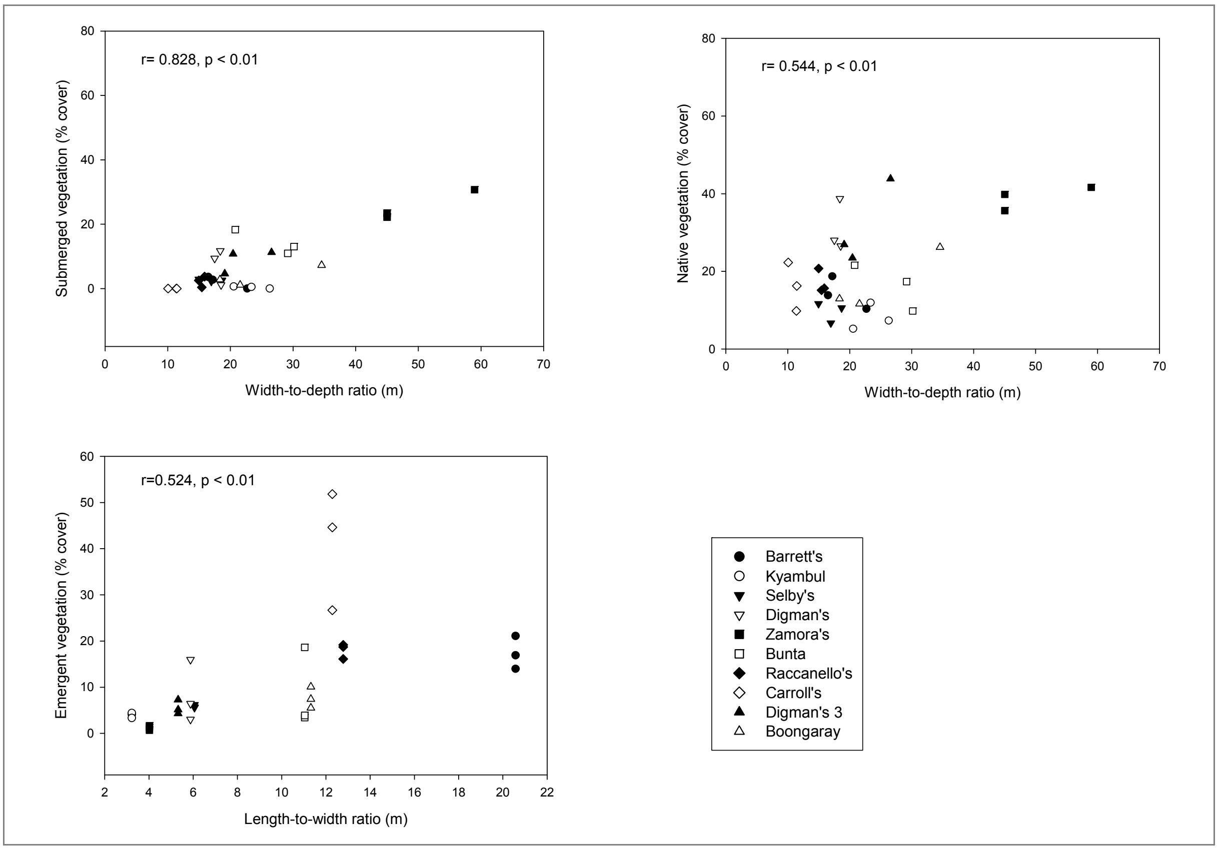

Figure 3.3: Relationship between attributes characterising lagoon gross morphology and vegetation structure in ten floodplain lagoons of the Tully-Murray River system.

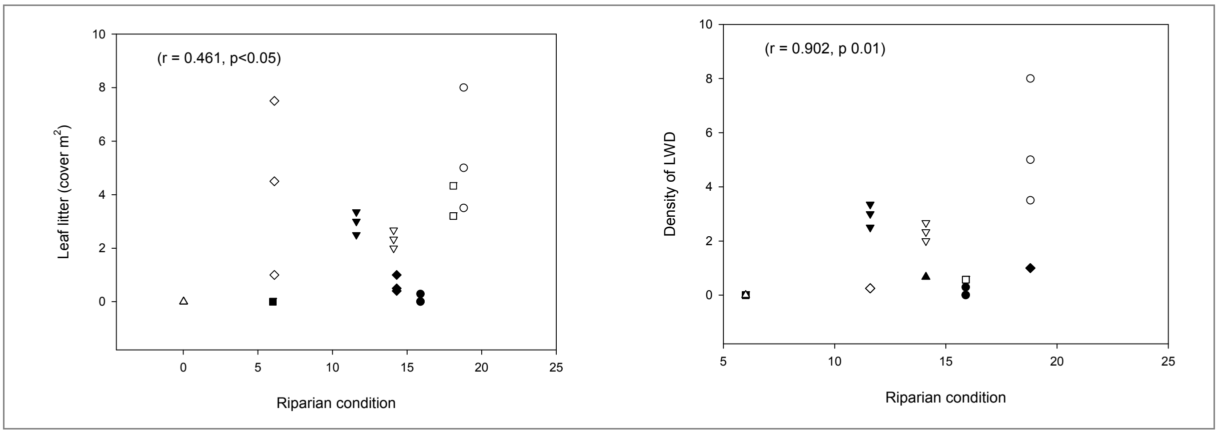

Figure 3.4: Relationships between riparian condition and habitat structure in ten floodplain lagoons of the Tully-Murray River system.

3.2 Zooplankton

Zooplankton were abundant, but were apparently not diverse, within the constraints of identification under rapid assessment protocols. The assemblages were dominated by two forms of Copepoda and two of Cladocera, with occasional occurrences of Oligochaeta, Chironomidae and Ephemeroptera.

Patterns in the distribution of the assemblages were investigated using non-metric multidimensional scaling (nMDS) in the PC-ORD package (McCune and Mefford 2006). Like a PCA (above), this analysis plots sites that are similar to each other (in this case on the basis of their zooplankton assemblages) close together, and separates those that are different. nMDS is preferred for biological data as it copes better with non-normal and non-linear data than PCA. Several axes for the plot (ordination) can demonstrate clear gradients which then can be associated with environmental variables, suggesting possible cause-effect relationships. Any *a priori* site groupings may be tested with multi-response permutation procedures (MRPP), also in PC-ORD.

Figure 3.5 shows the nMDS ordination plot. There was substantial overlap between sites. Some separation by month is evident in the right-hand plot with samples from May 2009 (month 5) separating on Axis 1.

Significant relationships between environmental variables and the first two nMDS axes are shown in Figure 3.6. Significant relationships were evident with size and morphometry variables (e.g. perimeter, width to depth ratio), water quality variables (transparency, hexazinone, dissolved oxygen) and habitat variables (alien plant species and alien plant cover) and with the Cardwell Values and Threats land-use criterion. The positive relationships with alien plants is hard to explain, given that the plankton were sampled in open water, well away from any direct influence of aquatic plants.

The water quality relationships are also equivocal, there being a positive relation with dissolved oxygen (expected) and with hexazinone (unexpected). The lack of diversity of zooplankton may have adversely affected the analysis, but further investigation of these relationships is warranted. Further, the sorting and identification of zooplankton is very time consuming, and for monitoring purposes probably not cost-effective. Therefore, currently the zooplankton cannot be proffered as appropriate health indicators.

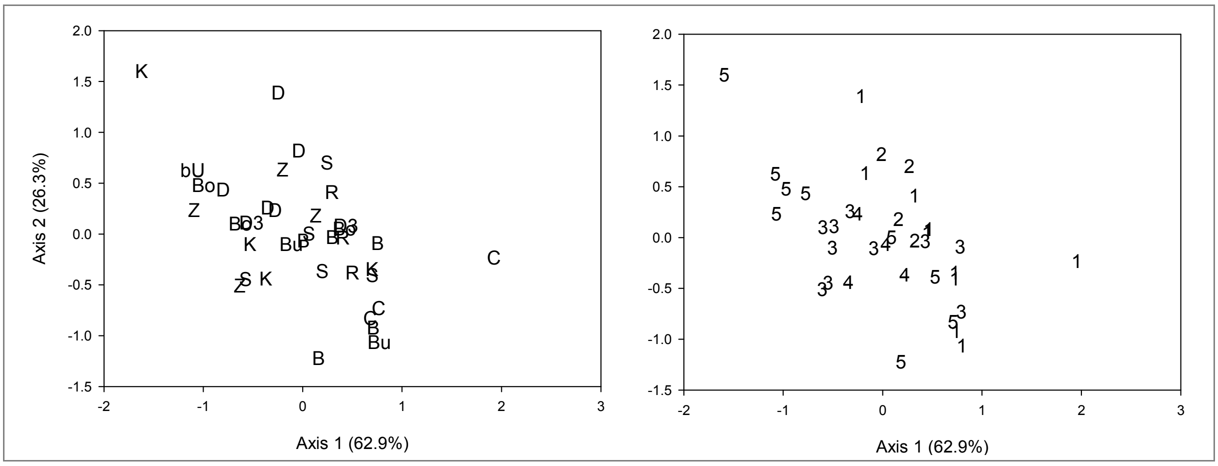

Figure 3.5: nMDS ordination of zooplankton samples. Site overlay is shown for Axis 1 vs. Axis 2 (left); sample overlays for the five monthly samples are shown for the same ordination on the right. Abbreviations for lagoons as in Figure 2.1 (not all shown in these figures if no relevant samples). Sample times are: 1: May 2008; 2: June 2008; 3: Sept 2008; 4: Nov 2008; and 5: May 2009.

Plankton sampling R. PEARSON

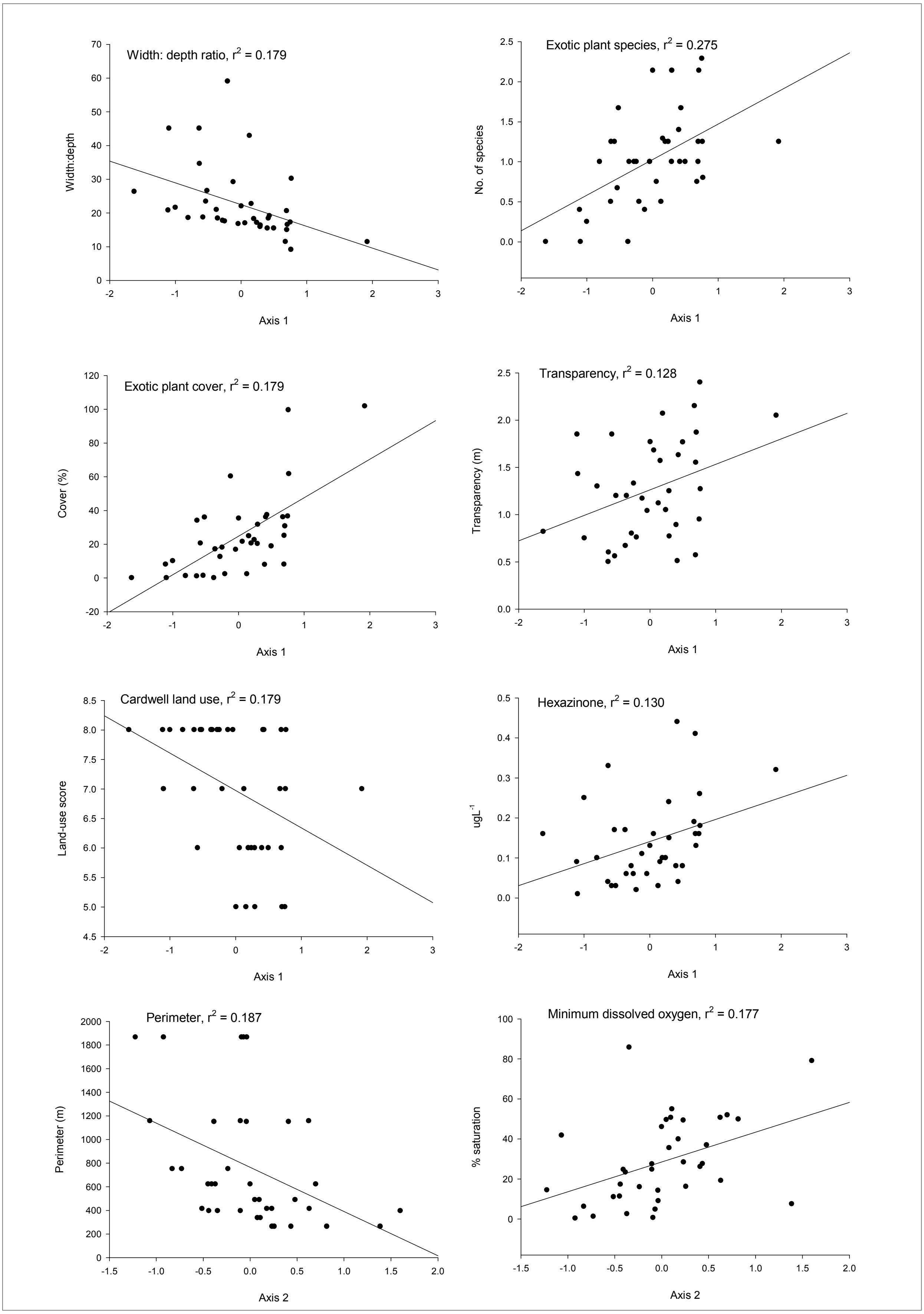

Figure 3.6: Zooplankton: relationships between nMDS axes 1 and 2; and selected environmental variables and Cardwell Values and Threats assessments.

3.3 Invertebrates

3.3.1 Assemblage composition

Diverse assemblages of invertebrates were found at each site (Table 3.3), representing the major taxa expected from floodplain wetlands in Queensland. Several ubiquitous taxa characterised the overall fauna. They included Oligochaeta (worms), Acarina (mites), Cladocera and Ostracoda (small crustaceans), Atyidae (shrimps), Pyralidae (moth larvae), Ceratopogonidae and Chironomidae (fly larvae), Corixidae (water boatmen), various Zygoptera and Epiproctophora (damselflies and dragonflies), and Leptoceridae (caddisflies). Many other taxa occurred at most or many sites. Taxonomic diversity showed some relationship with the numbers of samples collected and processed, with Digman's 3 lagoon having the lowest number of samples and taxa (Figure 3.7). The apparent (and expected) trend is not significant ($p = 0.108$), although the number of samples is low, reducing the power of the analysis. It is unlikely that this relationship would affect subsequent analyses which were largely based on the commoner species.

Patterns in the distribution of the invertebrate assemblages were investigated using nMDS in the PC-ORD package (McCune and Mefford 2006), as described for the zooplankton. analysis of the whole invertebrate data set produced clear separation of sites by habitat (MRPP for litter *vs.* macrophyte habitats: $A = 0.0435$; $p<0.001$), so all subsequent analyses were undertaken on separate litter or macrophyte samples. This separation led to different numbers of samples in the two analyses as some site-time combinations lacked one or other habitat.

3.3.2 Litter samples

Results of an nMDS ordination for samples from litter are shown in Figure 3.8. Many samples group tightly together (e.g. the S (Selby) samples), but some are widely spread (e.g. the D (Digman's) samples. Clustering of many sites towards the centre of the diagram suggests little separation between sites, with no major gradients in evidence. The right-hand diagrams shows the 'months' overlay, demonstrating substantial overlap among months. However, it is interesting to note that months 1 and 5 separate on Axis 2. These represent May 2008 and May 2009, indicating that years are not exact replicas of each other. Seeking representative sites and months for monitoring then becomes problematic. It is apparent that a suite of sites across several samples in any one year is required to create an adequate picture of the wetland communities.

Major drivers of the ordination are shown in Table 3.4, which lists species that correlate significantly ($p < 0.05$) with the nMDS axes. The different axes represent distributions among sites of different groups of species, with clear gradients between species groupings (positive and negative correlation coefficients on the same axis). Between-site similarities and differences based on MRPP analysis are shown in Table 3.5. Interestingly some of the least significant differences (e.g. Barrett's vs. Digman's) are between lagoons of apparently quite different character – Barrett's is a large deep lagoon with intact (though narrow) riparian vegetation while Digman's is small, man-made without riparian vegetation other than grasses.

Significant correlations between environmental variables and the axes are shown in Figures 3.9 and 3.10. A number of factors correlate with Axis 1. pH and dissolved organic nitrogen correlate positively and strongly, as does PC2 (axis score from an environmental principal components analysis). Plant species richness has a looser, negative, relationship. A positive relationship with atrazine, while significant, is driven mainly by three sites, and so appears a less robust relationship. The Cardwell recreation value has a negative relationship, as does the Cardwell threat – hydrology. Both indicate that the independently generated ratings are partly matched by the invertebrate data.

Axes 2 and 3 present a similar picture: several water quality and habitat variables, and Cardwell assessments, correlate with the axes. It is thus clear that there are gradients in invertebrate community composition that correlate with habitat, water quality and qualitative assessment variables. These relationships cannot be used to ascribe cause and effect, but they indicate that there is an ecological response to land-use effects on the habitats.

3.3.3 Macrophyte samples

Results of an nMDS ordination for samples from macrophytes are shown in Figure 3.11. There is no strong grouping of sites, with many samples from the same site spread across the ordination. Similarly, the monthly overlay shows substantial scatter, but again samples from May 2008 and May 2009 (labelled 1 and 5) are separated on Axis 2.

Major drivers of the ordination are shown in Table 3.6, which lists species that correlate significantly ($p < 0.05$) with the nMDS axes. The different axes represent distributions among sites of different groups of species, with clear gradients between species groupings (positive and negative correlation coefficients on the same axis). Between-site similarities and differences based on MRPP analysis are shown in Table 3.7. Many sites show little differentiation. But Digman's and Digman's 3 are mostly dissimilar from the rest. Interestingly, these are small artificial wetland sites and it may be that they have not yet had time to develop a diverse macrophyte fauna, reflected in the invertebrates.

Significant correlations between environmental variables and the axes are shown in Figures 3.12 and 3.12. A number of factors correlate with Axis 1 (Figure 3.12), although the r^2 values are not high, reflecting the scatter around the regression line. Transparency and depth are positively related to Axis 1, while plant cover is, interestingly, negative. Cardwell values are again related to the axis, with indigenous and fishery values being positive and vegetation and size negative.

Axes 2 and 3 correlate mainly with water quality and habitat variables (Figure 3.13). Axis 2 is related to dissolved inorganic nitrogen, but the relationship is largely driven by two sites. There is a positive relationship between Axis 3 and distance to the river, and to PC1 from the environmental principal components analysis; however, both relationships are driven by the contrast between one site (Barrett's) and the rest. There are more general negative relationships with pH, temperature and litter, but the r^2 value is low in all cases.

Again, there are gradients in invertebrate community composition that correlate with habitat, water quality and qualitative assessment variables, and again, these relationships cannot be used to ascribe cause and effect. Nevertheless, they indicate that there is an ecological response to land-use effects, albeit rather modest.

3.3.4 Invertebrate indicators

In an effort to seek possible indicators, similar to the invertebrate diversity indicators in the *Catchment to Reef* streams, composite habitat and water quality scores were derived by summing standardised values of key representative variables (selected from Table 2.2). No significant relationships were found between these scores and invertebrate diversity (Figure 3.14). Clearly invertebrates in the Tully-Murray wetlands relate primarily to habitat conditions, with water quality having some smaller influence.

Table 3.3: Invertebrate taxa recorded from each lagoon. Values are the number of samples in which each taxon was found at each site. Records from litter and macrophyte habitats are pooled. Note that in the total count, larvae and adults in a single taxon are treated separately.

		Lagoon									
		Barrett's	Kyambul	Selby's	Digman's	Zamora's	Bunta	Raccanello's	Carroll's	Digman3	Boongaray
Taxon	*No. of samples*	36	30	12	13	12	36	24	12	2	12
Nematoda		9	3		1	2	1	2			
Turbellaria			2		1						
Platyhelminthes						1		1			2
Hirudinea		1	3	1		1		1			4
Oligochaeta		9	16	2	2	1	12	7	3	1	2
Bivalvia	Corbiculidae				2		1	1			2
	Sphaeriidae		2	2	1		4				2
Gastropoda	Ancylidae		1	1	1						
	Hydrobiidae										
	Lymnaeidae, *Lymnaea*										8
	Planorbidae, *Amerianna*	12		6	3	4	7	10	7		8
	Planorbidae, *Gyraulus*	1	1		3		1	1			1
	Planorbidae, *Physastra*	2							1		1
	Planorbidae		1				1	2			
	Prosobranchia, *Coxiella*		2								2
	Gastropoda sp.		21								7
Acarina		16	8	3	6	8	20	13	6	1	6
Arachnida	Pisauridae	2		1	8		1	4	4		1
	Lycosidae		1		2	1					
	Tetragnathidae				1				1		1
	Terrestrial Arachnida	1	1		3		1				3
Cladocera		2	11	5	2	7	16	11	2	1	3
Copepoda		4	3	1	2	3	6	5	4		9
Ostracoda		9	20	6	5	7	24	13	10	1	20
Decapoda	Atiydae	33	30	12	13	12	36	24	11	2	12
	Palaemonidae		14	1	10		1	4			
Collembola										1	2
Lepidoptera	Pyralidae	1	6	2	6	5	7	5	3	1	2
Neuroptera	Sisyridae	6		1	6		2	1	1		
Coleoptera	Chrysomelidae adult			1							1
	Coleoptera indet. adult				3						1
	Curculionidae	2			3						1
	Dytiscidae l.			1	1	2	1	1		1	4

		Lagoon									
		Barrett's	Kyambul	Selby's	Digman's	Zamora's	Bunta	Raccanello's	Carroll's	Digman3	Boongaray
Taxon	*No. of samples*	36	30	12	13	12	36	24	12	2	12
	Dytiscidae adult		3		1	1	2	2			4
	Elmidae l.		11			1	12	2			2
	Elmidae adult		1	1			5	4			1
	Gyrinidae l.			3							
	Gyrinidae adult		1	2				1			
	Haliplidae l.					3	2	1			
	Haliplidae adult				1						5
	Hydraenidae adult		1	1	3	2		1	1		8
	Hydrophilidae l.	3	2	2	2	2	3	6	6		4
	Hydrophilidae adult	1									1
	Lampyridae adult		1								
	Noteridae										
	Scirtidae	3	2				2	1	3		1
	Staphylinidae adult	2									2
Diptera	Ceratopogonidae	18	15	4	6	9	18	10	3	1	11
	Ceratopogonidae pupa	9	2		12	6	8	6	2	1	4
	Chaoboridae				6		1				12
	Chironomidae	36	30	12	13	12	35	24	12	2	17
	Culicidae	4	13	1	11	5	5	4		1	3
	Dolichopodidae		2			1					
	Psychodidae							1			
	Tipulidae	1				1					
Ephemeroptera	Baetidae	1	18		6	12	19	11	1	2	13
	Caenidae	4	8	1	8	9	10	6		1	4
	Leptophlebiidae		4		2		3				
	Ephemeroptera sp.		1	1			3				3
Hemiptera	Belastomatidae,	1					2				8
	Corixidae	8	21	2	12	12	29	20	3	2	7
	Gerridae		8		14	2	4		2	1	2
	Hebridae		2		5	1	1	1	2	2	1
	Mesoveliidae, *Mesovelia*	2	3		5	1	1	1	3		3
	Naucoridae, *Naucoris*	3	3		3		2	7		2	5
	Nepidae	1	3	2	1	2	4			1	8
	Notonectidae	12	25	8	5	1	10	15	10		7
	Pleidae		1	2	10	5	1	2	1	2	2

		Lagoon									
		Barrett's	Kyambul	Selby's	Digman's	Zamora's	Bunta	Raccanello's	Carroll's	Digman3	Boongaray
Taxon	*No. of samples*	36	30	12	13	12	36	24	12	2	12
	Veliidae		5	1	6	1	3	3	3		2
Zygoptera	Coenagrionidae	12	10	12	7	11	16	17	7	2	11
	Isosticidae		9	2	7		3	1			
	Protoneuridae	1	4				4		1	1	8
	Zygoptera sp.	10	24	6	5	8	15	10	4	2	8
Epiproctophora	Aeshnidae				5		1	2			
	Gomphidae		6								1
	Hemi/Uroth/Libellulidae	6	1	7	8	7	15	13	7	2	1
	Lindeniidae	3	4	3	11	1	4	3	3		
	Macromiidae	1		1	5		2		2		5
	Epiproctophora sp.	17	3	3	8	6	10	9	6	2	5
Trichoptera	Calamoceratidae	5	17	4	7		10	6	7		
	Dipseudopsidae		5		1						1
	Ecnomidae	3		7		6	7		1	2	2
	Hydroptilidae *Acrit & Hel*	1		1		2	4	8	1		1
	Hydroptilidae, *Orthotrich*	1				2		1		1	
	Hydroptilidae, *Oxyethira*		1		1		3				
	Hydroptilidae, *Tricholei*				1		1				
	Hydroptilidae sp.		1			4	4	2			1
	Leptoceridae, *Oecetis*	14	1	8	1	5	11	12	10	1	1
	Leptoceridae, *Triplect*	5	13	5	3		13	3	2		
	Leptoceridae	1			3	1		2			3
	Trichoptera sp.	3	5	1	1	3	3	2		1	3
	Trichoptera pupa	2					1	1			3
Polyzoa	[statoblast]	10	1				3	10	1		3
No. of taxa		54	65	46	63	49	66	60	40	32	73

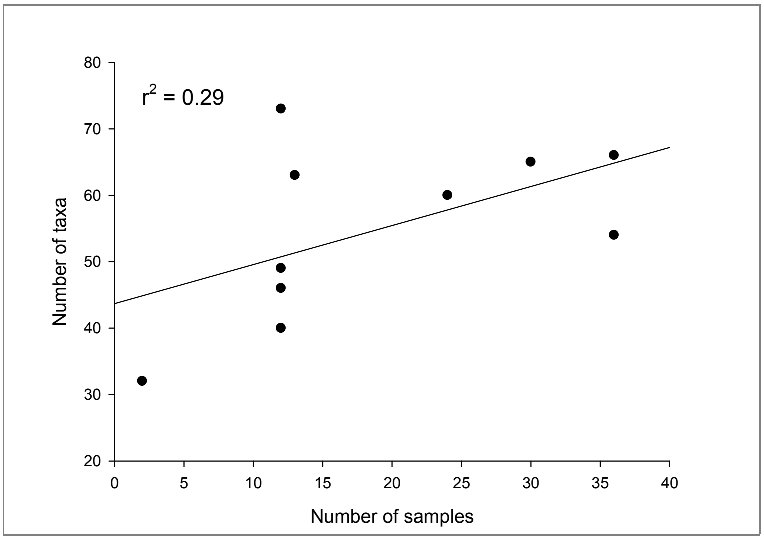

Figure 3.7: Number of invertebrate taxa vs. number of samples.

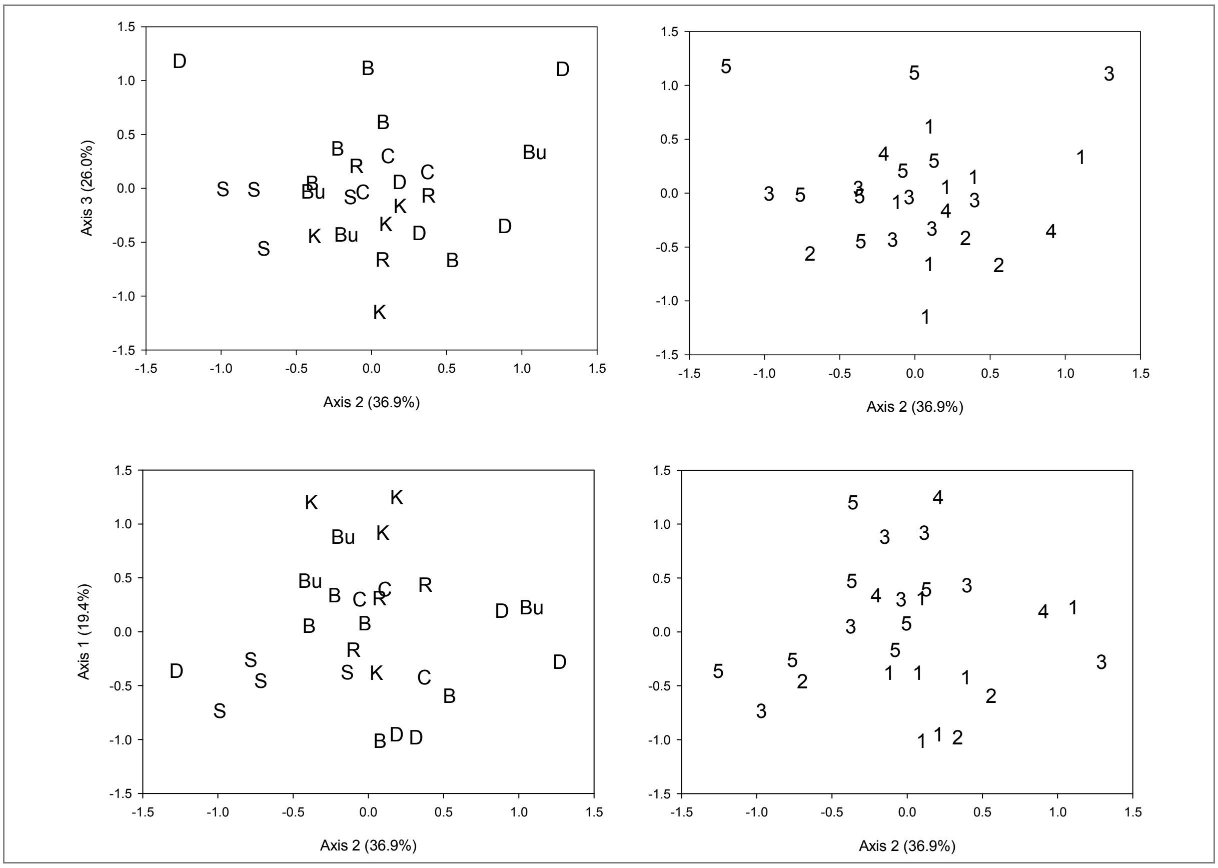

Figure 3.8: nMDS ordination of invertebrate samples from leaf litter. Site overlays for Axis 2 vs. Axis 3 (upper) and Axis 2 vs. Axis 1 (lower) are shown the left; sample overlays for the five monthly samples are shown for the same ordinations on the right. Abbreviations for lagoons as in Figure 2.1 (not all shown in these figures if no relevant samples). Sample times are: 1: May 2008; 2: June 2008; 3: Sept 2008; 4: Nov 2008; and 5: May 2009.

Table 3.4: Invertebrate taxa with significant correlations (p<0.05) with nMDS ordination axes in the litter samples.

	Axis 1			Axis 2			Axis 3	
% of variance	19.4			36.9			26	
Acarina	0.741	**	Culicid	0.653	**	Amerian	0.628	**
Oligo	0.734	**	Pleid	0.618	**	Chironom	0.509	**
ZygopSp	0.716	**	Pyralid	0.567	**	Protoneur	-0.462	*
Elmid	0.638	**	Cerato	0.563	**	Notonect	-0.499	**
Ostracod	0.611	**	Baetid	0.511	**	Atyid	-0.573	**
Caenid	0.597	**	Atyid	0.374	*	Calamo	-0.606	**
Triplect	0.578	**	Gerrid	0.372	*			
Calamo	0.556	**	Ecnomid	-0.51	**			
Protoneur	0.455	*	Triplect	-0.543	**			
Veliid	0.396	*						
Gerrid	0.374	*						
HemUrLib	-0.566	**						

p	0.05*	0.01**
Critical r	0.381	0.487

Table 3.5: MRPP results comparing sites, based on litter invertebrate assemblages. Values are for p, between sites, with values < 0.05 in boldface. Overall p = 0.002. Site abbreviations as in Table 2.1.

Sites	K	S	D	Bu	R	C
B	0.013	0.004	0.425	0.044	0.128	0.244
K		0.010	0.016	0.301	0.071	0.051
S			0.007	0.010	0.022	0.012
D				0.167	0.144	0.161
Bu					0.790	0.313
R						0.521

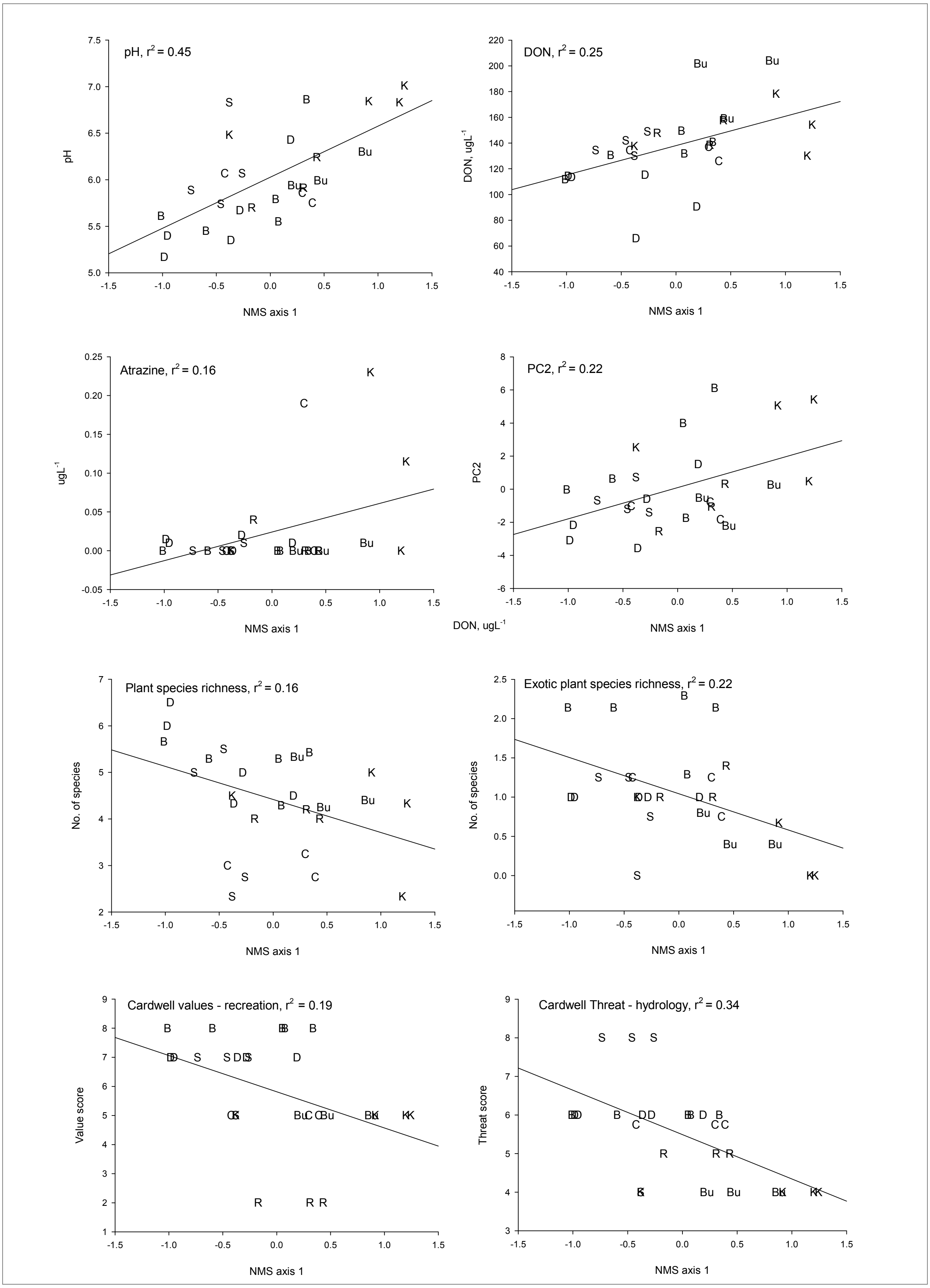

Figure 3.9: Invertebrates: relationships between nMDS Axis 1, selected environmental variables and Cardwell Values and Threats assessments. Site abbreviations as in Figure 2.1.

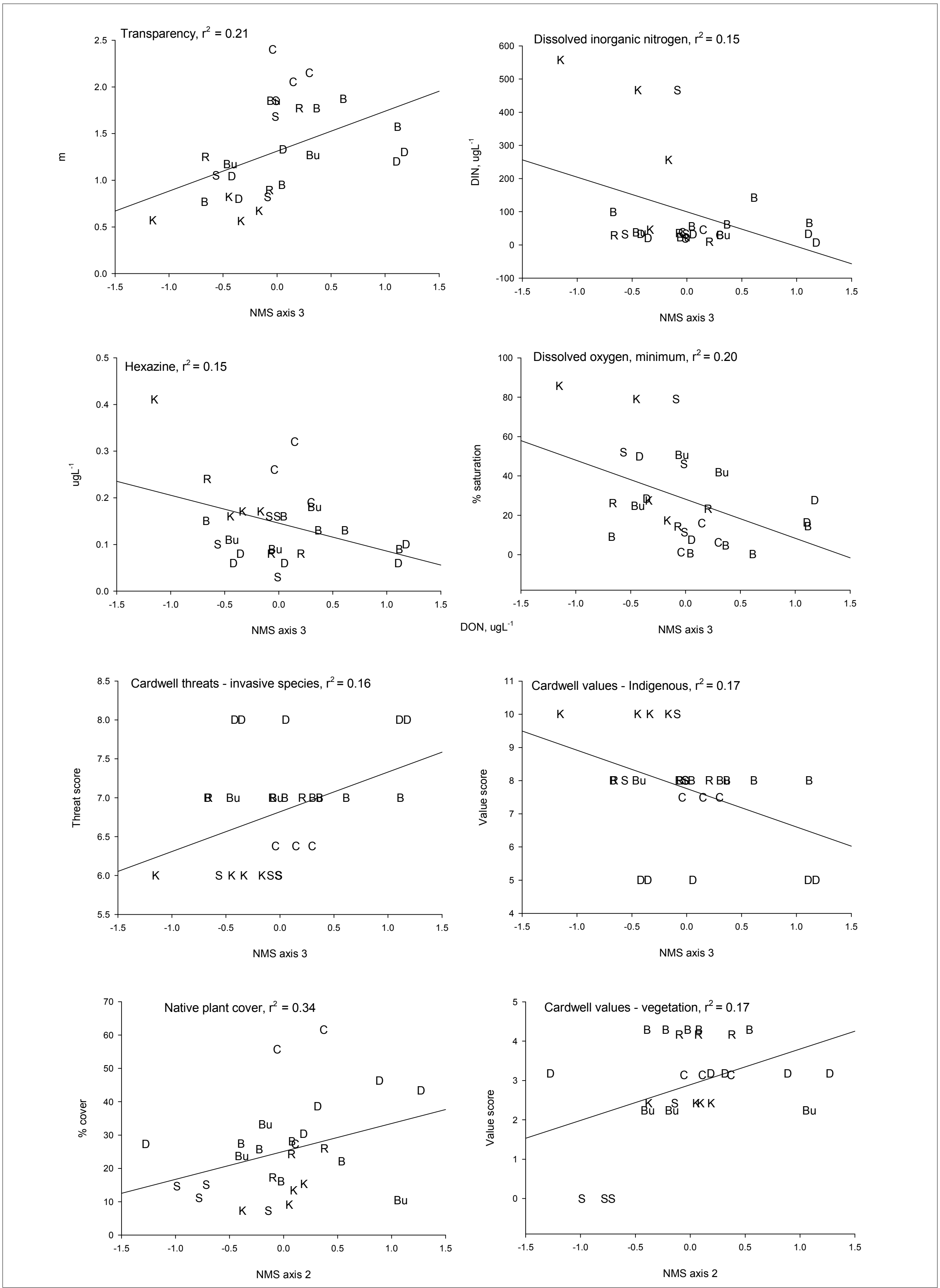

Figure 3.10: Invertebrates: relationships between nMDS axes 2 and 3, selected environmental variables and Cardwell Values and Threats assessments. Site abbreviations as in Figure 2.1.

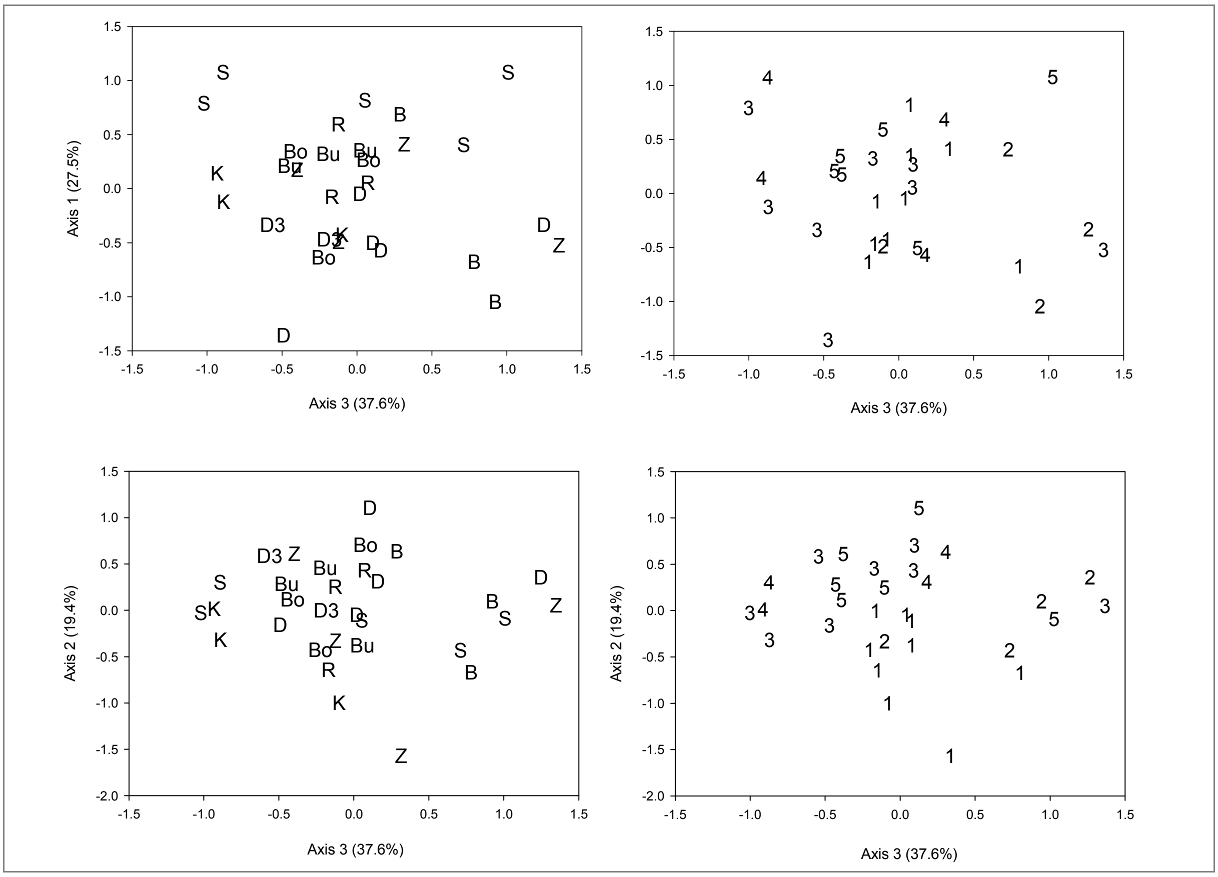

Figure 3.11: nMDS ordination of invertebrate samples from macrophytes. Site overlays for Axis 3 vs. Axis 1 (upper) and Axis 3 vs. Axis 2 (lower) are shown the left; sample overlays for the five monthly samples are shown for the same ordinations on the right. Site and time abbreviations as in Figure 2.1.

Table 3.6: Invertebrate taxa with significant correlations ($p<0.05$) with nMDS ordination axes in macrophyte samples.

	Axis 1			Axis 2			Axis 3	
% of variance	27.5			19.4			37.6	
Coenag	0.650	**	ZygopSp	0.647	**	Chironom	0.394	*
LeptocSpp	0.611	**	Caenid	0.556	**	HemUrLib	0.389	*
Ostracod	0.544	**	Cerato	0.529	**	Corixid	0.357	*
Atyid	0.538	**	Oligo	0.445	*	LeptocSpp	-0.394	*
ZygopSp	0.534	**	HemUrLib	0.443	*	Hydraeni	-0.400	*
Hydraeni	0.447	*	Acarina	0.439	*	Notonect	-0.436	*
Triplect	0.389	*	Pleid	0.408	*	Culicid	-0.480	**
Chironom	0.383	*	Coenag	0.386	*	Ecnomid	-0.482	**
Ecnomid	0.380	*	Calamo	-0.402	*	Acarina	-0.511	**
Oligo	0.358	*	Notonect	-0.565	**	Oligo	-0.534	**
Baetid	-0.549	**				Pleid	-0.535	**
Corixid	-0.645	**	p	0.05*	0.01**	Ostracod	-0.606	**
			Critical r	0.355	0.456	Pyralid	-0.689	**

Table 3.7: MRPP results comparing sites, based on macrophyte invertebrate assemblages. Values are for p, between sites, with values < 0.05 in boldface. Overall p = 0.002.

Sites	K	S	D	Z	Bu	R	D3	Bo
B	0.039	0.052	0.064	0.380	0.128	0.194	-	0.254
K		0.029	0.009	0.131	0.102	0.307	<0.001	0.271
S			0.003	0.156	0.188	0.178	0.041	0.150
D				0.612	0.009	0.105	0.015	0.370
Z					0.521	0.503	0.171	0.551
Bu						0.887	<0.001	0.895
R							<0.001	0.907
D3								<0.001

Adult dragonfly R. PEARSON

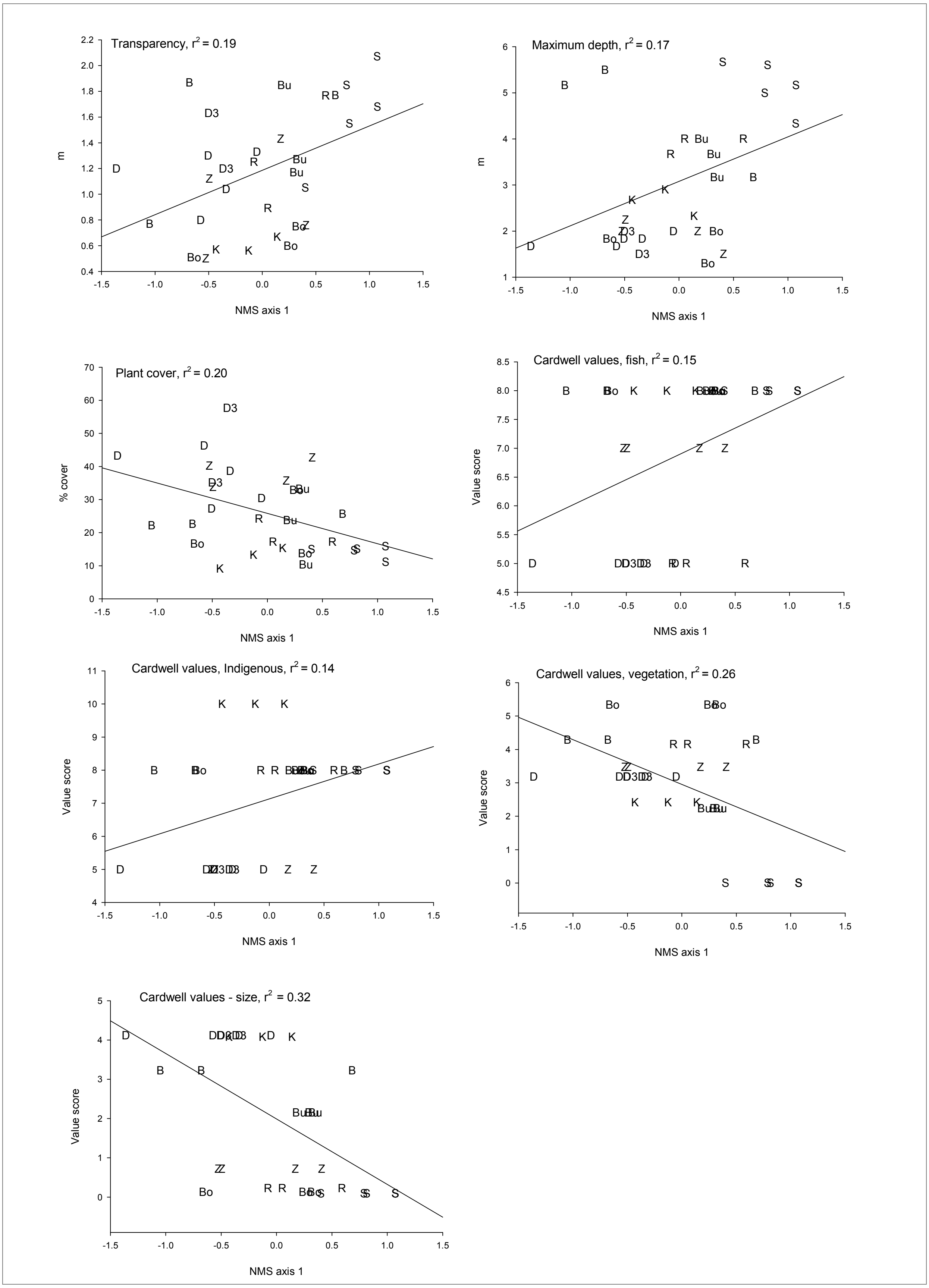

Figure 3.12: Relationships between nMDS Axis 1, selected environmental variables and Cardwell Values and Threats assessments. Site abbreviations as in Figure 2.1.

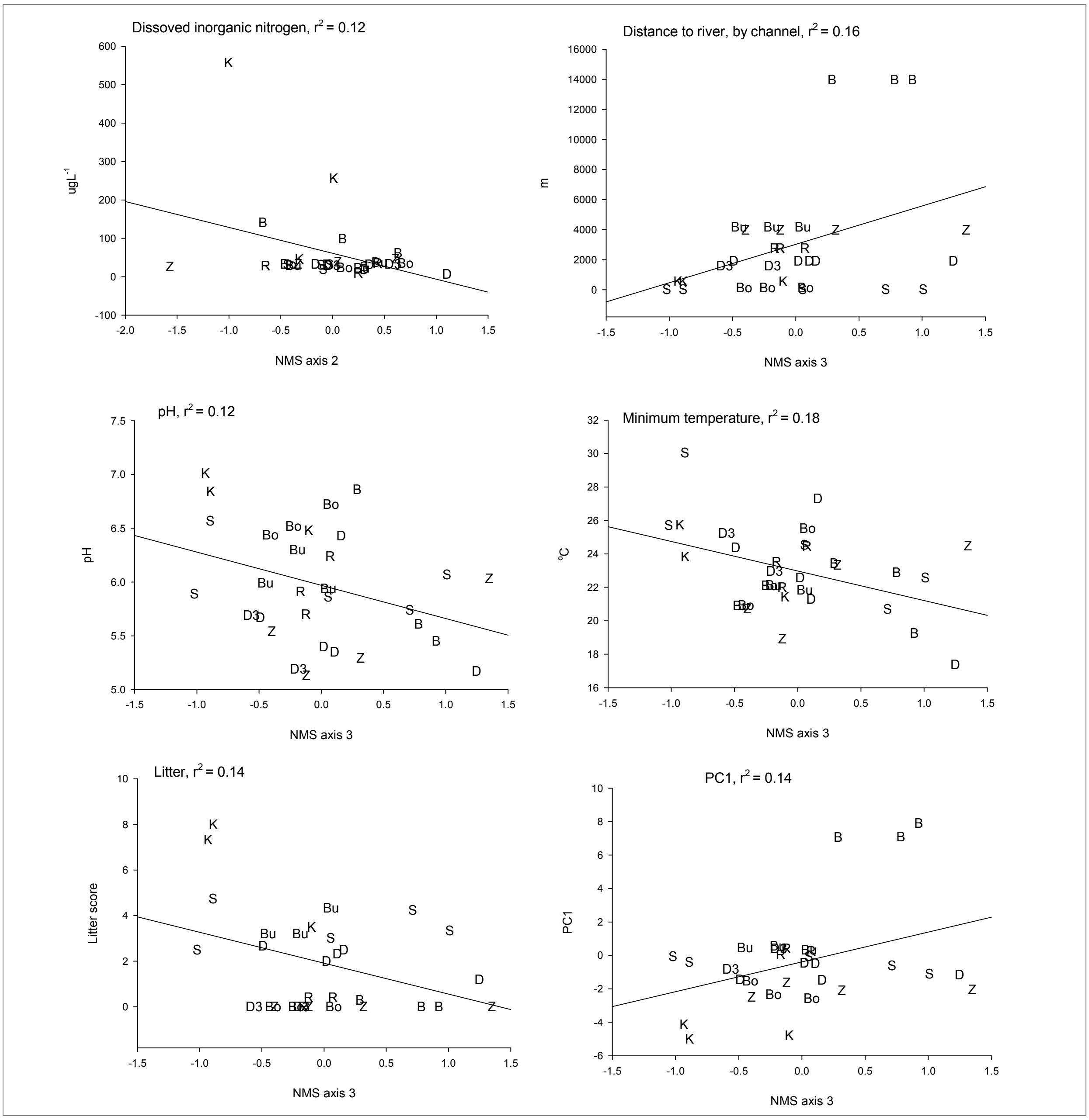

Figure 3.13: Relationships between nMDS axes 2 and 3; selected environmental variables and Cardwell Values and Threats assessments. Site abbreviations as in Figure 2.1.

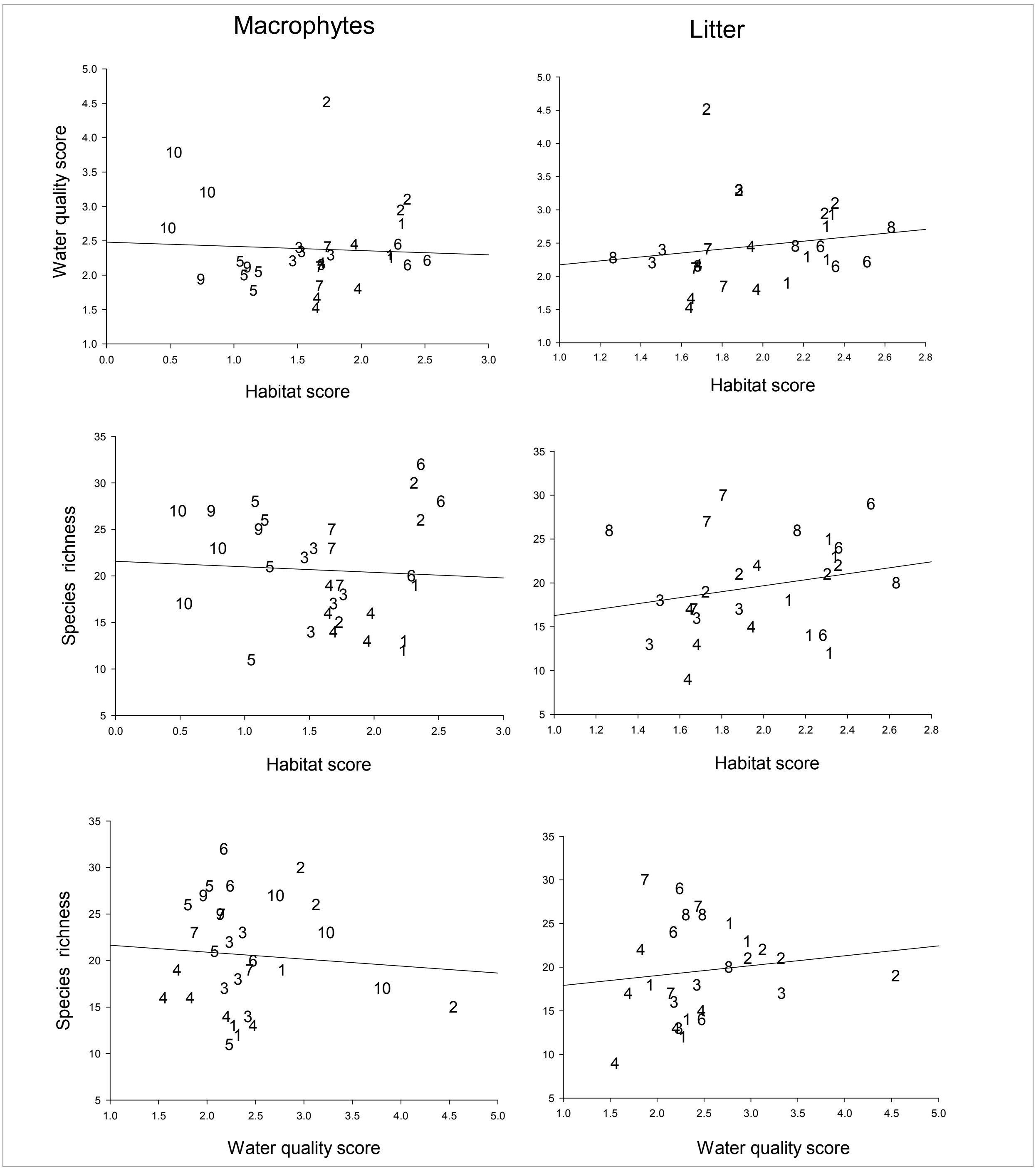

Figure 3.14: Relationships between composite habitat and water-quality scores and invertebrate species richness for samples from macrophytes (left-hand figures) and samples from litter (right-hand figures). Numerals represent sites; $r^2 < 0.05$ in all cases, $p >> 0.05$.

3.4 Fish

3.4.1 Fish fauna of the lagoons

In all 16,378 fish from 22 species (21 native 1 alien) representing 15 families were collected over the three comprehensive surveys conducted in May 2008, September 2008 and May 2009 (Table 3.8). Six of the native species require access to estuarine or marine areas for spawning and/or larval development with the life history of the remaining 15 species completed entirely in fresh water. The contribution by species with each life history mode to total species richness varied among lagoons. The two lagoons located closest to the mouth of the Tully and Murray Rivers contained a greater proportion of species that need access to the estuarine/marine environment (52% and 47% for Barrett's and Carroll's, respectively) than lagoons located further upstream.

Kyambul Lagoon contained a comparatively large proportion (45%) of estuarine dependent species despite being located the furthest upstream of all the lagoons. This pattern may be driven by the typology of Kyambul lagoon (providing riverine habitat as opposed to it being a 'true' lacustrine or palustrine floodplain wetland) and the permanent connection that exists between the lagoon, the stream network and the Murray River estuary. The proportion of total species richness that was exclusively freshwater species was greatest in two of the three lagoons positioned furthest from either of the two main rivers: Digman's (79%), Zamora's (75%) and Digman's 3 (72%).

Total species richness ranged from 15-17 species across the ten lagoons with Barrett's, Bunta and Boongaray all containing 17 species (Table 3.8). Nine species occurred across all ten lagoons with seven of these species completing their entire life history in freshwater. Four species were restricted to two lagoons or less – *Lates calcarifer, Hephaestus fuliginosus, Notesthes robusta* and *Melanotaenia maccullochi*. The low occurrence of barramundi (*L. calcarifer*) and sooty grunter (*H. fuliginosus*) may have been an artefact of the limited ability of the sampling gear to capture these large and highly mobile species. Bullrout (*Notesthes robusta)* prefer stream habitat and its capture in a single lagoon may have been incidental. It is notable that only a small number of Macculloch's rainbowfish (*M. maccullochi*), a species of conservation significance (Pusey *et al.* 2004), were recorded in two lagoons of the study area despite this species showing a strong preference for highly vegetated, still and slow flowing habitat that is a feature of the water bodies on the floodplain of the Tully-Murray system. A single, widely distributed, alien species, the platy (*Xiphophorus maculatus*) was collected in nine of the ten lagoons in the study area.

Five species – *H. compressa*, *Hypseleotris* sp. 1; *Craterocepehalus stercusmuscarum, Denariusa bandata*, *M. s. splendida* (in decreasing order of abundance) – together comprised 97% of the total fish abundance over the entire sampling period (May 2008; September 2008 and May 2009). These species were typically most abundant in all of the ten lagoons although a few individual species were extremely (i.e. disproportionately) abundant in one or two lagoons. For example, the catfish (*N. ater)* comprised 54% of the total catch in Digman's lagoon despite contributing only 0.33% to the entire fish assemblage. These six species, together with *Glossomia aprion*, *Anguilla reinhardtii*, *Mogurnda adspersa*, and *X. maculatus* were most frequently encountered, occurring in 10% or more of all sampling replicates.

Table 3.8: Standardised catch per unit effort (sum of the combined fyke net and electrofishing catches) in each of ten lagoons over the three comprehensive surveys (May 2008; September 2009 and May 2009). Lagoons are arranged in longitudinal order from downstream to upstream (based on channel distance from the river mouth). The number of replicates for each sampling method is provided. Method: El = electrofishing, Fy = fyke netting. An asterisk (*) denotes that a species was recorded in the electrofishing sample only. The larval habitat of each species is presented (based on information in Pusey *et al.* 2004 and Godfrey, unpub. data). EST/MAR = Estuarine or marine, FW = freshwater. Also shown is the proportional contribution by each reproductive mode to the species richness in each fishing replicate averaged over all replicates for the entire study period.

Family / Species	Larval habitat	*n* reps	Barrett's	Carroll's	Digman's 3	Digman's	Bunta	Boongaray	Selby's	Zamora's	Raccanello's	Kyambul	Freq. of occurrence (%)
		El	70	60	60	90	90	60	90	60	90	75	
		Fy	20	24	24	24	24	24	24	23	24	24	
Anguillidae													
Anguilla reinhardtii Steindachner, 1867	EST/MAR		9.52	2.96	5.29	4.66	2.85	2.54	6.35	6.03	12.53	1.77	13.88
Anguilla obscura Günther 1872	EST/MAR		-	-	0.27	0.66	-	-	-	0.27	-	-	0.51
Clupeidae													
Nematolosa erebi Günther, 1868	FW		-	-	-	-	-	3.04	-	-	0.72	0.04	0.82
Plotosidae													
Neosilurus ater (Perugia, 1894)	FW		8.30	3.64	7.45	29.19	0.96	1.69	0.84	1.00	0.40	0.22	11.63
Neosilurus hyrtlii (Steindachner, 1867)	FW		1.64	0.73	4.43	10.30	0.57	0.25	0.33	1.60	0.37	0.93	5.71
Atherinidae													
Craterocephalus stercusmuscarum (Günther, 1867)	FW		9.60	1.25	2.16	0.61	32.57	223.49	3.39	16.80	32.25	67.08	10.00
Melanotaenidae													
Melanotaenia maccullochi (Ogilby, 1915)	FW		0.77	-	-	-	-	3.85	-	-	-	-	0.41
Melanotaenia splendida (Peters, 1866)	FW		51.51	0.50	3.71	6.87	0.47	59.19	4.38	22.63	5.15	7.63	18.67
Pseudomugildae													
Pseudomugil gertrudae Weber 1911	FW		27.23	-	0.15	0.03	0.13	5.17	0.00	0.75	0.03	0.03	2.45
Synbranchidae													
Ophisternon cf. gutturale* (Richardson, 1845)	FW		-	-	-	-	0.07	-	0.03	-	-	0.30	1.02
Scorpaenidae													
Notesthes robusta (Günther, 1860)	EST/MAR		-	-	-	-	-	0.27	-	-	-	-	0.10

Family / Species	Larval habitat	*n* reps	Barrett's	Carroll's	Digman's 3	Digman's	Bunta	Boongaray	Selby's	Zamora's	Raccanello's	Kyambul	Freq. of occurrence (%)
Chandidae													
Ambassis agassizii (Steindachner, 1867)	FW		2.12	0.54	6.51	3.63	0.27	38.08	1.97	8.35	19.59	13.43	6.43
Denariusa bandata (Whitley, 1948)	FW		39.18	4.13	23.85	27.54	1.72	28.20	9.53	27.71	19.62	-	16.12
Centropomidae													
Lates calcarifer (Bloch, 1790)	EST/MAR		3.32	0.05	-	-	-	-	-	-	-	-	0.41
Terapontidae													
*Hephaestus fuliginosus** (Macleay, 1883)	FW		-	-	-	-	0.03	-	-	-	-	-	0.41
Apogonidae													
Glossomia aprion (Richardson, 1842)	FW		2.78	4.98	11.93	10.56	2.54	12.62	17.71	8.16	32.96	2.93	25.00
Gobiidae													
Redigobius bikolanus (Herre, 1935)	EST/MAR		0.35	0.05	-	-	0.27	-	1.10	-	2.10	2.23	3.16
Eleotridae													
Giurus margaritacea (Valenciennes, 1837)	cf. FW		0.05	0.23	0.20	0.30	0.17	0.15	0.13	0.15	-	0.05	3.16
Hypseleotris compressa (Krefft, 1864)	EST/MAR		7693	417	258	93	186	1523	1144	702	288	826	68.16
Hypseleotris sp. 1	FW		29.22	167	40.4	7.0	170	369	223	99.8	680	221	42.9
Mogurnda adspersa (Castelnau, 1878)	FW		23.66	0.10	3.28	8.74	1.20	0.30	0.30	6.50	1.05	0.33	17.65
Poeciliidae													
Xiphophorus maculatus (Günther, 1866)	FW		5.77	28.2	2.81	2.78	20.04	2.61	6.66	1.92	2.29	0.00	10.41
Species richness			17	15	15	15	17	17	16	16	15	15	
Proportion (%) EST/MAR species			52.97	47.4	27.86	20.81	41.10	35.29	41.63	26.49	25.24	44.9	
Proportion (%) FW species			47.03	52.5	72.12	79.19	58.90	64.71	58.37	73.52	74.75	55.03	

3.4.2 Lagoon differences in fish assemblage characteristics

Significant differences in metrics characterising fish assemblage structure were limited in number when comparisons were made among all ten lagoons (Table 3.9). Firstly, significant differences in fish species richness were detected among lagoons. Although the SNK test was unable to differentiate lagoons based on species richness, there is some suggestion that species numbers were elevated in Digman's Lagoon and depressed in Barrett's and Carroll's lagoons (Figure 3.15a). There were no significant differences in total abundance of fish among lagoons although inspection of the data (presented in Figure 3.15b) suggests that fish numbers were lower in Bunta and the two Digman's lagoons and higher in Barrett's Lagoon.

The difference in species richness among lagoons was reinforced and differences in additional metrics became apparent when comparisons were confined to those four lagoons sampled on more than three occasions (Table 3.10). Species richness was significant higher in Digman's Lagoon than in all other lagoons (Figure 3.15c). Mean evenness was significantly higher in Digman's Lagoon than in Barrett's Lagoon but all other comparisons were not significantly different (Figure 3.15d). Estimates of total abundance were variable, contributing to the lack of a statistically significant difference among lagoons. However, the comparatively low significance level (i.e. p=0.0587) and the pattern in the data presented in Figure 3.15e suggest that difference in total abundance existed between Barrett's and Digman's lagoons. Significant differences in the proportion of species abundance contributed by alien species in Kyambul and Digman's lagoons became evident over the extended period of sampling (Figure 3.15f).

Table 3.9: F-values and their associated levels of significance for two-way ANOVAs on freshwater fish assemblage characteristics measured in ten lagoons over the three comprehensive surveys in May 2008, September 2008 and May 2009. * P<0.05, ** P <0.01, *** P <0.001.

Variable	Source of variation					
	d.f.	Lagoon	d.f.	Survey	d.f.	Survey x lagoon
Species richness	9,60	2.41*	3,60	$1.09^{n.s}$	18,60	$0.34^{n.s}$
Evenness	9,60	$1.03^{n.s}$	3,60	$2.07^{n.s}$	18,60	$0.26^{n.s}$
Total abundance	9,60	$1.01^{n.s}$	3,60	2.72*	18,60	$0.30^{n.s}$
Proportion alien species	9,60	$1.58^{n.s}$	3,60	$0.68^{n.s}$	18,60	$1.21^{n.s}$

Post hoc multiple comparison tests		
Factor	Variable	Comparisons
Lagoon	Species richness	-
Survey	Total abundance	May08^{A} Sep08^{AB} May09^{B}

Table 3.10: F-values and their associated levels of significance for two-way ANOVAs on freshwater fish assemblage characteristics measured in four lagoons over the extended surveys in May 2008, July 2008, September 2008, November 2008 and May 2009. * P<0.05, ** P <0.01, *** P <0.001.

Variable	Source of variation					
	d.f.	Lagoon	d.f.	Survey	d.f.	Survey x lagoon
Species richness	3;40	4.33***	4;40	0.39n.s	12;40	0.54n.s
Evenness	3;40	4.53***	4;40	0.77n.s	12;40	0.26n.s
Total abundance	3;40	2.57n.s	4;40	1.04n.s	12;40	0.99n.s
Proportion alien species	3;40	3.02*	4;40	0.83n.s	12;40	1.05n.s

Post hoc multiple comparison tests

Factor	Variable	Comparisons
Lagoon	Species richness	Kya^{A} Bar^{A} Sel^{A} Dig^{B}
	Evenness	Bar^{A} Kya^{AB} Sel^{AB} Dig^{B}
	Proportion alien species	Kya^{A} Bar^{AB} Sel^{AB} Dig^{B}

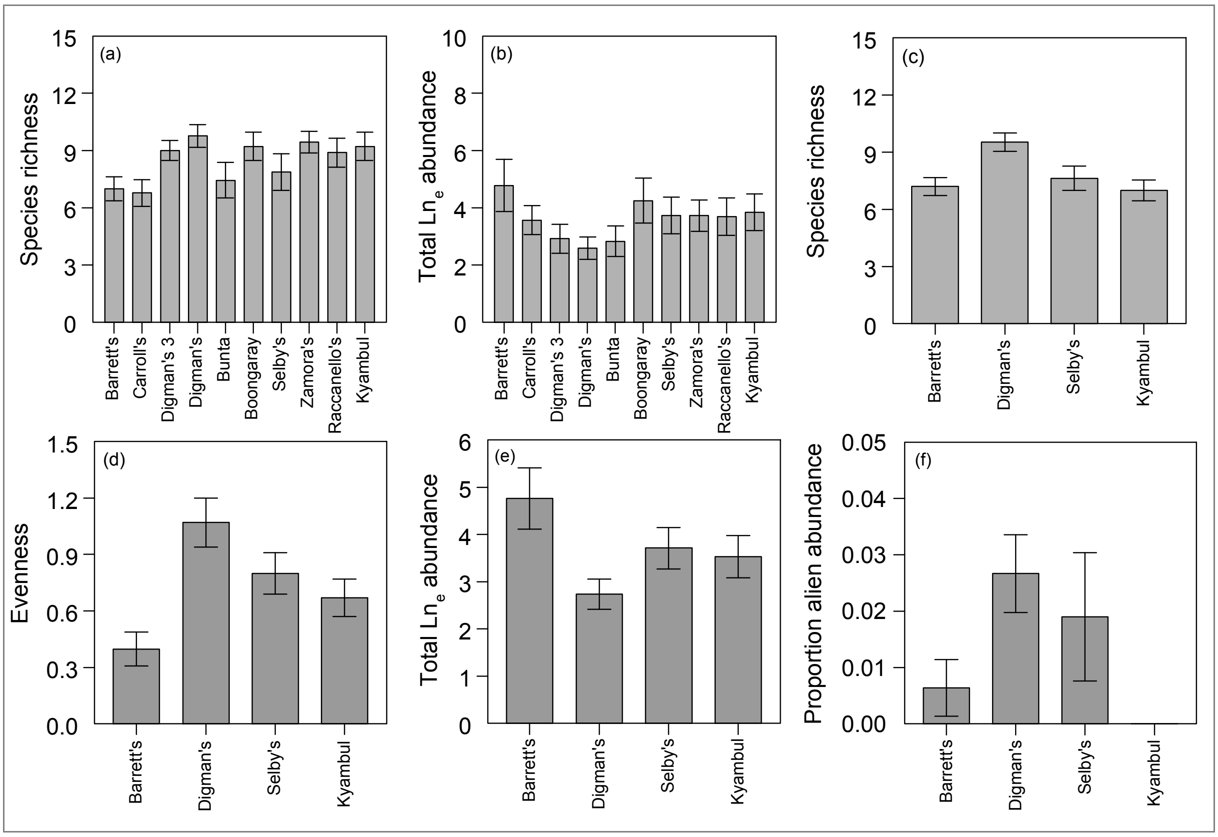

Figure 3.15: Average (± s.e.) values for fish assemblage characteristics for ten lagoons that were sampled comprehensively (a and b) and the four lagoons (d-f) sampled extensively.

3.4.3 Lagoon differences in fish assemblage structure

Significant variation in fish assemblage structure was detected among lagoons (ANOSIM: Global $R = 0.534$; $P < 0.001$). These differences are illustrated in the nMDS ordination where all lagoons are separated in the space defined by 'x' and 'y' axes although the extent of this separation varied among lagoon combinations (Figure 3.16a). Barrett's Lagoon, located closest to estuary, separated most clearly from all other lagoons on the 'y' axis. This lagoon contained a complex mixture of species dependent on estuarine or marine habitats for larval production, such as *H. compressa*, *A. reinhardtii, L. calcarifer* as well as some exclusively freshwater species that are associated with abundant vegetation cover including *M. s. splendida*, *Pseudomugil gertrudae, M. maccullochi* and *M. adspersa* (Figures 3.16b,c). The two lagoons (Digman's and Digman's 3) located most distant from either of the two main rivers lie in a similar position in ordination space (i.e. middle right region). Such lagoons were dominated by a suite of exclusively freshwater dependent species (*N. ater*, *N. hyrtlii* and *Giurus margaritacea*) with habitat characterised by a high proportion of submerged vegetation cover.

Bunta Lagoon was positioned at the opposite extreme to Barrett's on the 'y' axis and was characterised by a group of freshwater dependent species (*Ophisternon* cf. *gutturale*, *H. fuliginosus* and *X. maculatus*). It is notable that Carroll's Lagoon was located relatively close to the river mouth and contained a comparatively high proportion of species dependent on estuarine/marine habitats for larval proportion yet it also comprised a large population of the introduced *X. maculatus* which accounted for it aligning in ordination space with Bunta Lagoon (Figure 3.16a). Both these lagoons possessed a high proportion of leaf litter habitat which contributed to the strong correlation between the large abundance of *X. maculatus* and leaf-litter habitat within each lagoon. Kyambul was also positioned in the upper region on the 'y' axis and was most distant of all lagoons from the estuary. It contained a group of exclusively freshwater dependent species (*Hypseleotris* sp. 1; *O*. cf. *gutturale*); however, it also supported a comparatively high abundance of *Redigobius bikolanus,* a goby which requires access to estuarine areas during its life history. This access may be afforded by the permanent stream connection that exists between Kyambul Lagoon and the Murray River and its estuary.

Lagoon position also correlated with aspects of water chemistry including some features that suggest anthropogenic influences from the change in land use that has occurred in the surrounding catchment (Figure 3.16d). High concentrations of herbicides and/or dissolved inorganic nitrogen were aligned with Kyambul, Raccanello's and Boongaray lagoons positioned to the left on the 'x' axis although these chemicals appeared to have had no effect on water quality attributes (e.g. by lowering the concentration of dissolved oxygen) that would be harmful to aquatic biota. The detection of strong correlations between these anthropogenic water chemistry influences and fish assemblage structure more likely represents co-variation as a result of the effect of a suite of landscape and habitat differences and their influence on fish assemblage structure associated with location of spawning and recruitment grounds.

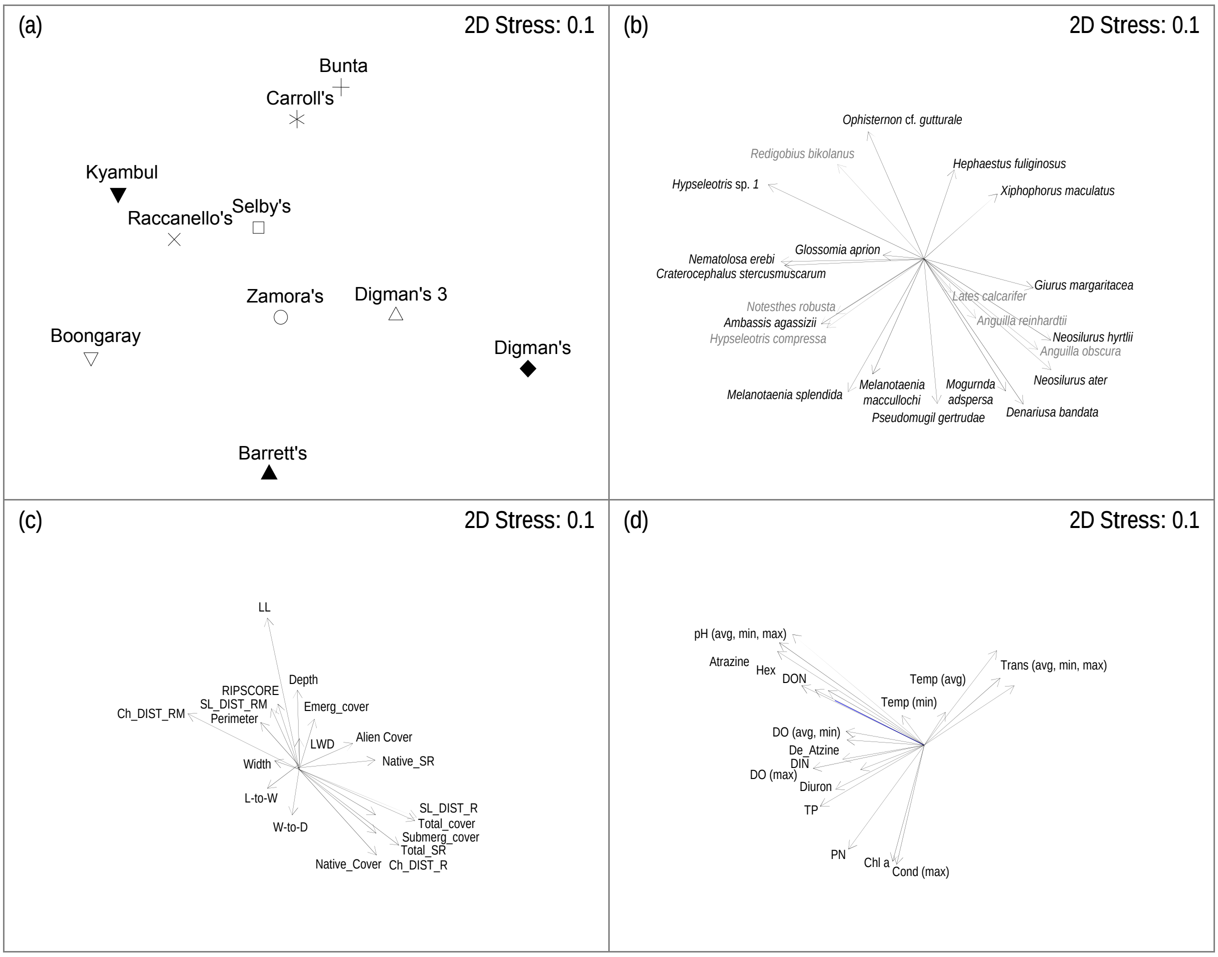

Figure 3.16: Spatial variation in fish assemblage structure in ten lagoons in the Tully-Murray river system based on nMDS ordination of lagoons by species abundance (Ln transformed). Each point in plot (a) represents the centroid of the three fish surveys conducted in each lagoon. Dominant correlations between species abundance and lagoon position in the ordination space are displayed in plot (b). Estuarine/marine dependent species are shaded grey. Dominant correlations between landscape factors, gross lagoon morphology and aquatic microhabitat and fish assemblage in each lagoons are displayed in plot (c). Dominant correlations between water chemistry variables and fish assemblage structure in each lagoon are displayed in plot (d). Abbreviations explained in Table 2.2.

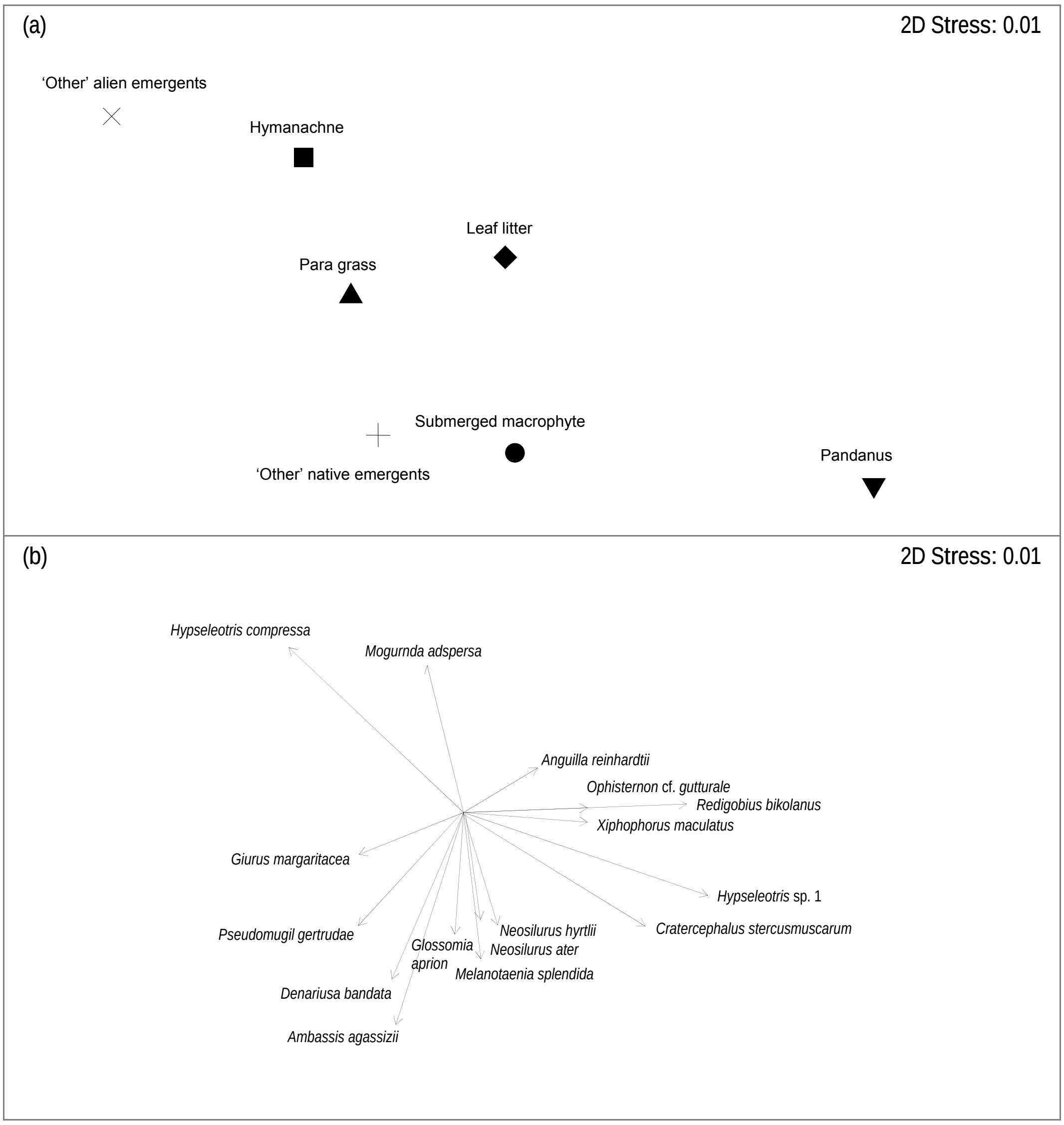

Figure 3.17: Spatial variation in fish-habitat association based on nMDS ordination of microhabitats by species abundance (Ln transformed). Each point in plot (a) represents fish assemblage composition associated with each microhabitat type averaged across all electrofishing replicates performed in each habitat over the three comprehensive surveys. Dominant correlations between fish species abundance and habitat position in ordination space are displayed in plot (b).

3.4.4 Habitat structure and fish associations

The previous section illustrated the influence of landscape position, habitat and water quality in contributing to differences in the composition of the fish assemblage among lagoons. The finer grain of microhabitat structure also influenced fish assemblage structure in lagoons. Significant differences in fish assemblage structure were detected when comparing fish assemblage composition among the seven major microhabitat groups that occurred most consistently across all the lagoons (Global $R = 0.037$; $P < 0.001$).

It is evident from this ordination (Figure 3.17) that the composition of the fish assemblage associated with the alien emergent vegetation – *H. amplexicaulis* and 'other' alien taxa (a combination of *Isolepis* sp., *Echinochloa* sp. and *Cyperus* sp.) formed a distinct group in the space defined by 'x' and 'y' axes (Figure 3.17a). Such groups were characterised by the ubiquitous gudgeon, *H. compressa* and *M. adspersa* (Figure 3.17b). The fish assemblage associated with leaf litter was positioned in the middle region of the ordination plot and was dominated by a suite of benthic-dwelling species including the native *A. reinhardtii, R. bikolanus*, *O*. cf. *gutturale* and the alien *X. maculatus.* The assemblages associated with para grass were situated in comparatively close proximity to leaf litter and *H. amplexicaulis* habitat types and were represented by a combination of species recorded across these two habitats.

Fish assemblages associated with the two native emergent vegetation groups were both positioned in the lower region of the ordination and were separated clearly on the 'x' axis. The assemblages associated with the range of native emergent sedges and herbs (*Eleocharis* sp., *Lepironia articulata*, *Persicaria* sp. *Schoenoplectus mucronatus*) that occurred across lagoons were dominated by a diverse mixture of native fish species including *Ambassis agassizii*, *P. gertrudae*, *M. s. splendida, G. margaritacea* and *Glossomia aprion*. The assemblage associated with *Pandanus* was less speciose and dominated by *Hypseleotris*. sp. 1; and to a lesser extent, *C. stercusmuscarum*. Habitats formed by submerged vegetation lay in between these two vegetation groups in ordination space as defined by the 'x' axis with their fish assemblages comprising a mixture of species typical of both native emergent vegetation groups as well as some species that were closely associated with submerged vegetation, including the catfishes *N. ater* and *N. hyrtlli*.

These analyses suggest the structure of the aquatic habitat is pivotal in driving differences in the composition of fish assemblages across lagoons due to the differences in the contribution of habitat groups to the overall habitat availability (see above) in each lagoon. The fish assemblages that are associated with the alien emergents (particularly *H. amplexicaulis* and a range of other perennials) were generally species poor and structurally very different from those that occupied the equivalent native form of emergent sedges and herbs.

3.4.5 Temporal patterns of fish occurrence

Approximately 19,762 fish were recorded from the ten lagoons over the extended period of sampling (May 2008; July 2008; September 2008; November 2008 and May 2009) (Table 3.11). This total included ~3,383 fish recorded during the two additional surveys in July and November 2008. No additional species were recorded during these two intervening surveys beyond those that were collected over the three comprehensive surveys conducted in May 2008; September 2008 and May 2009. An early developmental stage (i.e. larva or young-of-the-year) was represented in 16 of the 22 species captured in the floodplain lagoons over the entire sampling period. Those species not represented by their early life history stages in the lagoons included two freshwater spawning species – *M. maccullochi* and *H. fuliginosus* – and four species dependent on estuarine and marine habitats during their early life history – *A. obscura*, *L. calcarifer, N. robusta* and *R. bikolanus.*

Considerable variation in total abundance was observed among surveys (Table 3.11). Temporal variability in total fish catch fluctuated by an order of ~2 over the extended period of sampling with the largest catches recorded in May 2009 and the smallest catches during the corresponding sampling period one year earlier (May 2008). The contribution of the early developmental stage to the entire fish assemblage was ~3%, which was evenly proportioned between the freshwater and obligate estuarine/marine life history groups. The relatively small contribution of the early developmental assemblage to total fish abundance was accounted for by the large contribution of the juvenile/adult form of *H. compressa* and *Hypseleotris* sp. 1 to total abundance (~92%). The nursery habitat for both these species (i.e. the salt water of the near shore marine environment and the open water habitat in the lagoons, respectively) was not surveyed during in this investigation, and thus under-represented the contribution of the entire early life-history assemblage to the total fish assemblage.

Fyke nets for fish capture, in position R. PEARSON

Retrieval of fyke nets R. PEARSON

Table 3.11: Standardised catch per unit effort (sum of the combined fyke net and electrofishing catches) collectively from ten lagoons over the extended sampling period (May 2008, July 2008, September 2009, November 2008 and May 2009). The number of replicates for each sampling method is provided. Method: El = electrofishing, Fy = fyke netting. The larval habitat of each species is presented (based on information in Pusey *et al.* 2004 and P. Godfrey, unpub. data). EST/MAR = Estuarine or marine, FW = freshwater. Total (relative) abundance was calculated by dividing the total catch in each method by the respective number of fishing replicates performed by each method on each sampling occasion. Also shown is the proportional contribution by each species/ development stage to the total collected during the study.

Family / Species	Developmental stage / size category	Larval habitat	*n* reps	May 2008	July 2008	September 2008	November 2008	May 2009	%
			El	260	120	245	120	240	
			Fy	80	29	80	32	84	
Anguillidae									
Anguilla reinhardtii Steindachner, 1867	Glass eel	EST/MAR		-	-	-	-	-	-
	<100 mm			0.07	3.30	-	0.03	-	0.02
	100-300 mm			5.40	6.91	6.93	7.62	6.18	0.16
	301-600 mm			9.95	2.00	16.23	3.54	9.75	0.21
	>600 mm			-	-	-	-	-	-
Anguilla obscura Günther 1872	Glass eel	EST/MAR		-	-	-	-	-	-
	<100 mm			-	-	-	-	-	-
	100-300 mm			-	-	-	-	-	-
	301-600 mm			-	-	0.27	-	0.68	<0.01
	>600 mm			0.24	-	-	-	-	<0.01
Clupeidae									
Nematolosa erebi Günther, 1868	Post-flexion	FW		0.49	-	-	-	-	<0.01
	<100 mm			1.24	-	-	-	1.03	0.01
	>100 mm			-	0.50	0.86	0.28	0.18	<0.01
Plotosidae									
Neosilurus ater (Perugia, 1894)	<75 mm	FW		2.11	0.78	0.46	-	9.07	0.06
	76-150 mm			14.42	13.05	5.69	4.17	15.94	0.27

Family / Species	Developmental stage / size category	Larval habitat	*n* reps	May 2008	July 2008	September 2008	November 2008	May 2009	%
	>150 mm SL			3.31	0.25	1.41	1.55	1.28	0.04
Neosilurus hyrtlii (Steindachner, 1867)	<75 mm	FW		1.36	0.78	0.76	0.03	1.97	0.02
	76-150 mm			6.98	4.19	2.56	3.00	5.91	0.11
	>150 mm SL			0.64	-	-	0.67	0.97	0.01
Atherinidae									
Cratercephalus stercusmuscarum (Günther, 1867)	Pre-flexion	FW		0.05	-	0.07	-	-	<0.01
	Flexion			-	-	-	-	-	-
	Post-flexion			6.00	1.07	0.15	0.45	-	0.04
	Juv/adult			116.65	11.66	150.98	6.71	115.30	2.04
Melanotaenidae									
Melanotaenia maccullochi	Pre-flexion	FW		-	-	-	-	-	-
	Flexion			-	-	-	-	-	-
	Post-flexion			-	-	-	-	-	-
	Juv/adult			-	-	-	-	4.62	0.02
Melanotaenia splendida (Peters, 1866)	Pre-flexion	FW		-	1.27	-	0.03	-	<0.01
	Flexion			0.03	1.20	0.20	-	-	<0.01
	Post-flexion			1.41	0.31	2.85	0.47	0.27	0.02
	Juv/adult			33.50	8.72	56.00	1.28	67.77	0.80
Pseudomugildae									
Pseudomugil gertrudae (Weber, 1911)	Post-flexion	FW		0.03	-	0.05	-	-	<0.01
	Juv/adult			0.52	2.10	0.20	-	32.74	0.18
Synbranchidae									
Ophisternon cf. *gutturale** (Richardson, 1845)	<100 mm	FW		0.03	-	-	0.17	-	<0.01
	>100 mm			0.07	0.05	0.11	0.10	0.18	<0.01

Family / Species	Developmental stage / size category	Larval habitat	*n* reps	May 2008	July 2008	September 2008	November 2008	May 2009	%
Scorpaenidae									
Notesthes robusta (Günther, 1860)	<100 mm	EST/MAR		-	-	-	-	-	<0.01
	>100 mm			-	-	0.27	-	-	<0.01
Chandidae									
Ambassis agassizii (Steindachner, 1867)	Post-flexion	FW		-	-	-	2.25	-	0.01
	Juv/adult			14.23	5.84	33.71	4.70	46.53	0.53
Denariusa bandata (Whitley, 1948)	Post-flexion	FW		0.38	0.47	0.73	2.93	0.66	0.02
	Juv/adult			48.27	29.92	41.90	49.16	89.55	1.30
Centropomidae									
Lates calcarifer (Bloch, 1790)	<100 mm	EST/MAR		-	-	-	-	-	-
	>100 mm			3.32	0.54	0.05	-	-	0.02
Terapontidae									
*Hephaestus fuliginosus** (Macleay, 1883)	<100 mm	FW		-	-	-	-	-	-
	>100 mm			-	-	0.03	-	-	<0.01
Apogonidae									
Glossomia aprion (Richardson, 1842)	Post-flexion	FW		0.23	-	7.75	6.65	0.20	0.07
	Juv/adult			31.79	12.29	28.56	30.12	38.65	0.69
Gobiidae									
Redigobius bikolanus (Herre, 1935)	Post-flexion	EST/MAR		0.42	0.50	0.45	0.07	0.03	-
	Juv/adult			1.21	0.50	3.11	0.56	0.89	0.03
Eleotridae									
Giurus margaritacea (Valenciennes, 1837)	<100 mm	cf. FW		0.07	-	-	-	0.37	<0.01
	>100 mm			0.17	0.43	0.50	-	0.33	<0.01
Hypseleotris compressa (Krefft, 1864)	Post-flexion	EST/MAR		8.49	4.37	12.29	3.23	234.44	1.34
	Juv/adult			884.35	1461.79	4513.90	1213.00	7478.98	78.93

Family / Species	Developmental stage / size category	Larval habitat	*n* reps	May 2008	July 2008	September 2008	November 2008	May 2009	%
Hypseleotris sp. 1	Post-flexion	FW		10.81	22.00	136.97	39.20	13.88	1.13
	Juv/adult			362.44	199.45	1091.86	188.03	392.83	11.30
Mogurnda adspersa (Castelnau, 1878)	Post-flexion	FW		0.18	0.03	-	-	0.22	<0.01
	Juv/adult			3.06	4.64	8.69	3.19	33.32	0.19
Poeciliidae									
Xiphophorus maculatus (Günther, 1866)	Metalarvae	FW		0.03	-	2.14	-	0.13	0.01
	Juv/adult			3.81	7.94	53.31	1.22	13.69	0.39
Total (relative) abundance				19.2	61.5	76.6	48.6	101.8	

3.4.6 Temporal variation in fish assemblage characteristics over the two time scales

Significant temporal differences in attributes characterising fish assemblage structure was limited to total abundance when comparisons were confined to the three comprehensive surveys (Table 3.9). However, the SNK test was unable to distinguish where these temporal differences lay. Nonetheless, it would appear from the data presented in Figure 3.18a that there was an increase in total abundance over time. Mean evenness was highly varied (and thus not significant) although there is some suggestion that evenness diminish over time (Figure 3.18b). No statistically significant differences in any of four metrics describing change in fish assemblage characteristics were detected over the extended period of sampling (Table 3.10), although the trend of an increase in total abundance over time remained apparent (Figure 3.18c).

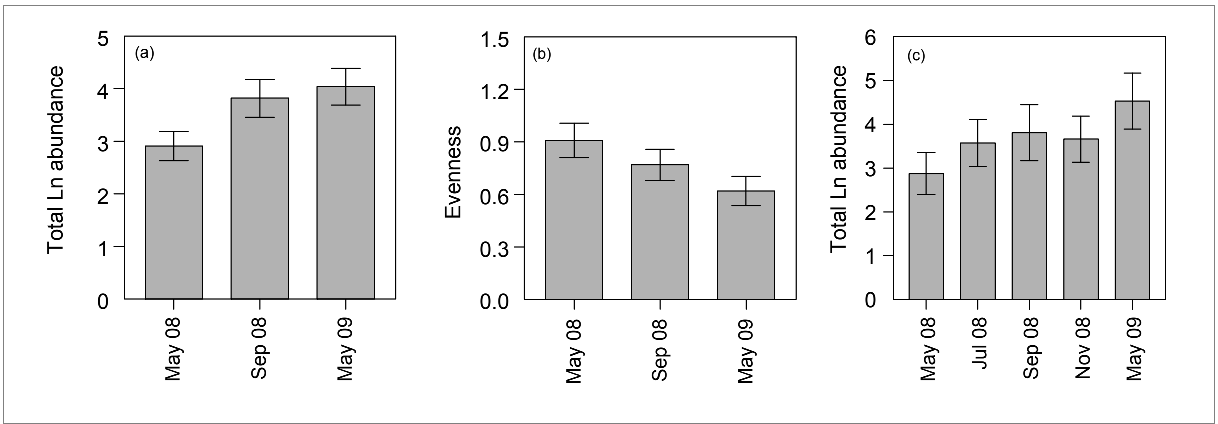

Figure 3.18: Average (± s.e) values for fish assemblage characteristics for the three comprehensive surveys (a-b) and the five (c) extended surveys.

3.4.7 Fish assemblage patterns among ten lagoons over the three comprehensive surveys

Temporal variation in the species composition of lagoon fish assemblages was not accompanied by a statistically significant change in the fish assemblage structure over time (when averaged across lagoons) (Global R = 0.067). However, an ordination plot revealed the extent of differences in temporal variability in fish assemblage structure among individual lagoons (Figure 3.19a). For example, the fish assemblage in Barrett's, Bunta and Selby's Lagoons underwent considerable structural change from May 2008 to May 2009 as is illustrated by the marked repositioning of these lagoon groups in ordination space. In contrast, points representing Digman's 3 and Zamora's Lagoon fish assemblage remained tightly clustered in ordination space over the same time period. Despite the lack of consistency in the magnitude of fish assemblage change among lagoons, the ordination plot does to some extent illustrate similarity in structural changes to the fish assemblage as is indicated by the direction of movement in lagoons groups across ordination space. For example, fish assemblages in most lagoons followed a similar trajectory with respect to the 'y' axis over the 12-month sampling period (i.e. tracked down the plot then back up). In the six lagoons that followed a similar trajectory, the fish assemblage in Raccanello's and Carroll's Lagoons and to a lesser extent Bunta Lagoon appeared remarkably similar to that described 12 months earlier. This compared with the greater divergence over time in the fish assemblage structure in Digman's, Selby's and Boongaray Lagoons.

Associations between the abundance of species and fish assemblage structure illustrate that a broad suite of species and life-history stages were highly correlated with points in ordination space and are therefore driving the patterns of change in assemblage structure over time. Furthermore, there were examples of the early developmental stages of species being strongly aligned with one particular survey date (e.g. *Hypseleotris* sp. 1 with September 2008 surveys and *Neosilurus ater* with May 2008 and May 2009 surveys). These relationships provide evidence for interspecific differences in the timing in appearance of new recruits within the fish assemblage (Figure 3.19b). Temporal variation in fish assemblage structure was best explained by connectivity variables; this will be further investigated in a MTSRF transitional project and reported subsequently.

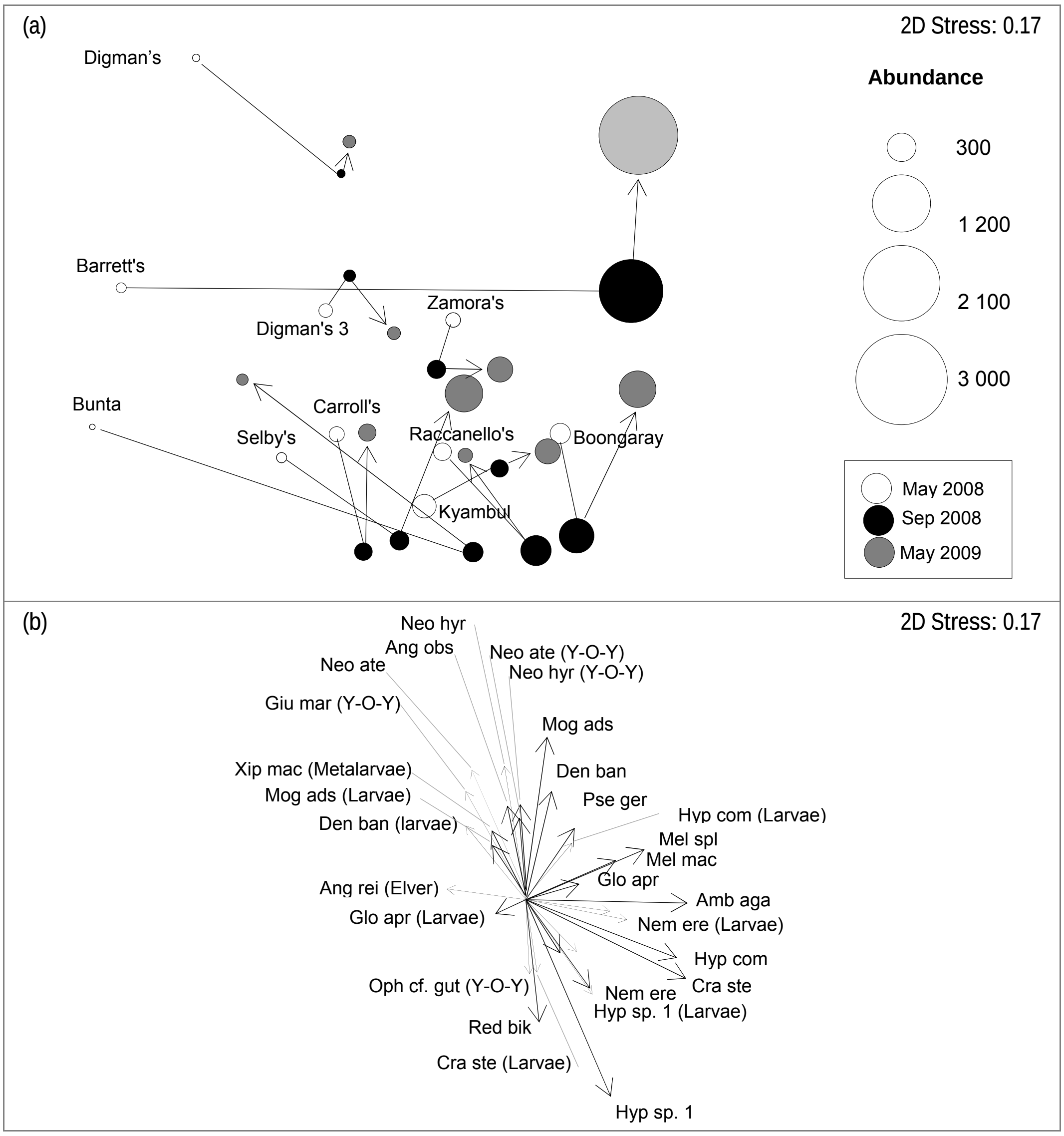

Figure 3.19: Temporal variation in fish assemblage structure in ten lagoons of the Tully-Murray river system based on nMDS ordination of lagoon by species abundance (Lne transformed). Each point in plot (a) represents the total fish abundance (CPUE) recorded from one of the three comprehensive surveys. The diameter of the bubble represents absolute abundance on a monotonic scale. Dominant correlations between fish species/ stages abundance and lagoon position in the ordination space are displayed in plot (b). Early developmental stages are represented as either larvae or young-of-the-year (Y-O-Y) when the species does not occur as larvae in the study area. Species abbreviations are based on the first three letters of the generic and species epithets.

3.4.8 Fish assemblage patterns among five lagoons over the extended period of sampling

The ordination plot constructed from the extended fish data set reinforced the differences in temporal variation in assemblage structure among lagoons (Figure 3.20a). Barrett's lagoon was characterised by comparatively large changes in total (relative) abundance, particularly between May and July 2008 when numbers increased ~15-fold, and between November 2008 and May 2009 when catches increased ~4-fold. In contrast, the fluctuation in abundance in Zamora's and Digman's Lagoons were far more modest with the largest change in abundance between successive surveys ~2 orders of magnitude. The largest changes in total abundance between surveys in Selby's and Kyambul Lagoons over time lay between these extremes.

The ordination plot revealed that fish assemblage structure changed in similar ways in four of the five lagoons. Surveys in Barrett's, Selby's, Digman's lagoons and to a lesser extent Zamora's Lagoon tracked from right to left after May 2008 before trending to a similar position on the 'x' axis twelve months later (Figure 3.20a). In contrast, the surveys in Kyambul Lagoon followed a different trajectory across the 'x' axis suggesting that temporal changes in fish assemblage structure in this lagoon were driven by a different suite of environmental factors from those driving the patterns of temporal change more generally across off-channel lagoons in the Tully-Murray catchment. Relationships between individual fish species and survey dates plotted in ordination space highlight the separation of species into three broad groups depending on the timing of their appearance in the lagoons as early life-history stages. The first represented a group of species concentrated in samples from May 2008 and/or May 2009 or the immediate 'post-wet' season (April-May) including *N. ater, N. hyrtlli, G. margaritacea and M. adspersa*; a second group of species was aligned with samples collected in July, September and/or November 2008 or the dry season (July-November) including *Hyseleotris* sp. 1; *Ambassis agassizii* and *D. bandata*; and the third group comprised a suite of species that were evenly associated with samples from both seasons. This group included the remaining nine fish species that were recorded as larvae in lagoons of the study area (Figure 3.20b). These observations reinforce the patterns of partitioning over time of reproductive effort that occurs among species within the lagoon fish assemblage.

3.4.9 Population size structure and recruitment patterns

Examination of length-frequency plots in association with graphs illustrating temporal shifts in abundance (CPUE) indicate episodes of recruitment in four fish species that represent the diversity of recruitment strategies within the lagoon fish assemblage (Figure 3.21-3.23a-c). Peak abundance in *N. ater* occurred during May 2008 and/or May 2009 in four of the five lagoons (Figure 3.20). The occurrence of the 50-100 mm SL size class at these times suggests successful spawning over the wet season leading to juvenile recruitment of *N. ater* on the declining limb of the wet season spike on the hydrograph. While new catfish recruits first appeared in Zamora's Lagoon in May 2008, it would appear that their recruitment was prolonged over time, as indicated by the increase in *N. ater* numbers from May to July 2008 and the gradual shift in size structure of the population from one dominated by 50-75 mm SL individuals in May to 75-100 mm SL individuals in July. Sufficient numbers of young-of-the-year catfish were recorded in Digman's and Barrett's lagoons to illustrate the persistence of the juvenile cohort in these water bodies over time.

The patterns in abundance and size structure detected in *Hypseleotris* sp. 1 across all five lagoons suggest that this species spawns and recruits primarily under the low-flow conditions characteristic of the dry season (Figure 3.22). The occurrence of new recruits (in the 10-14 mm SL size class) was confined to samples from July, September and/or November 2008 and coincided with the highest catches of this species in all lagoons apart from Kyambul. The large abundance of *Hypseleotris* sp. 1 recorded in Kyambul Lagoon during the immediate post-wet survey (May) of 2008 consisted entirely of juveniles and adults (all individuals ≥18 mm SL), suggesting that the last recruitment event had occurred prior to the commencement of elevated wet season discharges in January 2008. *Melanotaenia s. splendida* displayed more than one episode of recruitment in four of the five lagoons with these multiple recruitment events not always restricted to either the wet or dry season (Figure 3.23). For example, new rainbowfish recruits appeared in Selby's and Kyambul lagoons during surveys both immediately following the wet season (May) and the dry season (September and November). In contrast, the appearance in Digman's and Zamora's lagoons of individuals of the same size class was confined to the period immediately following the phase of elevated discharge, whereas young rainbowfish were recorded in Barrett's Lagoon only during the dry season in September. The appearance of new recruits across both seasons suggests recruitment is unrelated to the influence of the seasonal pattern of discharge.

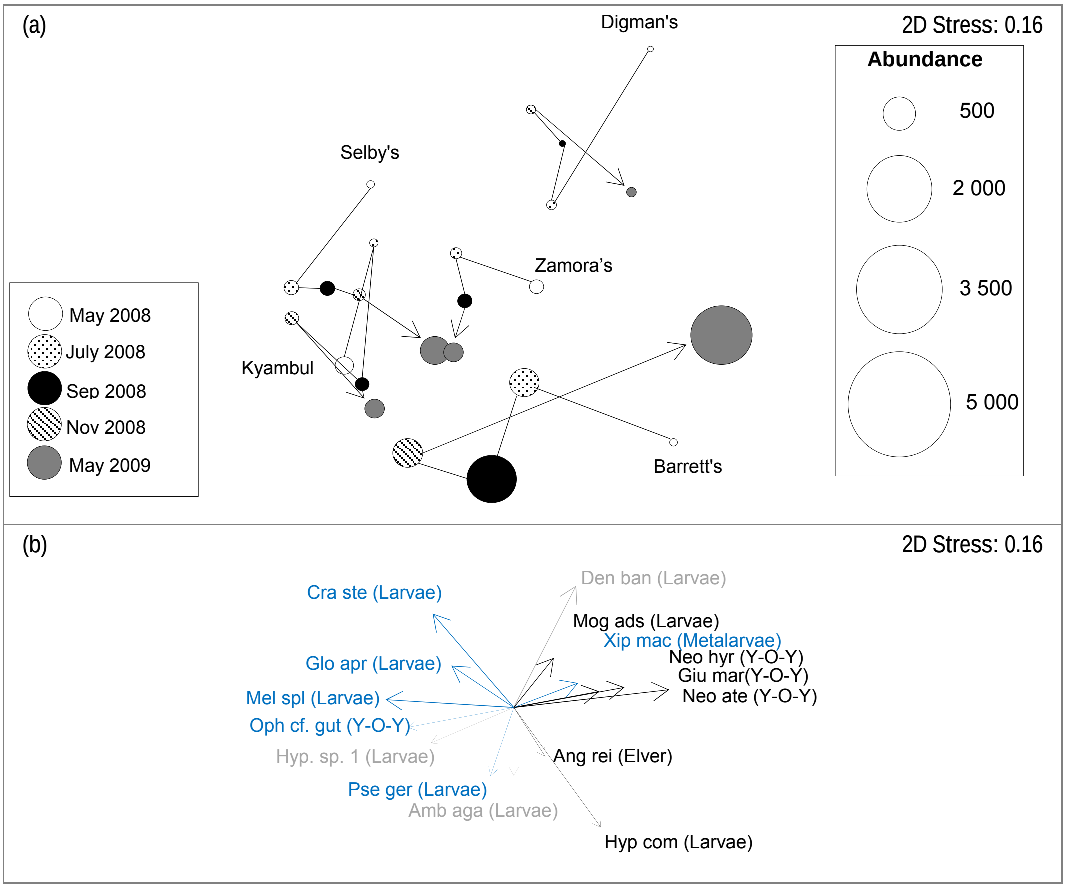

Figure 3.20: Temporal variation in fish assemblage structure in five lagoons of the Tully-Murray river system based on nMDS ordination of lagoon by species abundance (Ln transformed). Each point in plot (a) represents the total fish abundance (CPUE) recorded from over the extended survey period in each of the five lagoons. The diameter of the bubble represents the magnitude of abundance on a monotonic scale. Dominant correlations between the abundance of individual species (only the early developmental stage is represented) and lagoon position in the ordination space are displayed. In plot (b) the early developmental stage is represented as either larvae or as young-of-the-year (Y-O-Y) when the species does not occur as larvae in the study area. Species are coloured-coded according to their seasonal occurrence as early life-history stages in the lagoon fish samples (black = immediate post-wet-season samples, grey = dry-season samples and blue = occurring in both the immediate post-wet and dry-season samples). Species abbreviations are based on the first three letters of the generic and species epithets.

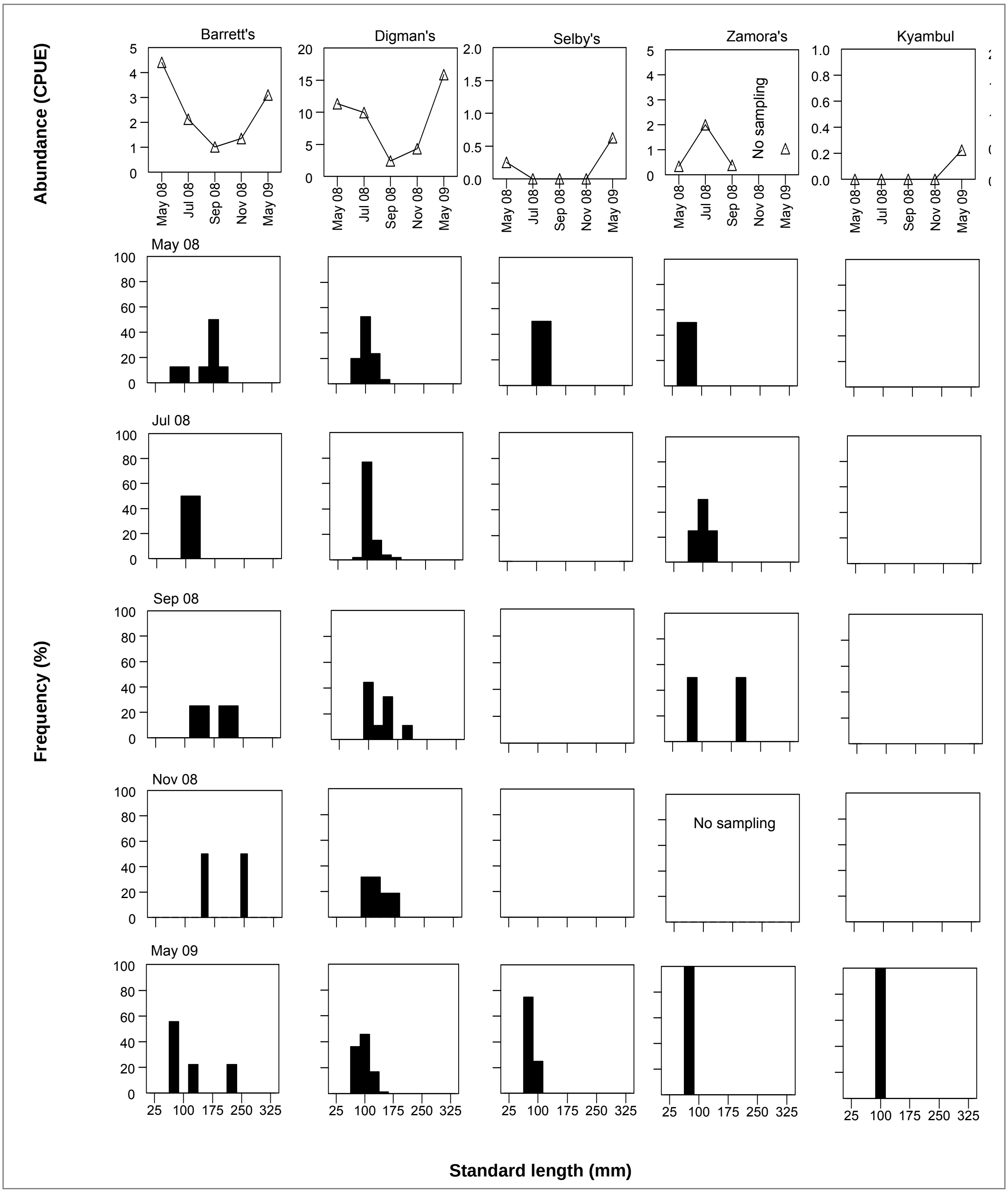

Figure 3.21: Length-frequency plots for *Neosilurus ater* in five lagoons of the Tully-Murray river system on five sampling occasions. Total abundance (CPUE) recorded on each sampling occasion is the sum of the combined fyke net and electrofishing catches.

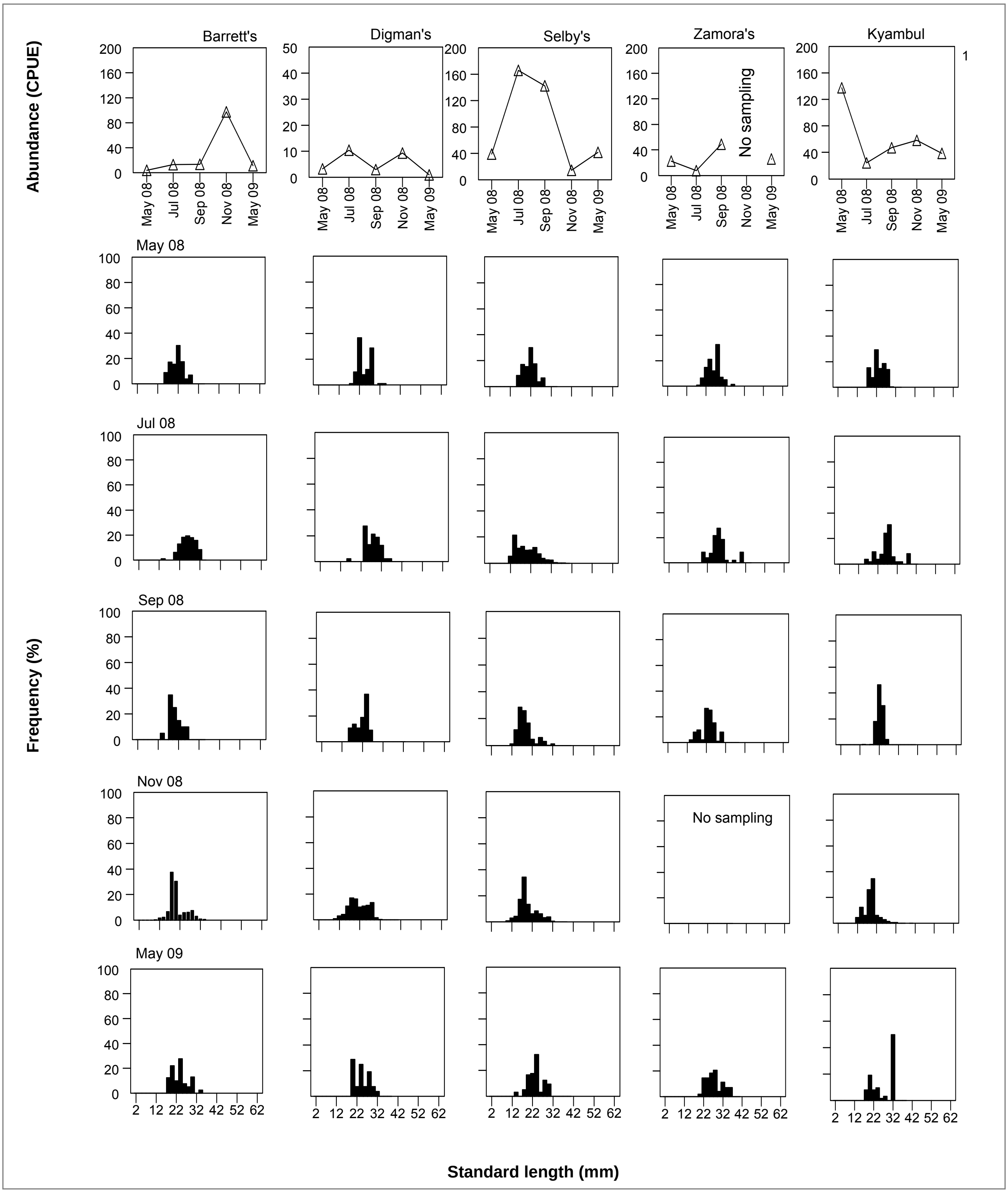

Figure 3.22: Length-frequency plots for *Hypseleotris* sp. 1 in five lagoons of the Tully-Murray river system on five sampling occasions. Total abundance (CPUE) recorded on each sampling occasion is the sum of the combined fyke net and electrofishing catches.

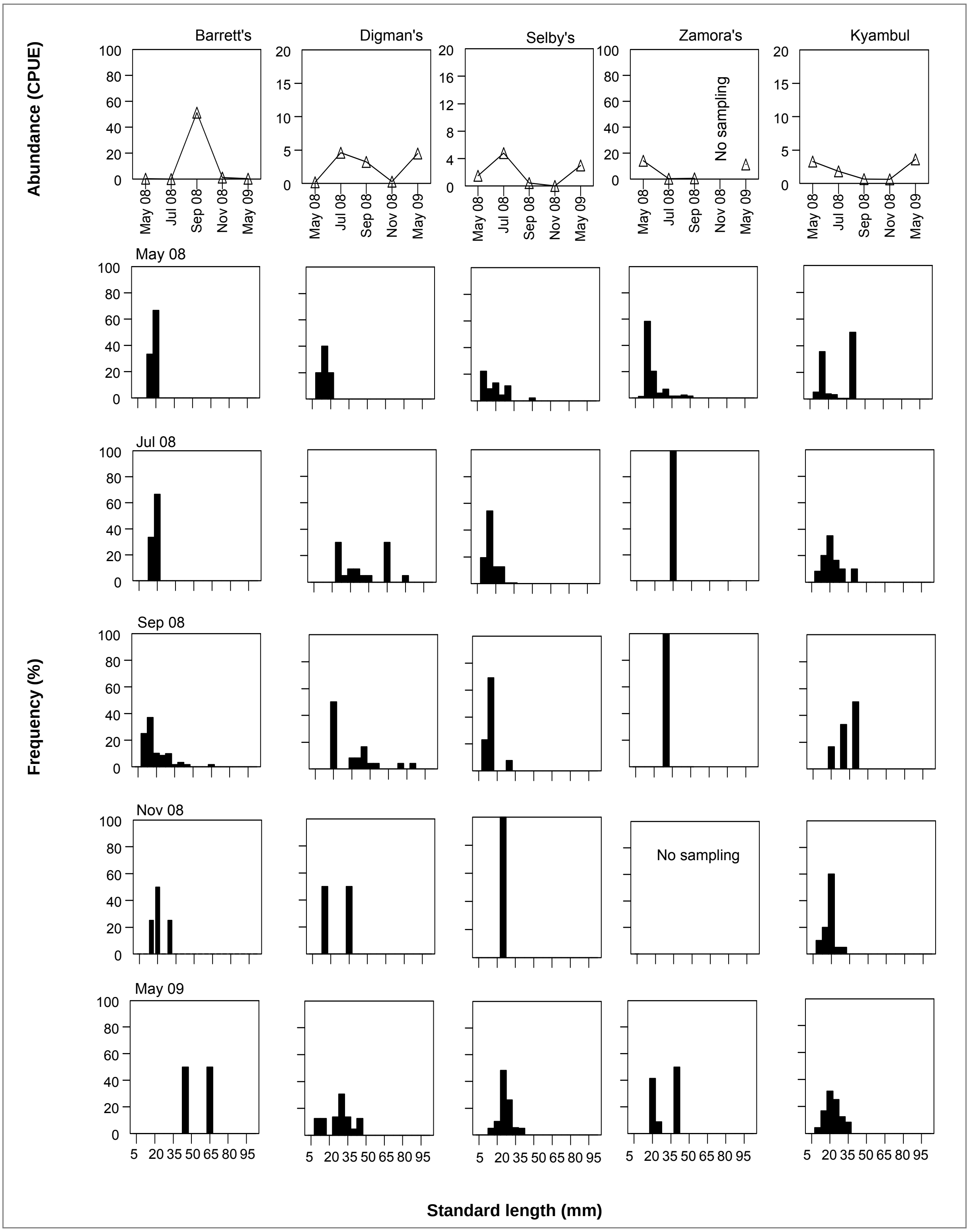

Figure 3.23: Length-frequency plots for *Melanotaenia splendida splendida* in five lagoons of the Tully-Murray river system on five sampling occasions. Total abundance (CPUE) recorded on each sampling occasion is the sum of the combined fyke net and electrofishing catches.

3.4.10 Influence of flow seasonality and hydrological connectivity on fish recruitment

The population structure of *H. compressa* occupying the five lagoons that were sampled comprehensively was shaped by the seasonal pattern of river discharge and the extent of hydrological connectivity between the lagoon, the stream network and the Tully and Murray estuaries. The empire gudgeon migrates upstream as larvae from salt water and enters freshwater habitat of the river systems on the declining limb of the wet season spike of the hydrograph. This pattern was observed in the Tully-Murray floodplain lagoons during both 2008 and 2009, although the precise timing in the appearance of the new cohort in the lagoons varied between years (Figure 3.24). In 2008, new recruits were not detected across all five lagoons prior to July, and the bulk of the 0+ cohort was not recorded in most lagoons (Digman's Lagoon being the exception) until after this point in time. In contrast, the 2009 cohort was present in all five lagoons by May (i.e. two months earlier than in the previous year) and at this time occurred in numbers similar to, or greater, than those recorded at the height of recruitment in September of the previous year. This suggests that spawning had occurred several months earlier in 2009 than in 2008, leading to the earlier appearance of new recruits in 2009. Alternatively, the upstream migration of the new recruits may have been inhibited in 2008, thus delaying their arriving in the lagoons by several months.

Variation in the extent of hydrologic connectivity between the lagoons and the stream network during channelised flow appeared to contribute to among-lagoon differences in the size structure of *H. compressa*. The stream network comprising natural channels and cane drains provide the only path for recruiting empire gudgeon to access floodplain lagoons during the non-flood period, with the opportunity for lagoon access diminishing over the dry season as each lagoon progressively disconnects from the stream network. Over the transitional phase between wet and dry seasons (i.e. May-July), the lagoon population of *H. compressa* became heavily skewed towards a higher proportion of small-bodied individuals resulting from the arrival of young gudgeons from downstream natal habitat (Figure 3.24). With the passage of the dry season, the lagoon population of the empire gudgeon underwent a shift in size structure (to a higher proportion of larger individuals) in three of the five lagoons as new recruits failed to appear in the lagoons. The shift in size structure first occurred in Digman's Lagoon (between surveys in July and September 2008) and was followed in Kyambul and Barrett's lagoons three months later. New gudgeon recruits never ceased entering Selby's or Zamora's lagoon over 2008, although fish were not sampled in Zamora's during November 2008 to confirm their sustained recruitment into this lagoon during the height of the dry season. Hydrological modelling over 2007 (i.e. the year preceding the biological investigation) illustrates the existence of a gradient in connectivity across floodplain lagoons that is driven by spatial variation in the pattern of rainfall, lagoon elevation and lagoon position with respect to the Tully and Murray Rivers (F. Karim, unpub. data). The most probable explanation for the difference in size structure of *H. compressa* among lagoons over 2008 and 2009 is the variation in the timing and duration of connectivity that exist between individual lagoons and the stream network. This will be explored further in the MTSRF transitional project.

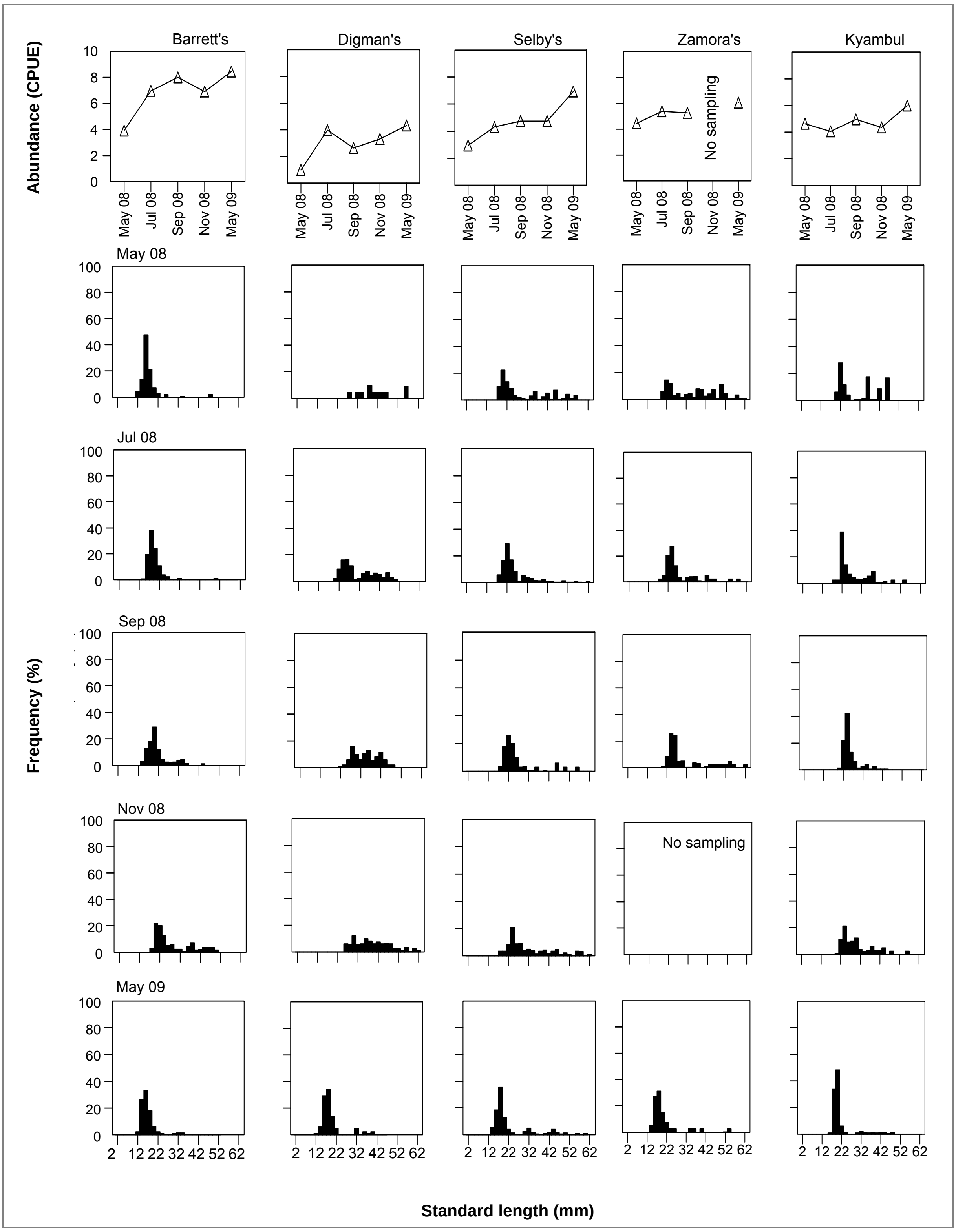

Figure 3.24: Length-frequency plots for *Hypseleotris compressa* in five lagoons of the Tully-Murray river system on five sampling occasions. Total abundance (CPUE) recorded on each sampling occasion is the sum of the combined fyke net and electrofishing catches.

4. Discussion

4.1 Tully-Murray wetlands

Field studies for this project were largely confined to palustrine (shallow vegetated) lagoons of the Tully-Murray system within the Wet Tropics bioregion of Queensland (Figure 2.1) (classification according to the Queensland Government; see DERM 2009). These two catchments share a common floodplain in their lower reaches and during flood events runoff from both catchments merges together on the floodplain. We therefore considered the floodplains of these two catchments as a single unit, referred to as the Tully-Murray floodplain. The Tully and Murray Rivers are the two main waterways that receive catchment runoff through numerous creeks and they both discharge into the GBR lagoon.

The Tully-Murray floodplain consists of an extensive network of permanent and semi-permanent wetlands, many of which are located off-stream (Figure 2.1). The main wetlands are complemented by a network of smaller natural and artificial wetlands, some of which have been developed primarily to reduce the impacts of adjacent farming by acting as sediment and nutrient sinks, flood detention basins or both (Veitch and Sawynok 2005). A few of these wetlands connect directly to the rivers, but the majority connect to the rivers indirectly through creeks, drains, or a combination of both. Semi-permanent wetlands are located relatively distant from the main streams and have less stream connection with the rivers. These wetlands become connected to the rivers during floods. The Tully-Murray floodplain receives flood waters three to four times a year during the wet season (Wallace *et al.* 2009b) and these floods connect a number of wetlands depending on the flood magnitude.

Floodplain wetlands remain a prominent landscape feature within the Tully-Murray catchment (DERM 2009) despite the changes in land use that have occurred in riverine lowlands and floodplains since the introduction/inception of agriculture (Furnas 2003; Hogan and Graham 1994; Vallance and Hogan 2004). From an environmental perspective they are considered very significant as they provide habitat for aquatic and riparian biota, in addition to potentially improving water quality delivered to rivers and to the GBR lagoon. These wetlands are also of high value from a fisheries perspective (Veitch and Sawynok 2005). In recent years, due to changed hydrological conditions and land clearing for agriculture, many of these wetlands have been removed (Johnson *et al.* 1997) and the remainder appear to have become degraded as wildlife habitats and resource areas for freshwater and estuarine species. The significance of floodplain wetlands and discrete lagoons as habitat for freshwater biota and for recruitment of species that move between estuarine and freshwater areas in Wet Tropics systems is not well understood given the lack of investigation of these wetland types prior to the present study (Pusey *et al.* 2004; Gehrke and Sheaves 2006).

Project 3.7.3 field studies measured spatial and temporal variability of biophysical factors and potential indicators of ecological condition in floodplain wetlands of the Tully-Murray catchment along two sets of gradients:

1. natural environmental gradients, including position in the floodplain landscape, wetland size and morphology, hydrology and connectivity
2. gradients of disturbance, including gradients of land use, hydrology and connectivity, water chemistry and habitat quality, riparian disturbance, and alien species.

Natural gradients affecting floodplain wetlands in the Tully-Murray system may include position in the landscape (i.e. proximity to coastal waterways, lateral distance of wetlands from river channels, elevation), wetland size and morphology, hydrology (frequency and extent of inundation by flood waters) and the natural extent of connectivity of each wetland with associated creeks, cane drains and main river channels. Each or all of these physical factors and processes can interact to shape biotic assemblages in floodplain river systems (Junk *et al.* 1989; Winemiller *et al.* 2000; Arthington *et al.* 2005; Tockner *et al.* 2008). Disturbance gradients may be superimposed on these natural gradients, for example disturbances many involve gradients of land use, altered hydrology and connectivity, water chemistry and habitat quality, riparian disturbance, and alien species.

4.2 Natural environmental gradients

Natural gradients describing the floodplain lagoons in the Tully-Murray system and identified by the principal components analysis of 48 variables describing position in the landscape, local habitat structure and human land use, to identify dominant gradients of environmental variation (i.e. variables loaded on the same component) in our environmental data set, included (1) water quality, (2) lagoon size and morphology, (3) position in the landscape and associated water quality, and (4) riparian condition and associated variables. Other influences not picked up by the PCA but likely to have a bearing on ecosystem health are hydrology (frequency and extent of inundation by flood waters) and the degree of connectivity of each wetland with associated creeks, cane drains and main river channels. Each or all of these physical factors and processes can interact to shape biotic assemblages in floodplain river systems (Junk *et al.* 1989; Winemiller *et al.* 2000; Arthington *et al.* 2005; Tockner *et al.* 2008).

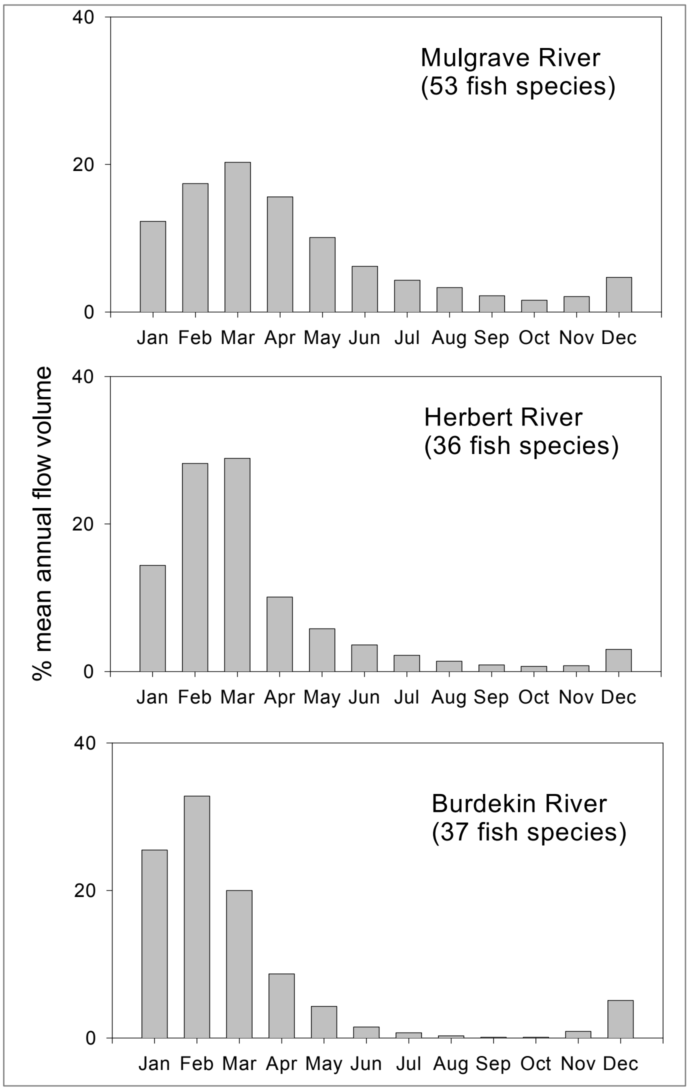

Figure 4.1: Monthly flow regime (expressed as percentage of annual flow) in a central Wet Tropics river (Mulgrave), a Dry Tropics river (Burdekin) and an intermediate river (Herbert). The number of fish species recorded from each system (Pusey and Kennard 1996; Pusey *et al.* 1998) is indicated.

These results provide a description of the sites studied and do not imply health issues. Thus the gradient of size and morphology reflects the range of lagoons included in the study. Nevertheless such variables may influence the biota and need to be considered when interpreting the ecological analyses (as in the *Catchment to Reef* program, where the natural gradient in substrate size needed to be removed before assessing stream ecosystem health). More interesting from an ecosystem health perspective are gradients in water quality and riparian condition – very likely resulting from human activity and very likely to have a bearing on health The results demonstrate that there are significant gradients of a range of variables, which inter-relate as might be predicted in a landscape that is strongly influenced by human activity.

Nevertheless, the gradients do not suggest that extremely poor conditions occurred during our sampling program. In contrast with the Herbert and Burdekin systems (Pearson *et al.* 2003; Perna and Burrows 2005), the Tully-Murray system appears to be relatively well flushed because of the central Wet Tropics rainfall regime. The Burdekin River and its floodplain are mostly well within the Dry Tropics, and experience highly seasonal flows, such that lagoons reflect local conditions for much of the year. Even lagoons that receive regular irrigation flows are not well flushed, and serious conditions (e.g. weed infestation and hypoxia) develop in them in most years, unless mitigation measures are taken. The Herbert system sits partly in the Dry Tropics and partly in the Wet Tropics, and has somewhat intermediate conditions. Flushing is greater in the Herbert than the Burdekin, but nothing like the regime in the Tully-Murray. There is thus a gradient of flow conditions across the GBR catchment that has a major bearing on the nature of the conditions and interactions in particular wetlands, and potentially on their biota. This is illustrated for the Burdekin, Herbert and Mulgrave rivers in Figure 4.1. Like the Tully-Murray, the Mulgrave River is situated in the central Wet Tropics and demonstrate much less variance in monthly flows than the other two rives. It also demonstrates a much higher fish diversity, despite being much the smaller of the three rivers.

4.3 Flow regimes and connectivity

Under the strategy for the conservation and management of Queensland wetlands, the Environmental Protection Agency (EPA) has documented detailed wetland information including type, size and location for selected regions of the state (EPA 2005). However, the EPA wetland maps only provide information for wetlands that are larger than 1 ha. To identify wetlands in the Tully-Murray floodplain and their connectivity with the stream network, a LiDAR survey was conducted in October 2007 for the area between the Tully and Murray Rivers and a 3 m grid DEM was produced. Arc-GIS was used to map wetlands from this high resolution DEM. Ten wetlands (ranging in size from 0.5 to 6.0 ha) were then selected for connectivity assessment based on their perceived importance for aquatic biota (e.g. type, size and location).

In the Tully-Murray floodplain, the majority of the wetlands selected for this study are regularly inundated as they connect to the rivers during floods with a one-year Average Recurrence Interval (ARI). Larger floods connect these wetlands for longer periods and also connect a few more wetlands that are located on higher ground further from the rivers (Karim *et al.* 2010b). However, connection time and the duration of connectivity of each wetland differ according to its location and/or proximity to the Tully and Murray Rivers. Flood size also affects the duration of wetland connectivity, which can range from 0 days (no connection) to 12 days as flood ARI increases from one to fifty years. Some wetlands appear to be connected only during large floods (e.g. Lagoon Creek wetland). Wetlands on the Tully-Murray floodplain are connected to the Murray River for longer periods than they are to the Tully River and this has been attributed to both their proximity to the river and the lower bank heights along the Murray River relative to the Tully River (Karim *et al.* 2010b) These variations in wetland connectivity will be important in helping explain the potential movement and recruitment patterns of aquatic biota during (and after) flood events, wetland habitat characteristics and water quality and the biodiversity of individual wetlands over time, and even to the potential for wetland processes to influence the quality of water flowing to the GBR lagoon.

4.4 Patterns of plankton and invertebrate assemblage composition and structure

Zooplankton were abundant but not diverse. They showed some response to habitat and water quality measures, but these responses were not strong and were sometimes equivocal. The plankton indicated that water quality was not bad in any lagoon, so discrimination of a 'disturbance fauna' was not feasible. Therefore, the zooplankton is not yet able to be developed as an appropriate health indicator.

Diverse assemblages of invertebrates were found at each site representing the major taxa expected from Queensland floodplain lagoons. Patterns in the distribution of the invertebrate assemblages showed clear separation of sites by habitat (leaf litter and submerged macrophytes). Separate analyses of samples from the two habitats indicated that while there were significant relationships between the invertebrate fauna and environmental metrics, there were no major gradients across lagoons. Some of the least significant differences between assemblages were between lagoons of different character, so it would appear that the invertebrates did not respond strongly to some of the apparent lagoon differences such as size and riparian condition. Additionally, temporal changes were not clear-cut, and months replicated over two years had rather different invertebrate assemblages. Thus it is apparent that a series of samples across sites and times are necessary to adequately assess the wetland communities. Nevertheless, there was a distinct ecological response to land-use effects on the habitats, even though these relationships cannot be used to ascribe cause and effect.

These general conclusions are applicable to both the leaf litter and macrophyte samples, and to some general composite indicators of condition. Clearly invertebrates in the Tully-Murray wetlands relate primarily to habitat conditions, with water quality having some smaller influence, probably because water quality at the time of this study was not sufficiently bad to show up strong responses. The lack of strong gradients across the lagoons is presumed to be because that there were no lagoons representing extremes of conditions of major influence on the invertebrates – there were no pristine reference sites, and there were no sites in bad condition. This is discussed further in Section 4.7.

4.5 Patterns of fish assemblage composition and structure

The fish fauna of lagoons included 22 species (21 native 1 alien) representing 15 families. The lagoon fish assemblage included a small suite of species not typically found in main channel or major tributaries of Wet Tropics systems (*Denariusa bandata*, *Pseudomugil gertrudae, Melanotaenia maccullochi*) (Pusey and Kennard 1996; Pusey *et al.* 2004; 2007a; 2007b; Rayner *et al.* 2008). In addition, there are approximately eight species that occur exclusively in floodplain water bodies during their early life history highlighting the fundamental role of floodplain wetlands in the provision of nursery habitat for these species (P. Godfrey, unpub. data). Total species richness ranged from 15-17 species across the ten lagoons with Barrett's, Bunta and Boongaray lagoons containing all 17 species. A single, widely distributed alien species *Xiphophorus maculatus* (the platy) was collected in nine of the ten lagoons in the study area.

Most species complete their entire life history in fresh water but six native species require access to estuarine or marine areas for spawning and/or for larval development. Barrett's Lagoon, located closest to the estuary, contained a complex mixture of species dependent on estuarine or marine habitats for larval production, such as *Hypseleotris compressa*, *Anguilla reinhardtii, Lates calcarifer* but also some exclusively freshwater species that were associated with abundant vegetation cover including *Melanotaenia splendida splendida*, *P. gertrudae, M. maccullochi* and *Mogurnda adspersa.* The two lagoons located most distant from the two main rivers (Digman's and Digman's 3) were dominated by a suite of exclusively freshwater species (*N. ater*, *N. hyrtlii* and *Giurus. margaritacea*) associated with high levels of submerged vegetation cover.

Kyambul was the most distant of all lagoons from the estuary yet it supported a number of estuarine dependent species including a comparatively high abundance of the goby *Redigobius bikolanus* which occur in estuarine areas during its early life history (P. Godfrey, unpub. data). This access may be afforded by the permanent stream connection that exists between Kyambul Lagoon and the Murray River. This particular lagoon provides riverine habitat (as opposed to it being a 'true' lacustrine or palustrine floodplain wetland) and also contained a group of exclusively freshwater-dependent species (*Hypseleotris* sp. 1; *Ophisternon* cf. *gutturale*).

Carroll's Lagoon, located comparatively close to the river mouth, also contained a high proportion of species dependent on estuarine/marine habitats yet it also had a large population of the introduced *X. maculatus*, as did Bunta Lagoon. Both these lagoons possessed a high proportion of leaf litter habitat which contributed to the strong correlation between the large abundance of *X. maculatus* and leaf litter within each lagoon.

Microhabitat structure was pivotal in driving differences in the composition of fish assemblages across lagoons. Fish assemblages associated with the alien emergents (particularly *Hymenachne amplexicaulis* and a range of other perennials) were generally species-poor and structurally very different from those that occupied the comparable native emergent sedges and herbs. Fish assemblages associated with the range of emergent native sedges and herbs (*Eleocharis* sp., *Lepironia articulata*, *Persicaria* sp. *Schoenoplectus mucronatus*) were dominated by a diverse mixture of native species including *Ambassis agassizii*, *P. gertrudae*, *M. s. splendida, G. margaritacea* and *Glossomia aprion*. The assemblage associated with *Pandanus* was less speciose and dominated by *Hypseleotris* sp. 1; and to a lesser extent the hardyhead, *Craterocephalus. stercusmuscarum*. Habitats formed by submerged vegetation supported a mixture of species typical of both native emergent vegetation groups (sedges and herbs, *Pandanus*) as well as the catfishes, *N. ater* and *N. hyrtlli*, that tended to be closely associated with submerged vegetation. These patterns point to strong effects of vegetation composition and diversity on the characteristics of aquatic habitats preferred by different fish species across lagoons. Fish species richness was depressed in beds of alien emergent plants, particularly the introduced ponded pasture grass, *H. amplexicaulis*. The pattern of species composition and assemblage structure in individual lagoons was less predictable than that of stream fish assemblages in terms of the influence proximity to the river mouth has in shaping composition and structure (Pusey *et al.* 2004; 2007a; 2007b), and probably reflects more the contribution of lagoon position on the floodplain courses and the extent (i.e. permanency vs. intermittency) of hydrological connectivity between individual lagoons and the stream network.

Aspects of water chemistry varied among lagoons, including some features that suggest anthropogenic influences from land uses in the surrounding catchment. Kyambul, Raccanello's and Boongaray lagoons contained high concentrations of herbicides and/or dissolved inorganic nitrogen but it is doubtful that concentrations were high enough to be harmful to aquatic biota. The detection of significant correlations between these anthropogenic water quality influences and fish assemblage structure more likely reflected co-variation with landscape position and habitat characteristics that influence spawning and rearing requirements.

Fish assemblage structure changed over time in similar ways in four of the five lagoons – Barrett's, Selby's, Digman's Lagoons and to a lesser extent Zamora's Lagoon. In contrast, change in fish assemblage structure in the most upstream lagoon (Kyambul) did not follow a similar trajectory, suggesting that temporal changes in this lagoon were driven by a different suite of environmental factors from those driving the patterns of temporal change more generally across off-channel lagoons in the Tully-Murray catchment. It is notable that Kyambul lagoon is defined as riverine habitat whereas each of the other four lagoons are palustrine water bodies under the Queensland Government's scheme of wetland classification (DERM 2009). As a riverine system, Kyambul lagoon is distinguished from the off-channel wetlands by the greater amount of water body that is contained within the buffered channel. This probably results in different hydrological regimes (e.g. flushing/turnover rate) that contribute to the differences in biological patterns.

The sixteen species of fish that were recorded as early life-history stages were categorised into one of three groups depending on their timing in appearance in the lagoons as early life history stages. The first group represented fish species collected primarily in samples from May 2008/2009 immediately following the wet season (January-April) including *N. ater, N. hyrtlli, G. margaritacea and M. adspersa*; a second group of species was aligned with samples collected primarily in July, September-November 2008 or the dry season, including *Hyseleotris* sp. 1, *Ambassis agassizii* and *D. bandata*; and the third group of species was associated with samples from both seasons. This group included the remaining nine fish species that were recorded as larvae in lagoons of the study area. This result indicates that the fish assemblage of the Tully-Murray encompasses species that fit each of the two current (and competing) theories that speculate on the drivers of fish recruitment in river systems – the *flood recruitment model* (Harris and Gehrke 1994) and the *low flow recruitment hypothesis* (Humphries *et al.* 1999). The fish assemblage also includes members with an extended recruitment interval, meaning aspects from both recruitment models may be applicable to this group of fishes. Therefore, a single recruitment model appears to be unsuitable for characterising the recruitment dynamics in this fish assemblage.

4.6 Potential indicators of ecosystem health

Our results suggest that much of the fauna in the wetlands of the Tully-Murray floodplain is in good condition. It may even be resistant to the immense changes in land use that have taken place across the floodplain in the last century. Potential indicators such as benthic invertebrates tend to correlate very closely with habitat changes, which themselves are quite straightforward to monitor (e.g. floods, riparian condition, alien species and weeds). Fish appear to provide a more robust indication of overall wetland condition and, importantly, connectivity.

During sampling in May and October we worked with the Project 3.7.4 team to ground-truth models of wetland connectivity – a key to the ecology of these systems. We found that several species of small-bodied fish were able to traverse waterways with very shallow water (~1cm). We now have the basis for developing indicators and protocols for monitoring, subject to analysis of all our samples and data.

The *Catchment to Reef* program of the former Rainforest and Reef CRCs identified a range of indicators of ecological health in Wet Tropics streams. It was clear that, even in the presence of intense agricultural development, some coastal streams could be 'healthy' (e.g. Behana Creek) as demonstrated by all biological indicators (invertebrates, plants and fish), providing that riparian vegetation remained in good condition and an adequate buffer zone was maintained between adjacent agricultural land and the stream channel (Arthington and Pearson 2007; Mackay *et al.* 2010). The major impact of a reduction in riparian integrity appeared to be a loss of shade, 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 assemblages and 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.

The *Catchment to Reef* stream project recommended the following indicators and factors for stream assessment in the Wet Tropics:

- flow regime of the stream;
- physical condition of the stream sites, including current velocity and 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; and
- proportion of fish abundance due to alien species.

The MTSRF field program examined many of these variables as possible indicators of the ecological health of floodplain lagoons, with reference to the three major faunal groups investigated – zooplankton, benthic invertebrates and fish – discussed below. We also examined habitat features that may act as drivers and/or indicators of health. We hoped to include birds as an easily assessable group with potential value as indicators, but aquatic bird populations were not high on the lagoons, so we abandoned bird surveys. Nevertheless it should be noted that small numbers of birds were observed, especially fish-eating birds such as darters and cormorants, and the rare Great-billed heron.

Results from this MTSRF study provide a benchmark against which improvement or deterioration in wetland condition can be judged. By taking comparable data from a suite of indicators (see Section 4.4) and performing the same analyses, it will be possible to chart the trajectory of lagoon condition with time. This we see as a major area for future development from the current work, using protocols established here (Section 4.9). We would also aim to combine data from other regions (Dry Tropics, Cape York) to help determine thresholds of concern for the northern region.

4.6.1 Zooplankton

Zooplankton live in the water column and are potentially highly susceptible to water quality impact, although the effects of changes to peripheral habitat may only be felt through reduction in the area of open water. Some species (e.g. of Cladocera) have been developed for bioassay monitoring or deriving water quality criteria (e.g. Sunderam *et al.* 1994). As they are easy to sample, and are typically abundant in lagoon habitats, we expected that the zooplankton assemblages would be useful indicators of condition. Despite the ease of sampling, however, laboratory processing can be time-consuming, especially the detailed identification of species. We hoped to find good diversity of taxa that could be readily separated, and thus by analysis of assemblages, provide good indication of lagoon condition.

Our samples were dominated by Cladocera and Copepoda as expected. However, the task of separating any species within these groups by non-specialists became prohibitive, and our analyses based on these groups alone were rather equivocal.

Significant relationships were evident with variables related to morphometry, water quality variables (transparency, hexazinone, dissolved oxygen) and habitat (alien plant species and alien plant cover). The positive relationship with alien plants is hard to explain, given that the plankton were sampled in open water, well away from any direct influence of aquatic plants. The water quality relationships are also equivocal, there being a positive relation with dissolved oxygen (expected) and with hexazinone (unexpected). The lack of diversity of zooplankton may have adversely affected the analysis, but further investigation of these relationships is warranted. Further, the sorting and identification of zooplankton is very time consuming, and for monitoring purposes probably not cost-effective. Therefore, currently the zooplankton are not appropriate health indicators.

4.6.2 Benthic invertebrates

Invertebrate assemblages are very widely used as indicators of condition in freshwater systems, especially streams, because they represent many diverse groups of organisms and so demonstrate a range of response to a range of biophysical variables. For example, the Australian AusRivAS monitoring system (Norris and Hawkins 2000), derived from the British RIVPACS scheme (Wright 1995), has been applied nationally with some success, although not always as usefully as hoped. In Wet Tropics streams, invertebrate diversity was shown to strongly indicate condition (Connolly *et al.* 2007), providing the opportunity to develop a straightforward monitoring framework.

In the Tully-Murray wetlands, invertebrate assemblages clearly reflect condition of the wetlands, but while there are significant relationships, the overall pattern is not strong. The greatest contrast in the invertebrate samples was between the habitats (litter and macrophytes), with water quality a secondary factor. This suggests that for the invertebrate assemblage at least, the lagoons under study (and at the time of the study) were in quite good condition, despite the wholesale clearing and agriculture in their immediate surroundings. It is probable that the hydrological regime, with regular flushing and through-flow, helps maintain reasonable water quality conditions. As has been noted in the *Catchment to Reef* study, water quality is often a secondary factor, habitat quality being much more important.

The lack of major gradients makes determination of indicators and thresholds problematic. Ideally, we would have been able to select otherwise similar wetlands (in size etc.) along a strong gradient of land use (and therefore water quality), against which to examine ecological health indicators. The limited number of sites available and their lack of contrast in terms of water quality has militated against identification of major invertebrate indicators and thresholds of concern (see also Section 4.7). This is a result of the limited number of wetlands, such that our study was of unique rather than representative systems. Gradients may be identifiable from repeated samples of the same wetlands over several years. A least now we have an appropriate benchmark against which to measure change.

4.6.3 Fish

Fish have been advocated as useful indicators of biotic integrity or river health (e.g. Fausch *et al.* 1990; Harris 1995; Karr and Chu 1999; Kennard *et al.* 2005; 2006a,b) because:

- they are almost ubiquitous components of aquatic ecosystems;
- they are relatively long-lived and mobile and therefore reflect conditions over broad spatial and temporal scales;

- local assemblages generally include a range of species representing a variety of trophic levels and therefore integrate effects from lower trophic levels;
- fish are at the top of the aquatic food web and are consumed by humans, making them important for assessing contamination;
- environmental and life history requirements are comparatively well understood; and
- fish are relatively easy to collect, identify and subsequently release unharmed.

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

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

This MTSRF investigation has provided the insight into the ecological condition of Tully-Murray floodplain wetlands and sufficient understanding of fish-environment relationships to underpin the development of fish as indicators of the ecological health of freshwater floodplain wetlands in Queensland's Wet Tropics. .

Summary measures describing the fish fauna of streams (e.g. fish species richness) typically used to assess ecosystem health in freshwater habitats (Kennard *et al.* 2006a, 2006b; Arthington and Pearson 2007) showed no clear relationship to anthropogenic stressors in the Tully-Murray system. This finding may reflect the general suitability of all lagoons as habitat for this 22-member fish assemblage, and also the fact that it was not possible to survey the ideal disturbance gradient (i.e. from near pristine to highly impacted by anthropogenic disturbances such as agriculture chemicals or loss of hydrological connectivity). More impacted lagoons may well support a far less diverse fish assemblage but to date no readily accessible and severely impacted lagoons have been surveyed in the Tully-Murray system. This is good outcome, suggesting that these lagoons are not yet seriously impacted by water quality stressors associated with the surrounding agricultural land use. However, the lack of earlier studies of these water bodies and the Tully-Murray river system in general limits our ability to interpret these results fully in relation to ecological condition of lagoons. It is possible that these floodplain lagoons once supported a more diverse fish fauna, and that contemporary measures of fish diversity actually reflect a history of disturbance that is now difficult to pinpoint and impossible to track over time. Nevertheless, the species recorded in this study are as expected (B. Pusey, pers. comm.).

Although fish species richness was similar across all lagoons (15-17 species), fish assemblage structure (i.e. relative abundance patterns of species) varied from lagoon to lagoon with variation being predictable in terms of habitat structure within individual lagoons, the position of the lagoon both with respect to its distance from the river mouth(s) and from the two major rivers. There was no evidence that water chemistry impacts (e.g. depressed dissolved oxygen) shaped fish assemblages, possibly because anthropogenic chemicals and associated water quality factors did not reach threshold levels that would be of concern to the biota of these floodplain lagoons. This suggestion must be treated cautiously given that sampling was not conducted at dawn when dissolved oxygen is at its lowest concentration

(Butler and Burrows 2007), and because previous reports had highlighted problems with dissolved oxygen in some lagoons (A. Hogan, pers. comm.).

The strong associations between fish species and individual microhabitat types within lagoons suggest that fish respond to the types of habitat disturbance typical of these floodplain wetlands. In particular, certain fish species appear to avoid stands of the alien ponded pasture grass *H. amplexicaulis* and other alien emergent plants. Interestingly, the fish assemblage associated with para grass was more similar to the equivalent native emergent form than were the assemblages associated with either *H. amplexicaulis* or the less common alien group (*Echinochloa* sp., *Cyperus* sp. and *Isolepis prolifera*). Evidently some aspects of microhabitat structure differ between these vegetation associations, a feature worthy of further investigation.

Previous studies in the Wet Tropics and south-eastern Queensland have demonstrated the depressing effects of para grass on fish assemblages (Arthington *et al.* 1983; Arthington *et al.* 1997). Para grass in particular has been shown to reduce structural habitat complexity in small streams by depressing the diversity of native aquatic vegetation (Arthington *et al.* 1983). Similarly, the introduction of *H. amplexicaulis* to the Fitzroy River, Central Queensland, resulted in a marked change in aquatic plant structure along the river margin which has resulted in changes to the littoral fish assemblages including an increase in *X. maculatus* numbers (Houston and Duivenvoorden 2002). Studies using stable isotope tracing techniques have identified that carbon from C4 plants (including grasses like *Urochloa mutica* and sugar cane) is not taken up into aquatic food webs (Bunn and Boon 1993; Forsberg *et al.* 1993). The reason for this is unknown, but neither aquatic herbivores nor detritivores consume C4 carbon directly, as living or dead plant matter, nor indirectly via a microbial loop. This is despite the fact that a high proportion of the organic matter in streams flowing through cane lands is of C4 origin (Bunn *et al.* 1999). For lagoon fish assemblages, any such consequences of the high densities of alien pasture grasses warrant further investigation.

It is interesting to speculate why *Hymenachne* and para grass are so well-established in floodplain lagoons and what environmental factors govern their relative abundance and growth rates in these systems. Certainly the physical nature of lagoon environments is suitable for their growth, as the long stolons of these grasses are capable of forming large beds extending some distance out from the banks and forming a dense matted cover over the water surface (Arthington *et al.* 1983; Arthington *et al.* 1997). The rapid adaptation of *Hymenachne* to changes in water level and nutrient update may give it a competitive advantage over native wetland vegetation (Sydes 2009). Januchowski (pers. comm.) has shown that the extent of invasion of *H. amplexicaulis* in wetlands throughout the Tully-Murray catchment is related to land use and the nature of soil with those areas of the catchment supporting intensive agriculture (e.g. sugar cane, banana plantation), in association with alluvium soils, more likely to support *H. amplexicaulis*. Dissolved inorganic nitrogen occurs in higher concentration in wetlands draining intensive agriculture (Bainbridge *et al.* 2009) suggesting that agriculture has contributed to the proliferation of *Hymenachne* across the catchment.

These results suggest that DIN may also be a useful indicator of anthropogenic influence on the ecological condition of floodplain water bodies. It also seems possible that low dissolved oxygen (DO) levels and/or hypoxia sufficient to discourage fish occupancy may develop within dense beds of *Hymenachne* and para grass. From these observations it is reasonable to recommend including DIN, DO, *Hymenachne* and para grass cover, and fish assemblage structure, as potentially powerful indicators of lagoon ecosystem health, and to undertake further work on these biophysical relationships and processes in lagoons of contrasting character.

This study has identified the recruitment styles of ~70% of the 22 fish species collected over two years, and confirmed the timing in the appearance of new recruits within the lagoon fish assemblage These findings add to the growing body of knowledge that coastal floodplain lagoons provide habitat for the early life-history stages of a range of freshwater fish species including iconic species (e.g. the +1 year class of barramundi – *Lates calcarifer*), species of conservation significance (e.g. the larvae of *Pseudomugil gertrudae*) and migratory species (e.g. *Hypseleotris compressa*) (Pusey *et al.* 2004; Gehrke. and Sheaves 2006). The small population size of Macculloch's rainbowfish (*M. maccullochi*) is of concern for the conservation of the species in the Tully-Murray catchment given the high presence of habitat attributes (e.g. highly vegetated slack waters) that should favour their wider occurrence in water bodies across this system. Research devoted to understanding the factors that shape population size (e.g. reproductive biology and habitat requirements throughout life history) would be a constructive pathway in the conservation management of this species.

For those species that migrate from saltwater to preferred adult freshwater habitats during their early life history, the extent of accessibility between salt- and freshwater habitats is an important factor contributing to the magnitude of temporal variation in fish populations and assemblages within individual lagoons. Differences in the extent of hydrologic connectivity among individual lagoons and the estuary (via the stream network) contributed to the differences in both fish assemblages and the size structure of individual species among water bodies (Gehrke and Sheaves 2006; Karim *et al.* 2010b). The influence of hydrological connectivity also applies to those species within the assemblage that migrate between lagoons and spawning habitat contained within the freshwater section of the stream network including the artificial cane drains. Flow regime alterations and human influences on floodplain hydrology typically translate into changes in floodplain habitat structure, connectivity and quality and, hence, ecological responses of aquatic biota (e.g. Overton 2005). However, little is known about the interactions between natural and altered flow regimes, hydrological connectivity and freshwater and estuarine fishes in the tropical floodplains and wetlands of the Queensland Wet and Dry Tropics, which are likely to be impacted by marked losses in habitat, connectivity and water quality brought about by floodplain modifications and climate change. This is an area requiring further research in both the Wet and Dry Tropics.

This study has enhanced our understanding of the drivers of temporal variation in the fish assemblages found in floodplain wetlands of the Tully-Murray including the contribution to this variation by those species that migrate between habitats. One approach to using fish as indicators of ecosystem health is to take advantage of this knowledge about the temporal dynamics of the fish assemblage and apply the established pattern of change over time as the reference (or near reference) condition. However, this approach requires a greater understanding of the extent of physical and hydrological impediments to the movement of aquatic organisms between individual lagoons and the surrounding aquatic habitat. It may then be possible to establish a 'connectivity disturbance gradient' embracing the full range of connectivity potential in the Tully-Murray floodplain landscape. Information contained in CSIRO Sustainable Ecosystems audit of physical barriers (e.g. culverts, flood mitigation works etc.) to fish passage in the Wet Tropics bioregion (Lawson *et al.* 2010) could contribute to the development of a gradient in 'connectivity disturbance'.

Early results from this MTSRF project suggest that a suite of small-bodied fish species are able to move through shallow streams and cane drains at depths to about one centimetre. However, data on fish movements are very limited and warrant further investigation. The *Catchment to Reef* program of the former Rainforest and Reef CRCs found that changes in stream habitat structure associated with immersion-tolerant alien vegetation appeared to inhibit the upstream migration of fish species with a marine or estuarine interval in their life history (Pusey *et al*. 2007b). Natural streams and cane drains on the floodplains of Wet and Dry Tropics rivers appear to be vulnerable to plant invasion and blockage such that fish movements may be inhibited. This aspect of connectivity warrants further investigation, as do other barriers to fish movement.

Alien fish species can be useful indicators of reduced stream health (Kennard *et al.* 2005; Arthington and Pearson 2007). However, only one alien species was collected from Tully-Murray lagoons – the platy (*Xiphophorus maculatus*) – an aquarium escapee often found in Queensland's coastal streams (Arthington *et al.* 1983). Although the platy occurred in nine of the ten lagoons investigated, its abundance was variable, being highest in lagoons with a high proportion of leaf litter habitat (i.e. Carroll's Lagoon and Bunta Lagoon). 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. Given the large pool of alien species in the region (Pusey *et al*. 2004b; 2007b) and recent trends (e.g. the spread of the cichlids *Tilapia mariae* and *Oreochromis mossambicus*), it seems likely that the number and abundance of alien species will increase over time in Wet Tropics streams and wetlands. Thus the inclusion of alien fish species will be even more important in the future and their utility as an indicator of stream and wetlands degradation warrants further investigation.

4.7 The Cardwell Values and Threats assessment

The Cardwell Values and Threats (CVT) assessments were partly corroborated by the independent results of this MTSRF study. Significant correlations between the CVT scores and biological variables are shown in Table 4.1. While the relationships were not comprehensive, some are quite strong, despite the apparent lack of extremes in the lagoon gradients (see Section 4.7). It is noteworthy that they are present at all, indicating that some broad assessments can provide a reasonable reflection of ecological conditions, especially in relation to the fish fauna. Values of the wetlands for vertebrates other than fish were not included, and it should be noted that these will be affected by habitat values such as riparian width and weed invasion that might not show up in the purely aquatic system.

With regard to indicators, it is clear that there is connection between the broad values and threats and the ecological status of the wetlands, and that any ecosystem health monitoring system should make use of metrics such as the CVT ones indicated here.

Table 4.1: Significant relationships (Pearson correlation) between Cardwell Values and Threats (CVT) scores and the three main biological components. X = $p < 0.05$; XX = $p < 0.01$.

	Biological component		
CVT category	Plankton	Invertebrates	Fish
Values			
Condition			XX
Taxa			XX
Fishery		X	XX
Wetland		X	XX
Indigenous		XX	XX
Size		XX	XX
Recreation		X	X
Birds		X	X
Vegetation		XX	X
Threats			
Land use 1			XX
Invasive spp.		X	XX
Land use	X		X
Connectivity			X
Land use 2			X
Water quality			X
Hydrology		XX	

4.8 Thresholds of concern

Project 3.7.3 has continued to assess existing approaches to determining and representing thresholds of potential concern. This project aimed to measure spatial and temporal variability of biophysical indicators in floodplain lagoons along natural environmental gradients and gradients of disturbance. The field study in the Tully-Murray catchment was designed such that stressor-response relationships along gradients of disturbance (supported by data from laboratory trials and the literature) would help to identify thresholds – points along each disturbance gradient where ecological changes of scientific or management concern become apparent. Early recognition of such thresholds can guide management actions to alleviate the associated stressors, or may signal environmental factors that could be restored to a more natural condition to improve the health of the aquatic ecosystem (Biggs and Rogers 2003; Arthington *et al*. 2006; Poff *et al*. 2010). Arthington *et al.* (2006) and Poff *et al*. (2010), discussing similar concepts with respect to flow regime, suggest that the 'reference state' has its own natural variation, and that it is change beyond this range that defines thresholds.

Thus, the concept of thresholds can be very simple – a species or a system is sustained while some environmental variable changes, until a critical point, when the species or system starts to show a negative response (or collapses altogether) (Figure 4.2). Previous work in the Wet Tropics has documented some thresholds for selected species and variables, including dissolved oxygen (Pearson and Penridge 1987; Connolly *et al.* 2004), nutrients (Pearson and Connolly 2000), ammonia (Økelsrud and Pearson 2007), substrate disturbance (Rosser and Pearson 1995), and sediment deposition (Connolly and Pearson 2007). Recently a large project by the Australian Centre for Tropical Freshwater Research (ACTFR) documented critical levels of dissolved oxygen and guidelines for many species of tropical Australian freshwater fish (Butler *et al.* 2007; Butler and Burrows 2007). This type of work is essential for understanding species' responses and thresholds, and in the case of the ACTFR work, developing guidelines against selected criteria (in this case, dissolved oxygen).

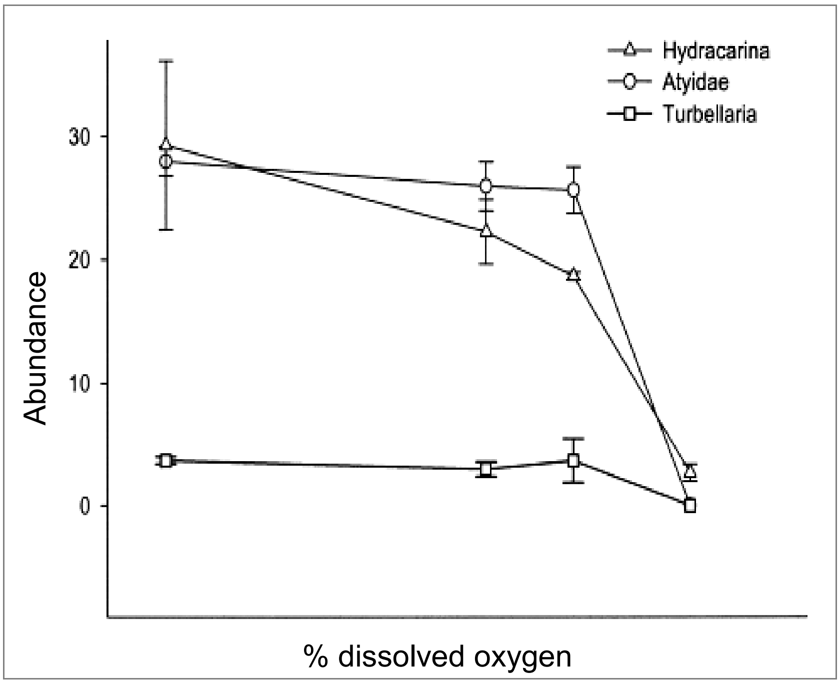

Figure 4.2: Threshold pattern for dissolved oxygen and several taxa from Wet Tropics streams (from Connolly *et al.* 2004).

However, in ecosystems there are multiple factors affecting species, some of which act on similar axes, but others of which act independently. Some factors may have no effect while others have a linear or gradual effect such that no clear threshold exists (Figure 4.3). In the wetland situation, for example, it is possible that light levels have a direct linear effect on plant abundance, but plants will also be affected by changes in habitat, nutrients, concentration of herbicide, etc., which may have non-linear effects. In the Tully-Murray we measured many habitat and water quality variables, each of which may have independent effects on each species of plant and animal. Moreover, many variables will act differently on different life stages of the biota, so the end result is a composite response to these multiple effects. In some situations a single variable overrides all others. Dissolved oxygen is one such variable that can control presence or absence of fish (as in Figure 4.2).

Our Tully-Murray analyses indicate, however, that many water quality variables and habitat variables act in concert such that there were significant relationships between the ordination axes and many of the variables. It is thus clear that the multiple responses to multiple variables are expressed quite generally. It is evident that in the Tully-Murray Wetlands, as in our Wet Tropics stream study (Arthington and Pearson 2007) many of the variables that affect the biota can be resolved into ‘habitat’ and ‘water quality’ composites. Thus we see a gradient of response to a gradient of land-use impact, defined by the variable composites.

The fact that we can pick up response to the various factors that we have measured indicates that some composite threshold has been crossed. Without reference sites it is difficult to establish where that threshold might be situated (Figure 4.4). Our best guess at sites closest to reference condition are notional lagoons 1 and 6; but in this schema they are somewhat removed from reference (we cannot be sure how far). This is how we perceive the Tully-Murray wetlands. Our goal in management is to have sites progressively take a trajectory up and beyond the notional ‘threshold of concern’.

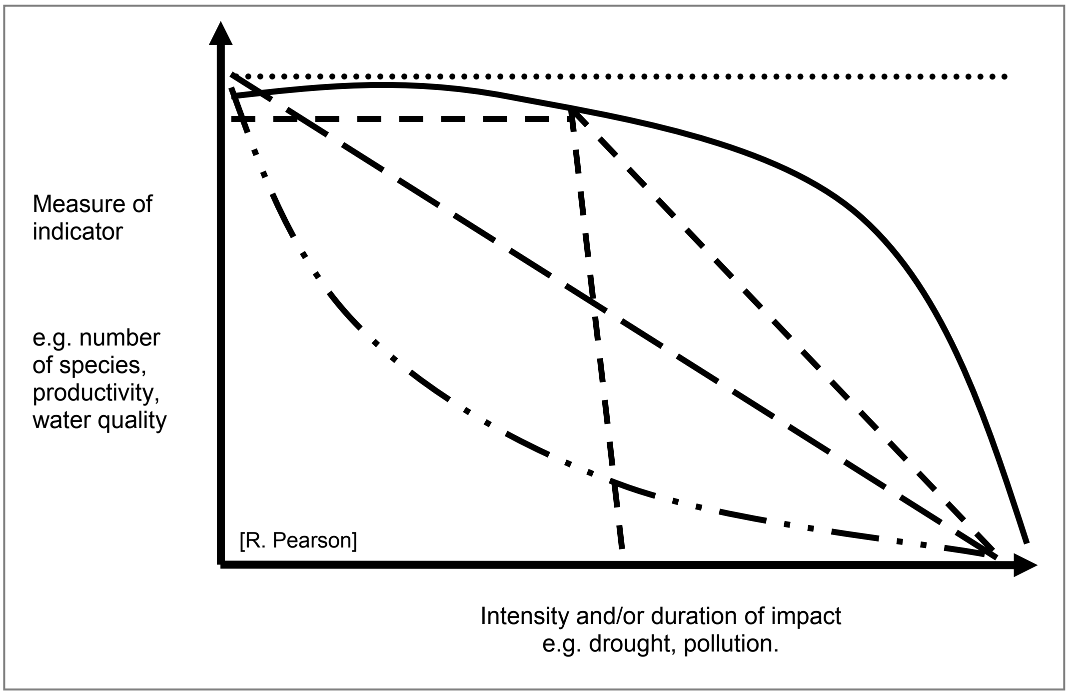

Figure 4.3: Some alternative threshold patterns for different indicator and pressure variables.

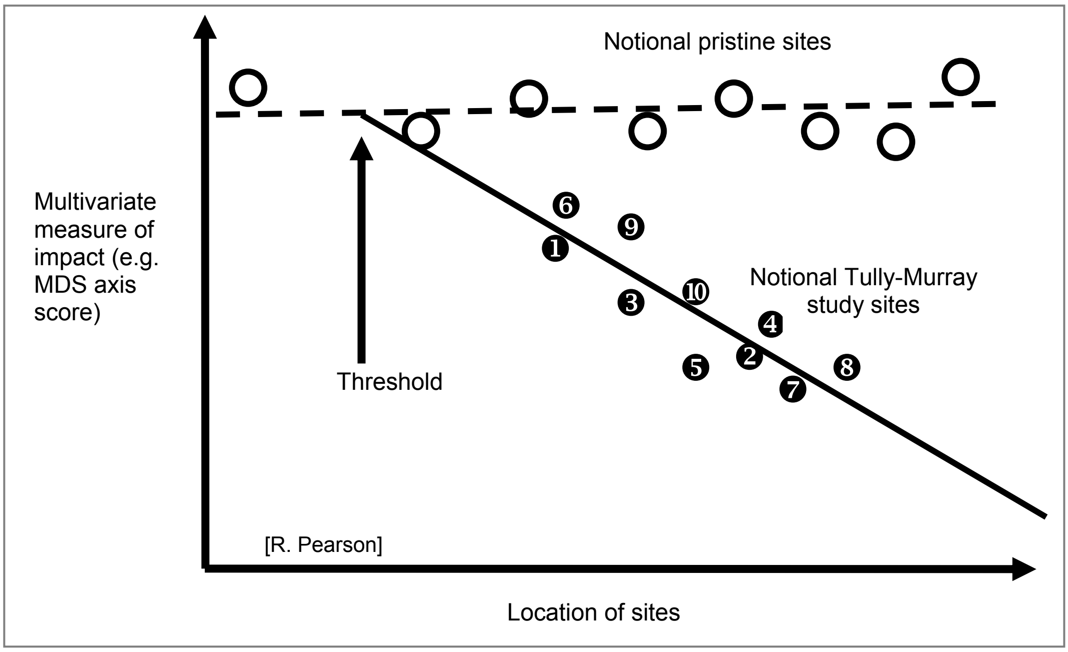

Figure 4.4: Notional position of Tully-Murray sites on an impact/location gradient.

Clearly, measuring such a threshold in the wetlands is not as easy as in the oxygen experiment (Figure 4.2). Furthermore, some parts of the gradient are entirely natural (e.g. distance from the coast; salinity) although even they may be affected by development (distance to the coast along channels might be greatly reduced by drainage works). Therefore natural gradients need to be removed from consideration, as we did in the stream study (Arthington *et al.* 2007). In that study multiple replicate sites, and comparisons between more and less impacted catchments, facilitated analysis and interpretation. In the Tully-Murray wetlands we were very constrained by the low number of sites available, creating a gradient of sites that was truncated at both pristine and disturbed ends of the spectrum, so our interpretation is necessarily more equivocal.

Nevertheless, as discussed above, our results provide a substantial benchmark against which improvement in the ecological condition of floodplain lagoons can be evaluated.

4.9 Links between catchment and reef

The lagoons investigated in this project are discrete permanent waterholes, but they are not disconnected from other water bodies or from the landscape. Moreover, they have clear connections with riverine and marine systems. The lagoons have many characteristics in common, but each lagoon also has its own characteristics, partly reflected in the biota. Each links to other systems by the network of natural waterways and drains, and each links to its immediate surroundings and catchment through direct inputs of materials, and through surface and groundwater flow. The lagoons might, then, be regarded as replicated systems nested within a pool of possible systems, lying within a catchment. Thus we expect the catchment to influence the condition of the lagoons via overland, instream and groundwater transport. The character of the lagoons is therefore a reflection of the character of its surroundings. The extent of these surroundings (immediate or distant) depends on connectivity via the surface and subsurface water regime.

Catchment links can vary substantially from system to system. With the regular flows in the Tully-Murray system, there is frequent if not constant connectivity between elements of the freshwater landscape. In contrast, where flows are much more seasonal, as in the Burdekin system (Pusey and Arthington 1996; Preite and Pearson, Blanchette and Pearson, unpub. data), or at a much smaller scale, in intermittent stream pools (Smith and Pearson 1987), pools and lagoons can quite substantially diverge in character as local factors come into play. The lack of differentiation observed in the aquatic communities in the Tully-Murray reflects this regular or constant connectivity and at least partial melding of characteristics.

The sites described here form part of a complex of wetlands in the Tully-Murray floodplain, which interlink across the landscape. They also link into the Tully and Murray Rivers and estuaries and thereby into the GBR lagoon. The catchment and its waterways are traditionally regarded as the source of materials (contaminants) for marine waters, and over recent years have received much attention (including through the MTSRF program) focused on reducing anthropogenically enhanced delivery of these materials (Figure 4.5a). This focus on delivery has taken little notice of the importance of streams and wetlands in retaining these materials through a variety of processes, especially in non-flood periods (Figure 4.5b), and they have been regarded as simple conduits. They have received relatively little attention for their own values (the present MTSRF Projects 3.7.3 and 3.7.4, and some recent CSIRO research on nutrient processing and in-catchment fish barriers – J. Wallace, pers. comm. – are exceptions). Moreover, there has been little cognisance of the importance of catchment waterways for components of the fauna that spend parts of their life there. Although our sampling only touched on this issue (e.g. the empire gudgeon, a major species in the lagoons, breeds in the estuaries and the near-shore marine environment), we know that species such as barramundi require the extensive wetlands to sustain their young stages, and that others such as mangrove jack use the lower freshwater reaches of rivers (P. Godfrey, unpub. data). The extent of this connectivity between catchment waterways and reef lagoon needs to be quantified so that we might manage it as part of the 'Greater Barrier Reef' system. Figure 4.6 aims to capture the range of scales that are involved in connectivity.

4.10 Representativeness of the Tully-Murray in the GBR catchment

We focused our research on permanent lagoons because they were perceived to have high conservation value by virtue of the habitat they provided, their likely connections with estuarine and marine environments (important for fish), and their limited extent. The Tully-Murray system was selected for study for several reasons:

- there was a substantial floodplain with potentially a large number of wetlands to investigate;
- the system is centrally located in the wetter part of the Wet Tropics and therefore might represent core Wet Tropics characteristics;
- substantial research on water quality was already in progress in the catchment, by CSIRO and ACTFR;
- CSIRO was undertaking important hydrological investigations in the catchment (including MTSRF Project 3.7.4), and collaboration between this project and that one on hydro-ecological studies was planned; and
- it was expected that the Tully-Murray wetlands would provide a good representation of Wet Tropics wetlands generally.

The degree of representation is an interesting issue, because it appears that there is a very limited number of lagoonal systems on the Wet Tropics floodplains. This is because either they never existed when European settlement began or because they have been heavily modified or removed by agricultural development. Thus the number of freshwater lagoons on the floodplains of the Daintree, Barron, Russell-Mulgrave, Johnstone and other rivers are very limited in number. The Herbert system has some lagoons comparable with those on the Tully-Murray floodplain, as does the Normanby system.

It is interesting to consider the extent of floodplains and wetlands during the period of lower sea level during the late Pleistocene, when the coastline was located on the continental shelf, where the outer reefs of the GBR are now located. At that time the GBR lagoon did not exist but was replaced by floodplain. A rough estimate of the change over the last 18,000 years in the Port Douglas – Townsville area is a reduction in width of the floodplain from about 90 km to 12 km, and in area from 26,000 to 4,000 km^2. This would represent a dramatic reduction in the area of wetlands and number of lagoons. Therefore, rather than being representative we now regard the lagoons as remnant – remaining firstly after rise in sea level and secondly after agricultural encroachment.

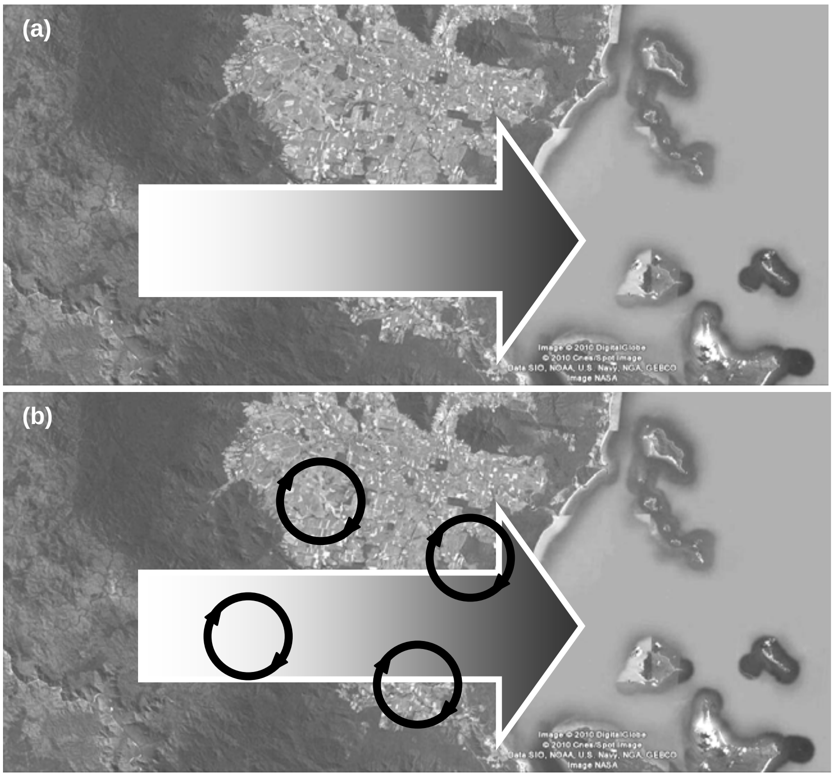

Figure 4.5: General perception of the relationship between catchment and reef (a) – the catchment is the source of contamination; while in reality (b) there are numerous interactions that modify the delivery of materials to the reef.

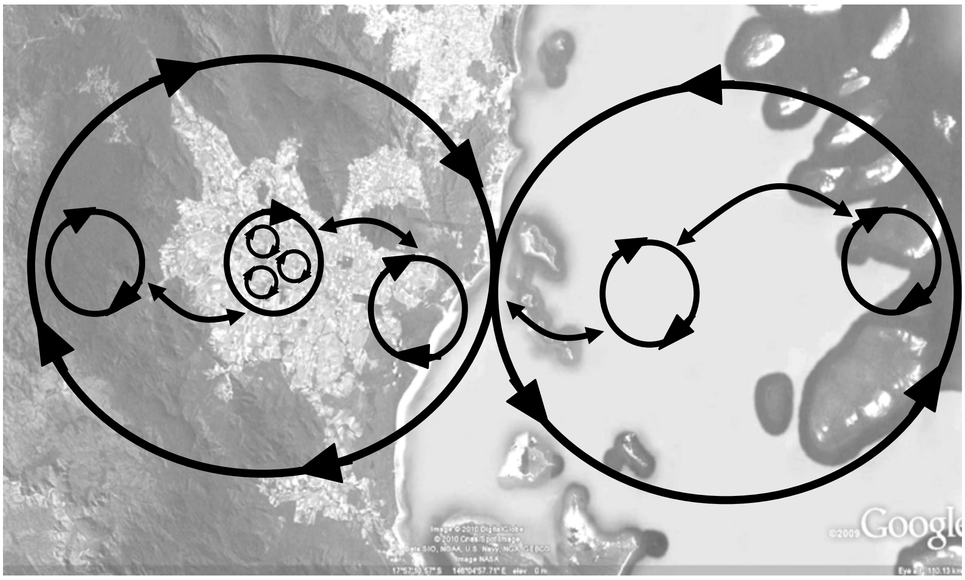

Figure 4.6: Interactions in and between catchment and reef are complex and occur at a number of scales, with linkages between components at each scale.

Nominally comparable wetlands in the Herbert floodplain appear to be rather different in character from those of the Tully-Murray. Waterways in the Herbert have typically been more contaminated than those in the Tully-Murray, by organic run-off from fields and by weed invasion (Pearson *et al.* 2003). Their condition is therefore not as good as that of the Tully-Murray lagoons. This difference is probably largely due to the different flow regimes – the Herbert has a drier wet season that the Tully-Murray, so the latter's regular flushing does not occur in the Herbert. The contrast on the Burdekin floodplain is even more pronounced, it being in the Dry Tropics, although irrigation water does provide some dry season flow there. The difference between these catchments is indicated by their annual mean rainfall: Tully (sugar mill) 4,100 mm, Ingham (Victoria Mill) 2,016 mm, Ayr (Shire Council) 1,075 mm.

The special nature of the Tully-Murray wetlands as a unique assemblage of Wet Tropics habitats, with functional links to the GBR lagoon, therefore needs to be specially recognised.

5. Framework for monitoring and management

In the GBR catchment there have been previous developments of region-specific monitoring systems and protocols (Pearson and Penridge 1979, Arthington *et al*. 2007), in addition to *AusRivas*, a national stream health monitoring system based on invertebrates (e.g. Smith *et al*. 1999, Davies 2000; Norris and Hawkins, 2000). Regional protocols are required when national systems are too broad-based to address local and regional needs (Connolly *et al.* 2007). Some regional protocols have been taken up for short periods, but currently there appear to be no mechanisms or resources available for systematic monitoring of ecosystem health of GBR catchments. In contrast, the GBR itself appears well served by established on-going monitoring overseen by the Great Barrier Reef Marine Park Authority. As there is no identifiable client with whom to develop monitoring protocols, despite strong interest and good will from many interested parties (Pearson *et al*. 2010), we have not developed a monitoring protocol for fresh waters here. However, on the basis of previous research, our *Catchment to Reef* project and the current project we here make some recommendations regarding development of a manual of appropriate protocols.

Bunn *et al*. (2010) have recently made recommendations regarding monitoring protocols for freshwater systems. They aim for monitoring and reporting as part of an adaptive process linked to values and objectives, and informed by rigorous science, that guides management and responds to stakeholders. Monitoring needs an understanding of the probable causal influences on the condition of waterways. They provide documentation of a large-scale monitoring program in south-eastern Queensland, which was supported by multiple collaborating sponsors (local government, etc.). Unfortunately, it is unlikely that anything comparable in scope can be applied to the GBR catchment because of the much lower population (by an order of magnitude) and concomitant lower level of resources available. Nevertheless, Bunn *et al*. (2010) provide a very cogent guide to development of monitoring programs.

5.1 General monitoring considerations

- Issues of interest need to be explicit – for example, is the goal to maintain high water quality for delivery to Reef waters, to sustain properly functioning catchment ecosystems, to sustain biodiversity in the catchment, or all of these?
- A useful framework is to consider the values of and threats to ecosystems to provide an appropriate context for monitoring.
- Monitoring needs to be question-driven to ensure explicit addressing of potential issues, for example:
  - are ecosystems in question supporting the full range of expected habitats?
  - are water quality metrics within normal levels of variability?
  - are ecosystems supporting a normal biota (plants, invertebrates, fish, etc.)?
  - are normal levels of ecosystem processes being maintained (e.g. productivity, connectivity and dispersal)?
  - is normal temporal and spatial variability and representativeness being maintained?
  - can likely causes of departures from normal or expected conditions?
  - are there clear management actions that can follow from monitoring investigations – i.e. is there a process in place?
- Benchmarks are needed to answer such questions: they might include undisturbed (and otherwise comparable) reference sites or, in their absence, trajectories of improvement as suggested for the Tully-Murray floodplain lagoons.

5.2 Ecosystem health monitoring of streams

- Our research on streams in the Wet Tropics (Pearson and Penridge 1987; Pearson *et al.* 2003; Arthington *et al.* 2007) and in the Mackay-Whitsunday region (Clayton and Pearson 1996; Leonard 2009) has greatly informed our knowledge of how these ecosystems respond to human impact.
- Details of the preferred monitoring strategies can be found in the above references and via the Web at http://www.rrrc.org.au/catchment-to-reef/downloads/C2R-Arthington-A-et-al-2007-Biological-Indicators-Wet-Tropics-Streams.pdf.
- Briefly, we showed that ecosystem health could be monitored by measuring a suite of variables at multiple sites along natural stream gradients as follows:
  - Habitat variables, such as flow regime, flow modification, stream geomorphic characteristics, riparian extent and condition, aquatic vegetation and alien plant infestation, excessive algal growth, leaf litter, etc.
  - Water quality characteristics, especially temperature, conductivity, turbidity, suspended solids, pH, dissolved oxygen, nutrients (mainly species of N and P) and short-, medium- and long-term variability in these metrics.
  - Invertebrate diversity (species and family levels) – particularly good at the site/reach level.
  - Fish species diversity – particularly good at the sub-catchment level.
  - Abundance and diversity of alien fish species.
  - Aquatic plants were not very useful for monitoring (apart from their habitat associations with the rest of the biota) because of their high level of variability (Mackay *et al.* 2010)
  - Monitoring in contrasting seasons (late wet/early dry and late dry) is required to understand extremes of conditions.

5.3 Ecosystem health monitoring of floodplain lagoons

- Previous research on floodplain lagoons (e.g. Pearson *et al.* 2003; Perna and Burrows 2005) and the present project, (MTSRF Project 3.7.3) in association with MTSRF Project 3.7.4 (floodplain hydrology) have greatly informed our knowledge of the nature of these ecosystems and their biota, and how they respond to human impact.
- We have shown here that there are neither good reference (undisturbed) sites or highly impacted sites, in terms of aquatic biota, so very strong gradients of condition are not evident in the Tully-Murray lagoons. Higher levels of disturbance were evident in the lagoons in the Herbert and Burdekin systems (Pearson *et al.* 2003; Perna and Burrows 2005).
- Nevertheless, gradients in environmental variables and significant associations of the biota with them do exist across all these systems, so we are able to outline approaches to monitoring.
- Ecosystem health can be monitored by measuring a suite of variables at multiple sites and times, with some exceptions, as follows:
  - Habitat variables, such as flow regime, flow modification, lagoon geomorphic characteristics (including size and depth), aquatic vegetation and alien plant infestation riparian extent and condition, alien plant infestation, leaf litter, etc.
  - Benthic habitat (plants vs. litter) and alien plant infestation were particularly important variables for invertebrates and fish, respectively.

- Water quality characteristics, especially temperature, conductivity, turbidity, suspended solids, pH, dissolved oxygen, nutrients (mainly species of N and P) and stratification; and short-, medium- and long-term variability in these metrics.
- Invertebrate diversity (mainly family levels) and assemblage structure – provides a good benchmark with regard to habitat and water quality.
- Fish species diversity and assemblage structure – provides a good benchmark with regard to habitat and water quality, and to connectivity and normal movements of fish.
- Abundance and diversity of alien fish species.
- Zooplankton assemblages were not very useful for monitoring because of their low diversity and the time involved in sample processing; however, presence or absence of zooplankton could be a useful and cost-effective measure in the event of severe deterioration of lagoon condition.
- Monitoring in contrasting seasons (late wet/early dry and late dry) is required to understand extremes of conditions, including connectivity and success of dispersal/migratory activity.

5.4 Ecosystem health monitoring of other wetland habitats

- Our MTSRF project was mainly restricted to the streams of the Wet Tropics and the lagoons of Wet Tropics floodplains, but previous research on floodplain lagoons in the Burdekin and Herbert systems system (e.g. Pearson *et al.* 1995; Pearson *et al.* 2003; Perna and Burrows 2005) in conjunction with results of the present project, allow comment on monitoring of floodplain lagoons across the GBR catchment.
- For floodplain lagoons, the suite of variables of utility in monitoring is the same as indicated above for Wet Tropics lagoons. While the character of Wet and Dry Tropics systems differ greatly (see Introduction), differences are captured in the recommended suite of variables (including flow regime, temporal variation, etc.).
- Riverine lagoons in the Dry Tropics (waterholes that remain when rivers cease to flow in the dry season) are the subject of two MTSRF-related PhD projects – one completed on water quality and algal dynamics (Preite 2009), the other continuing on invertebrate dynamics and food webs (M. Blanchette, unpub.). Results are not finalised but indications of metrics for ecosystem health monitoring are as follows:
  - Habitat variables, such as flow regime, flow modification, lagoon geomorphic characteristics (including size and depth), aquatic vegetation and alien plant infestation riparian extent and condition, alien plant infestation, leaf litter, etc.
  - Benthic habitat (edge, plants, sand, litter, riffle) and alien plant infestation are particularly important variables for invertebrates.
  - Water quality characteristics, especially temperature, conductivity, turbidity, suspended solids, pH, dissolved oxygen, nutrients (mainly species of N and P), chlorophyll and stratification; and short-, medium- and long-term variability in these metrics.
  - Invertebrate diversity (mainly family levels) and assemblage structure – provides a good benchmark with regard to habitat and water quality.
  - Variability among lagoons and sub-catchments requires that multiple sites be monitored.
  - Algae are time-consuming to identify and show mixed signals with regard to ecosystem health, so are not currently useful for ecosystem monitoring.

6. Future research needs

The recent MTSRF workshop, held in association with the Annual Conference (May 2010) and involving natural resource managers, land owners and scientists, identified the need for several areas of research to follow on from the achievements of Project 3.7.3 (*Indicators and thresholds* and Project 3.7.4 (*Wetlands and floodplains: connectivity and hydro-ecological function*) (Pearson *et al.* 2010). A key recommendation is that wetland research needs to be extended geographically to further validate indicators of wetland health elsewhere within the Wet Tropics and into the Dry Tropics (e.g. the Burdekin River system). The EPA also identified the need for broader wetland monitoring for Queensland as to date only one major type of wetland (apart from streams) has been studied in relation to indicators of ecological status (i.e, the palustrine wetlands of the Tully-Murray floodplain, and one lacustrine wetland, Kyambul Lagoon).

The following areas of further research were specifically identified:

1. Fish corridors and movement in floodplains and cane drainage systems.
2. Ecological condition of fish assemblages in floodplain wetlands given past wetland losses.
3. Macroinvertebrate monitoring program designs.
4. Focus on 'whole of catchment' research.
5. Grazing land research; methods and framework for monitoring and management.
6. Riparian health and its effect on water quality.
7. Groundwater dynamics and relationship to ecosystem health.
8. Environmental flows for the Wet Tropics.

Additionally, the research team suggests a need for the following:

9. Development of wetland monitoring protocols for other wetland types (only streams and lagoons included so far) for Queensland.
10. Quantification of ecosystem services provided by streams and wetlands.
11. Validating models in other Wet Tropics systems, in the Dry Tropics, and other contiguous systems (e.g. Cape York).
12. Identifying important taxa and ecological processes that are disturbance-prone.
13. Investigation of how drainage and similar works affect connectivity thorough barriers and through changing the distance between wetlands and river mouth.
14. Investigation of drains not only as connectors, but also as habitats and sites for important ecosystem processes.
15. Explicit tracking of fish movement within and across the floodplain to identify vital fish corridors between wetlands and the rivers, and across the agricultural landscape.
16. Quantitative assessment of the degree of historical and recent habitat loss for fish in the GBR catchment, and the ecological condition of fish assemblages in response.
17. Status and condition assessment for all GBR wetlands and waterways.
18. Identification of priority wetlands in Queensland.

The MTSRF workshop recommended further research into the ecology of fish movements in and out of floodplain wetlands and a better understanding of cane drains as connectivity corridors and even as habitat to replace that destroyed during land-use developments on floodplains (Pearson *et al.* 2010). Freshwater and estuarine fishes are clearly useful as indicators of wetland ecological status and potential as habitat during recruitment processes.

Catchment to Reef and MTSRF research demonstrate that fish can be affected by natural environmental gradients and anthropogenic stressors. One of the chief potential stressors on fish species and assemblages in riverine systems is alteration of the natural flow regime (Poff *et al.* 1997; Bun and Arthington 2002). In floodplain systems, flow regime alterations and human influences on floodplain hydrology typically translate into changes in floodplain habitat structure, connectivity and quality and hence, ecological responses of aquatic biota (e.g. see Overton 2005). However, little is known about the interactions between natural and altered flow regimes, hydrological connectivity and freshwater and estuarine fishes in the tropical floodplains and wetlands of the Queensland Wet and Dry Tropics, which are likely to be impacted by marked losses in habitat, connectivity and water quality brought about by floodplain modifications and climate change.

MTSRF collaborative research between CSIRO Land and Water, James Cook University (JCU) and Griffith University (GU) has developed an exciting new application of hydro-dynamic models to quantify the connectivity of floodplain wetlands and the impact of this on fish species, which use connectivity pathways to access fluctuating wetland resources (Fazlul *et al.* in review). These models offer the potential to predict the ecological implications of various scenarios of floodplain hydrological alteration on fish assemblages and recruitment patterns in wetlands. To further develop this area of research, a Post-Doctoral Fellowship application has been submitted to CSIRO to develop the following research program in collaboration with JCU and GU:

1. carry out new field measurements of fish (species, numbers, reproductive status, population age structure) in a range of wetland types,
2. identify wetland types, habitat features and water quality characteristics that promote fish diversity and recruitment within wetlands,
3. correlate spatial and temporal patterns of fish diversity and recruitment within wetlands with field evidence of directional movements along particular connectivity pathways,
4. seek relationships with calculations of the timing, frequency, duration and spatial pattern of wetland connectivity (to be carried out by CSIRO Land and Water), and
5. develop models that clarify the hydro-ecological links between catchment and reef environments.

If successful, this CSIRO Post-Doctoral Fellowship would enable more extensive research it the Wet and Dry Tropics, linked to further validation of fish as indicators of wetland and riverine health and trend.

7. Literature cited

ANZECC and ARMCANZ (2000) Australian and New Zealand guidelines for fresh and marine water quality. National Water Quality Management Strategy Paper No. 4. Australian and New Zealand Environment and Conservation Council (ANZECC) and Agriculture and Resource Management Council of Australia and New Zealand (ARMCANZ), Canberra.

Armour, J.D., Hateley, L.R. and Pitt, G.L. (2009) Catchment modelling of sediment, nitrogen and phosphorus nutrient loads with SedNet/ANNEX in the Tully-Murray basin. *Marine and Freshwater Research* 60: 1091-1096.

Arthington, A.H. and Pearson, R.G. (eds.) (2007) Indicators of Ecosystem Health in Wet Tropics Streams. *Final Report, Task 3, Catchment to Reef Research Program*. Cooperative Research Centre for Tropical Rainforest Ecology and Management (Rainforest CRC) and Cooperative Research Centre for the Great Barrier Reef World Heritage Area (CRC Reef), 247 pp.

Arthington, A.H., Milton, D.A. and McKay, R.J. (1983) Effects of urban development and habitat alterations on the distribution and abundance of native and exotic freshwater fish in the Brisbane region, Queensland. *Australian Journal of Ecology* 8: 87-101.

Arthington, A.H., Marshall, J., Rayment, G., Hunter, H. and Bunn, S. (1997) Potential impacts of sugarcane production on the riparian and freshwater environment. In: B.A. Keating and J.R. Wilson (eds.) *Intensive Sugar Cane Production: Meeting the Challenges Beyond 2000*, pp. 403-421, CAB International, Wallingford, UK.

Arthington, A.H., Balcombe, S.R., Wilson, G.A., Thoms, M. and Marshall, J. (2005) Spatial and temporal variation in fish-assemblage structure in isolated waterholes during the 2001 dry season of an arid-zone floodplain river, Cooper Creek, Australia. *Marine and Freshwater Research* 56: 25-35.

Arthington, A.H, Bunn, S.E., Poff, N.L. and Naiman, R.J. (2006) The challenge of providing environmental flow rules to sustain river ecosystems. *Ecological Applications* 16: 1311-1318.

Arthington A.H, Baran E., Brown C.A., Dugan P., Halls A.S., King J.M., Minte-Vera C.V., Tharme R.E. and Welcomme R.L. (2007) *Water requirements of floodplain rivers and fisheries: existing decision support tools and pathways for development*. Comprehensive Assessment of Water Management in Agriculture, Research Report 17. International Water Management Institute, Colombo, Sri Lanka (URL: http://www.iwmi.cgiar.org/assessment/files_new/ publications/CA%20Research%20Reports/CARR17.pdf)

Arthington A.H., Naiman, R.J. McClain, M.E. and Nilsson, C. (2010) Preserving the biodiversity and ecological services of rivers: new challenges and research opportunities. *Freshwater Biology* 55: 1-16.

Bainbridge, Z.T., Brodie, J.E., Faithful, J.W., Sydes, D.A. and Lewis, S.E. (2009) Identifying the land-based sources of suspended sediments, nutrients and pesticides discharged to the Great Barrier Reef from the Tully-Murray Basin, Queensland, Australia. *Marine and Freshwater Research* 60: 1081-1090.

Balcombe, S.R, Arthington, A.H., Foster, N.D., Thoms, M., Wilson, G.G. and Bunn, S.E. (2006) Fish assemblages of an Australian dryland river: abundance, assemblage structure and recruitment patterns in the Warrego River, Murray-Darling Basin. *Marine and Freshwater Research* 57: 619-633.

Balcombe, S.R. and Arthington, A.H. (2009) Temporal changes in fish abundance in response to hydrological variability in a dryland floodplain river. *Marine and Freshwater Research* 60: 146-159.

Barbour M.T., Gerritsen J., Snyder B.D. and Stribling J.B. (1999) *Rapid bioassessment protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrates and fish*. Second edition. EPA 841-B-99-022. Office of Water, US Environmental Protection Agency, Washington DC.

Baron J.S., Poff N.L., Angermeier P.L., Dahm C.N., Gleick P.H., Hairston N.G., Jackson R.B., Johnston C.A., Richter B.D. and Steinman, A.D. (2002) Meeting ecological and societal needs for freshwater. *Ecological Applications* 12: 1247-1260.

Bastian, M., Pearson, R.G. and Boyero, L. (2008) Effects of diversity loss on ecosystem function across trophic levels and ecosystems: A test in a detritus-based tropical food web. *Austral Ecology* 33: 301-306.

Biggs, H.C. and Rogers, K.H. (2003) An adaptive system to link science, monitoring, and management in practice. In: J.T. du Toit, K.H. Rogers and H.C. Biggs (eds.) *The Kruger Experience. Ecology and Management of Savanna Heterogeneity*, pp: 59-80, Island Press, Washington DC.

Brizga S.O., Arthington, A.H, Pusey, B.J., Kennard, M.J., Mackay, S.J., Werren, G.L., Craigie, N.M. and Choy, S.J. (2002) Benchmarking, a 'top-down' methodology for assessing environmental flows in Australian rivers. In: *Environmental Flows for River Systems. An International Working Conference on Assessment and Implementation, incorporating the 4th International Ecohydraulics Symposium*. Conference Proceedings. Southern Waters, Cape Town, South Africa, 32 pp.

Brodie, J., Pearson, R., Lewis, S., Bainbridge, Z., Waterhouse, J. and Prange, J. (2009) *Water Quality Research: Baseline Synthesis and Year 1 Summary.* Report to the Marine and Tropical Sciences Research Facility. Reef and Rainforest Research Centre Limited, Cairns (124pp.) (http://www.rrrc.org.au/publications/wq_synthesis.html)

Bunn, S.E., Abal, E.G., Smith, M.J., Choy, S.C., Fellows, C.S., Harch, B.D., Kennard, M.J . and Sheldon, F. (2010) Integration of science and monitoring of river ecosystem health to guide investments in catchment protection and rehabilitation *Freshwater Biology* 55 (Suppl. 1): 223-240.

Bunn, S.E. and Davies, P.M. (2000) Biological processes in running waters and their implications for the assessment of ecological integrity. *Hydrobiologia* 422/423: 61-70.

Bunn, S.E., Davies, P.M., Kellaway, D.M. and Prosser, I.P. (1998) Influence of invasive macrophytes on channel morphology and hydrology in an open tropical lowland stream, and potential control by riparian shading. *Freshwater Biology* 39: 171-178.

Bunn, S.E., Davies P.M. and Mosisch, T.D. (1999) Ecosystem measures of river health and their response to riparian and catchment degradation. *Freshwater Biology* 41: 333-345.

Bunn, S.E. and Arthington, A.H. (2002) Basic principles and consequences of altered hydrological regimes for aquatic biodiversity. *Environmental Management* 30: 492-507.

Butler B. and Burrows, D.W. (2007) *Dissolved oxygen guidelines for freshwater habitats of northern Australia.* Version 1.0. ACTFR Report No. 07/32. Australian Centre for Tropical Freshwater Research (ACTFR), James Cook University, Townsville (http://www-public.jcu.edu.au/public/groups/everyone/documents/technical_report/jcudev_017880.pdf)

Butler B., Burrows, D.W. and Pearson, R.G. (2007) Providing regional NRM with improved aquatic health risk assessment and monitoring tools: the nationally significant indicator – dissolved oxygen. ACTFR Report No. 07/31. Australian Centre for Tropical Freshwater Research (ACTFR), James Cook University, Townsville (http://www-public.jcu.edu.au/public/groups/everyone/documents/technical_report/jcudev_017879.pdf)

Cairns J. (1995) Ecological integrity of aquatic systems. *Regulated Rivers: Research and Management* 11: 313-323.

Cheshire, K., Boyero, L. and Pearson, R.G. (2005) Food webs in tropical Australian streams: shredders are not scarce. *Freshwater Biology* 50: 748-769.

Chessman, B.C. (1995) Rapid assessment of rivers using macroinvertebrates. A procedure based on habitat specific sampling, family level identification and a biotic index. *Australian Journal of Ecology* 20: 122-129.

Clarke, K.R. (1993) Non-parametric multivariate analyses of changes in community structure. *Australian Journal of Ecology* 18: 117-143.

Clarke, K.R. and Gorley, R.N. (2006) Primer v6: User Manual/Tutorial. Primer-E Ltd., Plymouth.

Clayton, P.D and Pearson, R.G. (1996) Instream effects of dunder application to canefields in the Sarina region, Queensland: continued monitoring. ACTFR Report No. 96/09. Australian Centre for Tropical Freshwater Research (ACTFR), James Cook University, Townsville (http://www-public.jcu.edu.au/public/groups/everyone/documents/staff_publications/jcuprd_057179.pdf)

Connolly, N.M. and Pearson, R.G. (2007) The effect of fine sedimentation on tropical stream macroinvertebrate assemblages: a comparison using flow-through artificial stream channels and recirculating mesocosms. *Hydrobiologia* 592: 423-438.

Connolly, N.M., Crossland, M.R. and Pearson, R.G. (2004) Effect of low dissolved oxygen on survival, emergence and drift in tropical stream macroinvertebrate communities. *Journal of the North American Benthological Society* 23: 251-270.

Connolly, N.M., Pearson, B., Loong, D., Maughan, M. and Pearson, R.G. (2007a) Hydrology, Geomorphology and Water Quality of Four Wet Tropics Streams with Contrasting Land-use Management. In: A.H. Arthington and R.G. Pearson (eds.) *Biological Indicators of Ecosystem health in Wet Tropics Streams. Final Report, Task 3, Catchment to Reef Research Program.* Cooperative Research Centre for Tropical Rainforest Ecology and Management (Rainforest CRC) and Cooperative Research Centre for the Great Barrier Reef World Heritage Area (CRC Reef), pp. 15-79 (http://www.rrrc.org.au/catchment-to-reef/downloads/C2R-Arthington-A-et-al-2007-Biological-Indicators-Wet-Tropics-Streams.pdf)

Connolly, N.M., Pearson, B. and Pearson, R.G. (2007b) Macroinvertebrates as Indicators of Ecosystem Health in Wet Tropics Streams. In: A.H. Arthington and R.G. Pearson (eds.) *Biological Indicators of Ecosystem health in Wet Tropics Streams. Final Report, Task 3, Catchment to Reef Research Program.* Cooperative Research Centre for Tropical Rainforest Ecology and Management (Rainforest CRC) and Cooperative Research Centre for the Great Barrier Reef World Heritage Area (CRC Reef), pp: 129-176 (http://www.rrrc.org.au/catchment-to-reef/downloads/C2R-Arthington-A-et-al-2007-Biological-Indicators-Wet-Tropics-Streams.pdf)

Costanza, R., d'Argo, R., de Groot, R., Farber, S., Grass, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P. and van den Belt, M. (1997) The value of the world's ecosystem services and natural capital. *Nature* 387: 253-260.

Davies P.E. (2000) Development of a national river bioassessment system (AUSRIVAS) in Australia. In: J.F. Wright, D.W. Sutcliffe and M.T. Furse (eds.) *Assessing the Biological Quality of Fresh Waters: RIVPACS and Other Techniques*, pp. 13-124. Freshwater Biological Association, Ambleside.

DERM (2009) Queensland Wetland data – Wetland system layer, Version 2.0. Queensland Department of Environment and Resource Management, Brisbane.

Downes, B.J., Barmuta, L.A., Fairweather, P.G., Faith, D.P., Keough, M.J., Lake, P.S., Mapstone, B.D. and Quinn, G.P. (2002) *Monitoring Ecological Impacts. Concepts and Practice in Flowing Waters.* Cambridge University Press, UK.

Dudgeon, D., Arthington, A.H., Gessner, M.O., Kawabata, Z.I., Knowler, D.J., Lévêque, C., Naiman, R.J., Prieur-Richard, A.H., Soto, D., Stiassny, M.L.J. and Sullivan, C.A. (2006) Freshwater biodiversity: importance, threats, status and conservation challenges. *Biological Reviews* 81: 163-182.

EPA (2005) Wetland Mapping and Classification Methodology – Overall Framework – A method to provide baseline mapping and classification for wetlands in Queensland, Version 1.2. Queensland Environment Protection Agency (EPA), Brisbane (ISBN 0-97573-44-6).

Fairweather, P.G. (1999) State of the environment indicators of 'river health': exploring the metaphor. *Freshwater Biology* 41: 211-220.

Fausch, K.D., Lyons, J., Karr, J.R. and Angermeier, P.L. (1990) Fish communities as indicators ofenvironmental degradation. *American Fisheries Society Symposium* 8: 123-144.

Fellows, C.S., Clapcott, J.E., Udy, J.W., Bunn, S.E., Harch, B.D., Smith, M.J. and Davies, P.M. (2006) Benthic metabolism as an indicator of stream ecosystem health. *Hydrobiologia* 572: 71-87.

Furnas, M. (2003) *Catchments and Corals: terrestrial runoff to the Great Barrier Reef*. Australian Institute of Marine Science, Townsville, 334 pp.

Gehrke, P. and Sheaves, M. (2006) *Research priorities to sustain coastal fisheries resources in the Great Barrier Reef region: A scoping study for the Tully-Murray catchment.* CSIRO Water for a Healthy Country National Research Flagship, Canberra, 56 pp.

Harris, J.H. (1995) The use of fish in ecological assessments. *Australian Journal of Ecology* 20: 65-80.

Harris J.H. and Gehrke P.C. (1994) Development of predictive models linking fish population recruitment with streamflow. In: D.A. Hancock (ed.) *Population Dynamics for Fisheries Management*. Australian Society for Fish Biology Workshop Proceedings, Perth, 24-25 August 1993, pp. 195-197, Australian Society for Fish Biology, Sydney.

Hogan, A. and Graham, P. (1994) *Tully-Murray floodplain fish distributions and fish habitat*. Report to the consultants, Tully-Murray Sugar Industry Infrastructure Package. Queensland Department of Primary Industries, Walkamin, 11 pp.

Hortle, K.R. and Pearson, R.G. (1990) The fauna of the Annan River, north Queensland, with reference to the effects of tin mining. I. Fishes. *Australian Journal of Marine and Freshwater Research* 41: 677-694.

Humphries, P., King, A.J. and Koehn, J.D. (1999) Fish, flows and floodplains, links between freshwater fishes and their environment in the Murray-Darling river system, Australia. *Environmental Biology of Fishes* 56: 129-151.

Jeffrey, S.W. and Humphrey, G.F. (1975) New spectrophotometric equations for determining chlorophylls a, b, c1 and c2 in higher plants, algae and natural phytoplankton. *Biochemie und Physiologie der Pflanzen* 167: 191-194.

Johnson, A.K.L., Ebert, S.P. and Murray, A.E. (1997) Spatial and temporal distribution of wetland and riparian zones and opportunities for their management in catchments adjacent to the Great Barrier Reef Marine Park. *Proceedings GBRMPA Cross-Sectoral Workshop on Wetlands and Water Quality*. Great Barrier Reef Marine Park Authority (GBRMPA), Townsville, pp 82-101.

Junk, W.J. and Wantzen, K.M. (2004) The flood pulse concept: new aspects, approaches and applications – an update. In: R. Welcomme and T. Petr (eds.) *Proceedings of the Second International Symposium on the Management of Large Rivers for Fisheries Volume II*, pp. 117-140. FAO Regional Office Asia and the Pacific, Bangkok, RAP Publication 2004/17.

Junk, W.J., Bayley, P.B. and Sparks, R.E. (1989) The flood pulse concept in river-floodplain systems. *Canadian Special Publication of Fisheries and Aquatic Sciences* 106: 110-127.

Karim, F., Wallace, J., Kinsey-Henderson, A., Hawdon, A., Keen, R., Arthington, A.H., Godfrey, P. and Pearson, R.G. (2010a) *Wetlands and floodplains: connectivity and hydroecological function. Part II – Quantification of overbank and channelised wetland connectivity in the Tully-Murray floodplain*. Final Project Report prepared for the Marine and Tropical Sciences Research Facility (MTSRF). Published by the Reef & Rainforest Research Centre Ltd., Cairns (45pp.).

Karim, F., Kinsey-Henderson, A., Wallace, J.S., Arthington, A.H. and Pearson, R.G. (2010b) *Hydrodynamic modelling of flood induced wetland connectivity in a tropical floodplain in north Queensland, Australia* (In review).

Karim, F.M. and Wallace, J.S. (2008) *Assessment of Sediment and Nutrient Transport across the Tully-Murray Floodplain using the SedNet Model.* Report to the Marine and Tropical Sciences Research Facility. CSIRO Land and Water Science Report 59/08, 18 pp.

Karr, J.R. (1981) Assessments of biotic integrity using fish communities. *Fisheries (Bethesda)* 6: 21-27.

Karr, J.R. (1996) Ecological integrity and ecological health are not the same. In: Schulze P. (ed.) *Engineering Within Ecological Constraints*. National Academy Press, Washington DC, pp. 97-109.

Karr, J.R. and Chu, E.W. (1999) *Restoring Life in Running Waters: Better Biological Monitoring*. Island Press, Washington, DC.

Kelso, K.E. and Rutherford, D.A. (1996) Collection, preservation and identification of fish eggs and larvae. In: B.R. Murphy and D.W. Willis (eds.) *Fisheries Techniques*, pp. 255-302, American Fisheries Society.

Kennard, M.J., Arthington, A.H., Pusey, B.J. and Harch, B.D. (2005) Are alien fish a reliable indicator of river health? *Freshwater Biology* 50: 174-193.

Kennard, M.J., Harch, B.D., Pusey, B.J. and Arthington, A.H. (2006a) Accurately defining the reference condition for summary biotic metrics: a comparison of four approaches. *Hydrobiologia* 572: 151-170.

Kennard, M.J., Pusey, B.J., Arthington, A.H., Harch, B.D. and Mackay, S. (2006b) Development and application of a predictive model of freshwater fish assemblage composition to evaluate river health in eastern Australia. *Hydrobiologia* 572: 33-57.

Kroon, F.J. (2009) Introduction. An integrated research approach to improving water quality in the Great Barrier Reef Region. *Marine and Freshwater Research* 60 (Special Issue: Catchment Management and Coastal Water Quality of the Great Barrier Reef): i-iii.

Lawson, T., Kroon, F., Russell, D.J. and Thuesen, P. (2010) *Audit and prioritisation of physical barriers to fish passage in the Wet Tropics region.* MTSRF Project 2.6.2 Milestone Report to the Marine and Tropical Sciences Research Facility (MTSRF), 14 pp.

Leis, J.M. and Carson-Ewart, B.M. (2000) *Larvae of Indo-Pacific Coastal Fishes.* Brill, Leiden, The Netherlands, 850 pp.

Leonard K. (2009) *Invertebrates as indicators of ecosystem health in streams of the Mackay-Whitsunday region*. MAppSci Dissertation submitted to the School of Marine and Tropical Biology, James Cook University.

Mackay, S.J., James, C. and Arthington, A.H. (2010) Macrophytes as indicators of stream condition in the wet tropics region, Northern Queensland, Australia. *Ecological Indicators* 10: 330-340.

McCune, B. and Mefford, M. J. (2006) PC-ORD. Multivariate Analysis of Ecological Data. Version 5.10. MjM Software, Gleneden Beach, Oregon, USA.

Millennium Ecosystem Assessment (2005) *Millennium Ecosystem Assessment Synthesis Report.* Island Press, Washington DC.

Naiman, R.J., Latterell, J.J., Pettit, N.E. and Olden, J.D. (2008) Flow variability and the vitality of river systems. *Comptes Rendus Geoscience* 340: 629-643.

Neira, F.J., Miskiewicz, A.G. and Trnski, T. (1998) *Larvae of Temperate Australian Fishes*. Laboratory Guide for Larval Fish Identification. University of Western Australia Press, Nedlands, Australia, 474 pp.

Norris, R.H. and Hawkins, C.P. (2000) Monitoring river health. *Hydrobiologia* 435: 5-17.

Økelsrud, A. and Pearson, R.G. (2007) Acute and post-exposure effects of ammonia toxicity to juvenile barramundi (*Lates calcarifer* [Bloch]). *Archives of Environmental Contamination and Toxicology* 53: 624-631.

Overton, I. C. (2005) Modelling floodplain inundation on a regulated river: Integrating GIS, remote sensing and hydrological models. *River Research and Applications* 21: 991-1001.

Paterson, A.W. and Whitfield, A.K. (2000) Do shallow-water habitats function as refugia for juvenile fishes? *Estuarine, Coastal and Shelf Science* 51: 359-364.

Pearson, R.G. (1991) *Ecology of the Burdekin River: Final Report*. AWRAC Project 85/86. ACTFR Report No. 91/1. Australian Centre for Tropical Freshwater Research (ACTFR), James Cook University, Townsville.

Pearson, R.G. and Connolly, N.M. (2000) Nutrient enhancement, food quality and community dynamics in a tropical rainforest stream. *Freshwater Biology* 43: 31-42.

Pearson, R.G. and Penridge L.K. (1979) An ecological survey of selected rivers in Queensland. James Cook University/SRI, 234pp.

Pearson, R.G. and Penridge, L.K. (1987) The effects of pollution by organic sugar mill effluent on the macro-invertebrates of a stream in tropical Queensland, Australia. *Journal of Environmental Management* 24: 205-215.

Pearson, R.G. and Stork, N.E. (2007) Catchment to Reef: Water Quality and Ecosystem Health in Tropical Streams. Chapter 45. In: N.E. Stork and S. Turton (eds.) *Living in a Dynamic Tropical Forest Landscape.* Blackwell Publishing, Carlton, Victoria, pp. 557-576.

Pearson, R.G., Arthington, A.H., Wallace, J., Blanchette, M. and Godfrey, P.C. (2010) *Freshwater Indicators Workshop Report.* Report on MTSRF Projects 3.7.3 and 3.7.4 Activities to the Marine and Tropical Sciences Research Facility (MTSRF), Cairns, May 2010 (18 pp.).

Pearson, R.G., Benson, L.J. and Smith, R.E.W. (1986) Diversity and abundance of the fauna in Yuccabine Creek, a tropical rainforest stream. In: P. de Deckker and W.D. Williams (eds.) *Limnology in Australia*, pp. 329-342, CSIRO, Melbourne.

Pearson, R.G., Crossland, M., Butler, B. and Manwaring, S (2003) *Effects of cane-field drainage on the ecology of tropical waterways*. Three volumes. Report on SRDC Projects JCU016 and JCU024. Australian Centre for Tropical Freshwater Research (ACTFR) Report No. 3/04-1-3. James Cook University, Townsville.

Perna, C. and Burrows, D.W. (2005) Improved dissolved oxygen status following removal of exotic weed mats in important fish habitat lagoons of the tropical Burdekin River floodplain, Australia. *Marine Pollution Bulletin* 51: 138-148.

Poff, N.L., Richter, B.D., Arthington, A.H., Bunn, S.E., Naiman, R.J., Kendy, E., Acreman, M., Apse, C., Bledsoe, B.P., Freeman, M.C., Henriksen, J., Jacobson, R.B., Kennen, J.G., Merritt, D.M., O'Keeffe, J.H., Olden, J.D., Rogers, K., Tharme, R.E. and Warne, A. (2010) The ecological limits of hydrologic alteration (ELOHA): a new framework for developing regional environmental flow standards. *Freshwater Biology* 55: 147-170.

Postel, S. and Richter, B. (2003) *Rivers for Life: Managing Water for People and Nature*. Island Press, Washington DC.

Preite, C. (2009) *Phytoplankton Dynamics of Riverine Waterholes in the Dry Tropics of North Queensland, Australia*. PhD thesis, James Cook University, Townsville, Australia.

Pusey, B.J. and A.H. Arthington (1996) Variability of flow regimes in the Burdekin River catchment. *Proceedings of the 23rd Hydrology and Water Resources Symposium*, Hobart, Tasmania: 213-219.

Pusey, B.J. and Kennard, M.J. (1996) Species richness and geographical variation in assemblage structure of the freshwater fish fauna of the wet tropics region of northern Queensland. *Marine and Freshwater Research* 47: 563-573.

Pusey, B.J. and Arthington, A.H. (2003) Importance of the riparian zone to the conservation and management of freshwater fish: a review. *Marine and Freshwater Research* 54: 1-16.

Pusey, B.J., Arthington, A.H. and Read, M.G. (1998) Freshwater fishes of the Burdekin River, Australia: biogeography, history and spatial variation in assemblage structure. *Environmental Biology of Fishes* 53: 303-318.

Pusey, B.J., Kennard, M.J. and Arthington, A.H. (2004) *Freshwater Fishes of North-Eastern Australia.* CSIRO Publishing, Melbourne.

Pusey, B.J., Kennard, M.J. and Arthington, A.H. (2007a) Origins and maintenance of freshwater fish biodiversity in the wet tropics region. Chapter 9. In: N.E. Stork and S. Turton (eds.) *Living in a Dynamic Tropical Forest Landscape*, pp: 150-160. Blackwell Publishing, UK.

Pusey, B.J., Kennard, M.J. and Arthington, A.H. (2007b) Freshwater Fish as Indicators of Ecosystem Health in Wet Tropics Streams. In: A.H. Arthington and R.G. Pearson (eds.) *Biological Indicators of Ecosystem health in Wet Tropics Streams*. *Final Report, Task 3, Catchment to Reef Research Program.* Cooperative Research Centre for Tropical Rainforest Ecology & Management (Rainforest CRC) and Cooperative Research Centre for the Great Barrier Reef World Heritage Area (CRC Reef), pp: 177-233, ISBN: 9780864437921.

Queensland Government (2003) *State of the Environment Report, 2003*. Queensland Government, Brisbane.

Rapport, D.J., Costanza, R. and McMichael, A.J. (1998) Assessing ecosystem health. *Trends in Ecology and Evolution* 13: 397-402.

Rayner, T.S., Pusey, B.J. and Pearson, R.G. (2008) Seasonal flooding, instream habitat structure and fish assemblages in the Mulgrave River, north-east Queensland: towards a new conceptual framework for understanding fish-habitat dynamics in small tropical rivers. *Marine and Freshwater Research* 59: 97-116.

Richards, C., Haro, R.J., Johnson, L.B. and Host. G.E. (1997) Catchment- and reach-scale properties as indicators of macroinvertebrate species traits. *Freshwater Biology* 37: 219-230.

Robins, J.B., Halliday, I.A., Staunton-Smith, J., Mayer, D.G. and Sellin, M.J. (2005) Freshwater-flow requirements of estuarine fisheries in tropical Australia: a review of the state of knowledge and application of a suggested approach. *Marine and Freshwater Research* 56: 343-360.

Rodriguez, M.A., Winemiller, K.O., Lewis, W.M. and Taphorn Baechle, D.C. (2007) The freshwater habitats, fishes, and fisheries of the Orinoco River basin. *Aquatic Ecosystem Health and Management* 10: 140-152.

Rosser, Z. and Pearson, R.G. (1995) Responses of rock fauna to physical disturbance in two Australian tropical rainforest streams. *Journal of the North American Benthological Society* 14: 183-196.

Sala, O.E., Meyerson, L.A. and Parmesan, C. (eds.) (2008) *Biodiversity Change and Human Health.* Island Press, Washington, DC.

Smith, B., Burrows, D.W., Veitch, V., Tait, J. and Hansen, D. (2007) Final Report, GBR Coastal Wetlands Protection Programme, 2005-2007 Pilot Programme.

Smith, M.J., Kay, W.R., Edward, D.H.D., Papas, P.J., Richardson, K.St J., Simpson, J.C., Pinder, A.M., Cale, D.J., Horwitz, P.H.J., Davis, J.A., Yung, F.H., Norris, R.H. and Halse, S.A. (1999) AusRivAS: using macroinvertebrates to assess ecological condition of rivers in Western Australia. *Freshwater Biology* 41: 269-282.

Smith, M.J. and Storey, A.W. (2001) *Design and Implementation of a Baseline Monitoring (DIBM3) Phase 1 Final Report.* Report to the South East Queensland Regional Water Quality Management Strategy (SEQRWQMS), Brisbane, Australia, November 2000.

Smith, R.E.W. and Pearson, R.G. (1987) The macro-invertebrate communities of temporary pools in an intermittent stream in tropical Queensland. *Hydrobiologia* 150: 45-61.

Southwood, T.R.E. (1977) Habitat, the templet for ecological strategies? *J. Anim. Ecol*. 46: 337-365.

Sunderam, R.I.M., Thompson, G.B., Chapman, J.C. and Cheng, D.M.H. (1994) Acute and chronic toxicity of endosulfan to two Australian cladocerans and their applicability in deriving water quality criteria. *Arch. Env. Contamination and Toxicology* 27: 541-545.

Suter, G.W. (2001) Applicability of indicator monitoring to ecological risk assessment. *Ecological Indicators* 1: 101-112.

Sydes, T. (2009) *Cross Regional Hymenachne Management Strategy. Johnstone, Tully-Murray, Lower Herbert and Black River Catchments*. Far North Queensland Regional Organisation of Councils (FNQROC), Cairns, Queensland.

Tait, J. (undated) *Lowland habitat mapping and management recommendations. Tully-Murray catchments.* Final Draft. An ICM initiative of the Queensland Department of Primary Industries.

Tockner, K., Bunn, S., Gordon, C., Naiman, R.J., Quinn, G.P. and Stanford, J.A. (2008) Flood plains: critically threatened ecosystems. In: N.V.C. Polunin (ed.) *Aquatic Ecosystems*, pp. 45-61. Cambridge University Press, Cambridge, UK.

Townsend, C.R. and Hildrew, A.G. (1994) Species traits in relation to a habitat templet for river systems. *Freshwater Biology* 31: 265-275.

Usseglio-Polatera, P., Bournaud, M., Richoux, P. and Tachet, H. (2000) Biomonitoring through biological traits of benthic macroinvertebrates: How to use species trait databases? *Hydrobiologia* 422: 153-162.

Vallance, T. and Hogan, A. (2004) *Final Report: Riversdale-Murray Valley SIIP fish biodiversity enhancement study*. Queensland Department of Primary Industries, 28 pp.

Vannote, R.L., Minshall, G.W., Cummins, K.W., Sedell, K.W. and Cushing, C.E. (1980) The river continuum concept. *Canadian Journal of Fisheries and Aquatic Sciences* 37: 130-137.

Veitch, V. and Sawynok, B. (2005) *Freshwater Wetlands and Fish*. Great Barrier Reef Marine Park Authority and Sunfish Queensland Inc. Report No. SQ200401, 136 pp.

Wallace, J.S., Arthington, A.H. and Pearson, R.G. (2007) *Hydro-ecological modeling in coastal catchments: connectivity and hydro-ecological function.* Report from the MTSRF Workshop held at the CSIRO Davies Laboratory, Townsville, 19-20 April 2007, CSIRO and James Cook University, 45pp.

Wallace, J.S., Hawdon, A., Keen, R. Stewart, L. and Kemei, J. (2009b) Flood water quality and marine sediment and nutrient loads from the Tully and Murray catchments in north Queensland, Australia. *Marine and Freshwater Research* 60: 1123-1131.

Wallace, J., Hawdon, A., Keen, R., Karim, F., Stewart, L. and Kemei, J. (2010) *Wetlands and floodplains: connectivity and hydroecological function. Part I – The role of overbank floods in transporting sediments and nutrients to the Great Barrier Reef lagoon*. Final Project Report prepared for the Marine and Tropical Sciences Research Facility (MTSRF). Published by the Reef & Rainforest Research Centre Ltd., Cairns (27pp.).

Watzin, M.C. and McIntosh, A.W. (1999) Aquatic ecosystems in agricultural landscapes: a review of ecological indicators and achievable ecological outcomes. *J. Soil Water Conservation* 54: 636-44.

Welcomme, R.L., Bene, C., Brown, C.A., Arthington, A.H., Dugan, P., King, J.M. and Sugunan, V. (2006) Predicting the water requirements of river fisheries, pp. 123-154, In: J.T.A. Verhoeven, B. Beltman, R. Bobbink and D.F. Whigham (eds.) *Wetlands and Natural Resource Management. Ecological Studies, Vol 90*, Springer-Verlag, Berlin.

Werren, G. and Arthington, A.H. (2002) The assessment of riparian vegetation as an indicator of stream condition, with particular emphasis on the rapid assessment of flow-related impacts. In: J. Playford, A Shapcott and A. Franks (eds.) *Landscape Health of Queensland*, pp 194-222, Royal Society of Queensland, Brisbane.

Winemiller, K.O., Tarim, S., Shormann, D., and Cotner, J.B. (2000) Fish assemblage structure in relation to environmental variation among Brazos River Oxbow lakes. *Transactions of the American Fisheries Society* 129: 451-468.

Wright, J.F. (1995). Development and use of a system for predicting the macroinvertebrate fauna in flowing waters. *Australian Journal of Ecology* 20: 181-197.