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Hydro-ecological relationships and thresholds to inform environmental flow management

AH Arthington, CS James, SJ Mackay, R Rolls, D Sternberg, A Barnes

RESEARCH PARTNER

Griffith UNIVERSITY

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Level 16, 333 Ann St
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Email: info@watercentre.org

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Abbreviations & acronyms

AICc	Akaikes Information Criterion
ANCOLD	Australian National Committee on Large Dams
ANOSIM	Analysis of Similarities
ARI	Average Recurrence Interval
ARF	Amphibious Fluctuation-Responders – floating/stranded
ARP	Amphibious Fluctuation-Responders – morphologically plastic
BACI	Before and after comparison of control and impact
BFI	Baseflow index
CCA	Canonical correspondence analysis
CPUE	Catch per unit effort
CEWH	Commonwealth Environmental Water-Holder
CONSTAN	Constancy of mean daily discharge
CRC	Cooperative Research Centre
CRCFE	Cooperative Research Centre for Freshwater Ecology
CV	Coefficient of variation
DEM	Digital elevation map
DERM	Department of Environment and Resource Management
D_INTER	Density of intermediate successful stages per ha
DISTLM	Distance-based linear modelling
D_LATE	Density of late successional stages per ha
DNRM	Department of Natural Resources and Mines
DPI	Department of Primary Industries and Fisheries
DRIFT	Downstream Response to Imposed Flow Transformation
EHMP	Ecosystem Health Monitoring Program
ELOHA	Ecological Limits of Hydrologic Alteration
GLM	Generalized linear models
GLS	Generalized least squares
HFC	Historic flow class
HSDis	High spell discharge
HSDur	High spell duration OR Duration of high pulses within each year
HSNum	Number of floods greater than the median flow OR Number of high pulses within each year
IDW	Inverse Distance Weighted
IHA	Indicators of Hydrologic Alteration
IQQM	Integrated Quantity Quality Model
IU	Intensive uses
IWC	International Water Centre
LSDis	Low spell discharge
LSDur	Low spell duration / Duration of low pulses within each year
LSNum	Low spell number / Number of low pulses within each year
LWRRDC	Land and Water Resources Research and Development Corporation
M	Number of clusters
MDB	Murray–Darling Basin
MDBC	Murray–Darling Basin Commission
MDBA	Murray–Darling Basin Authority
MDBF	Mean daily baseflow
Ml	Megalitres (1 000 000 litres)
nMDS	Non-metric multidimensional scaling
NSW	New South Wales
NWC	National Water Commission
NWI	National Water Initiative
OOB	Out-of-bag
PCA	Principal components analysis
PDA	Production from dryland agriculture and plantations
PERMANOVA	Permutational Multivariate Analysis of Variance
PIA	Production from irrigated agriculture and plantations
PLS	Partial least-squares projection to latent structures
PNE	Production from relatively natural environments
PREDICT	Predictability of mean daily discharge
QLUMP	Queensland Land Use Mapping Program
RAP	River Analysis Package
RDA	Redundancy Analysis
RFC	Reference flow class
RNWS	Raising National Water Standards
ROP	Resource Operations Plan
RVA	Range of Variability Approach
SEASON	Seasonality of mean daily discharge
SEQ	South-east Queensland
SEQWater	Queensland Bulk Water Supply Authority (trading as SEQwater)
Supplemented	In Queensland, water supply from releasts of water stored in infratstructure. Equivalent to a regulated water supply (NWI). (from National Water Comission 2011, National Water Planning Report Card 2011, NWC Canberra)
TNC	The Nature Conservancy
TRaCK	Tropical Rivers and Coastal Knowledge
Tukey's HSD	Tukey's Honestly Significant Difference (multiple comparisaon procedure)
Unsupplemented	In Queensland, water supply not involving releases of water stored in infrastructure. Equivalent to an unregulated water supply (NWI). (from National Water Commission 2011, National Water Planning Report Card 2011, NWC Canberra)
VIF	Variance inflation factor
WRP	Water Resource Plan

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Griffith University conducts research in accordance with the National Statement on Ethical Conduct in Research Involving Humans. Surveys involving land owners were conducted under human research ethics approval GU Ref No: ENV/36/08/HREC. All fish research work was carried out under Queensland Fisheries Permit PRM00157K and Griffith University Animal Ethics Approval No. ENV/21/08/AEC.

Executive summary

The *Waterlines* associated with this *Science Report* provides the key findings of the project *Hydro-ecological relationships and thresholds to inform environmental flow management and river restoration*. It is the first study in Australia to explore the scientific and management implications of the ELOHA (Ecological Limits of Hydrologic Alteration) framework for regional environmental flow assessment. The ELOHA trial was funded by the National Water Commission through the Raising National Water Standards (RNWS) program, hosted and managed by the International WaterCentre (IWC) and undertaken by the Australian Rivers Institute, Griffith University in Brisbane, Queensland.

The ELOHA framework is a new approach to informing the regional development of environmental flow guidelines that explicitly takes into account spatial variation in flow regimes as well as the potential influence of other environmental variables such as climate and land-use. In its entirety, the ELOHA framework includes both a biophysical and a social module. Using south-east Queensland (SEQ) as a study region, this project aimed to test the four central concepts of the ELOHA framework's biophysical module which can be summarised as:

1. Rivers of a chosen region can be grouped into distinctive flow regime classes on the basis of ecologically relevant flow metrics, such as measures of magnitude, duration, timing, frequency and variability of flows
2. Ecological characteristics of rivers within each flow regime class will be relatively similar compared to those of other classes. Therefore these flow regime classes represent distinct management units or groups of streams that can be managed in similar ways in terms of environmental flows
3. Rivers within each flow regime class that are 'regulated' (or supplemented) in the same way by dams and other infrastructure will show similar ecological responses to flow regime change
4. Increasing degrees of flow regime change will have increasing impacts on ecological response variables

In addition to testing the key concepts that underpin the ELOHA framework, this project analysed the full field database to provide new knowledge of relevance to the management of flow regimes and river ecosystems of SEQ. This includes:

i) information and guidelines on the relative influence of hydrology and other pressures, for example land-use, on river ecosystems in SEQ and advice on how to manage particular combinations of flow alteration and other pressures so as to achieve healthier rivers

ii) information about ecological responses to hydrologic alteration and quantitative relationships to inform the development of environmental flow requirements for the protection and/or restoration of selected ecological assets in rivers of different hydrologic character.

Guided by the ELOHA framework, the project comprised three major components:

i) an analysis of flow regimes of SEQ river basins including an assessment of hydrologic alteration

ii) a synthesis of existing knowledge of ecological responses to patterns of hydrologic variability and alteration across the study region

iii) a field research program to identify the impacts of hydrologic alteration within the study region on selected ecological assets - riparian vegetation, aquatic vegetation and fish. In doing so, the project aimed to identify linear relationships (or thresholds) of ecological response of these assets to hydrologic alteration. Responses of these assets to a limited suite of flow variables that collectively influence the condition or ecological 'health' of sites on each river system, were also determined.

Key findings

The key findings of this ELOHA trial of relevance to the management of flow regimes and river ecosystems of SEQ can be synthesised into the following major points:

1. **Unregulated and regulated flow regimes vary across SEQ mainly with respect to discharge magnitude:** A Reference classification of 'natural' flow regimes in SEQ has been developed in this project using modelled pre-development flow data derived from an "integrated quantity quality" (IQQM) model. Six Reference flow classes (RFCs) have been identified that vary primarily with respect to the magnitude of flows and, to a lesser degree, flow variability. All RFCs include localities from several catchments but low rainfall western and north-western localities tend to group together in one reference flow class (RFC 4) while coastal, eastern sites typically fall into another reference flow class (RFC 5) (see Table 2.4).

   An Historic flow regime classification has also been developed in this project using actual stream gauge data. Five Historic flow classes (HFCs) have been identified which also vary predominantly in relation to the magnitude of flows and in how these flows have been changed by the presence of dams (see Table 2.6).

2. **Flow regime alteration is widespread across SEQ but the overall degree of change is relatively minor:** All streams and rivers in the region exhibit some degree of flow regime alteration due to dams, weirs or land-use. The greatest change is generally apparent downstream of dams for Nerang River, Reynolds Creek, Yabba Creek, Lockyer Creek, Brisbane River and Burnett Creek. Some dams such as Six Mile Creek, have had a minor effect on the overall character of the downstream flow regime. Furthermore, high levels of flow regime alteration are also evident in streams without dams, for example, Running Creek, Mudgeeraba Creek and the South Pine River, possibly due to the impact of extensive land-use change associated with agriculture and urbanisation.

   The degree of overall flow regime alteration across the study area is relatively minor when expressed by a summary metric (Gower metric) of dissimilarity. The Gower metric is a statistical term calculated to express overall levels of flow regime alteration and has a potential range from 0 (indicating no change) to 1 (indicating complete change across all the hydrologic variables considered in the calculation). A maximum value of 0.25 was determined for the study region(below Hinze Dam on the Nerang River), which suggests a relatively low degree of overall hydrologic alteration based on the project's analyses of 35 different flow metrics.

   In some cases, particular characteristics of flow regimes in the study area have changed markedly from Reference conditions, especially those metrics describing the magnitude of flows, including the duration of periods of low flow (which increased), mean rates of rise and fall of flows (which also increased or became more variable), mean monthly discharge (which decreased) and annual minimum flows (which also decreased). Some aspects of flow frequency, duration, variability and seasonal timing have also changed from Reference conditions.

3. **Every dam in SEQ has altered downstream flow regimes in a different way**: The assessment of hydrologic alteration conducted in this project indicates that every dam in the study region has altered downstream discharge patterns in a different way. These alterations depend on the characteristics of each dam's flow class, location, storage capacity, water release strategies and downstream water extraction practices. Consequently, each dam in the study region has generated a unique downstream flow regime. This finding prevented confirmation of the ELOHA concept that rivers within each hydrologic class that are regulated in the same way by dams and other infrastructure, will show similar ecological responses to flow regime change.

4. **Flow regime alteration due to dams and other factors has had significant impacts on riparian and aquatic vegetation and fish in SEQ. These impacts vary between flow regime classes and downstream of particular dams:** The results of this study suggest that riparian vegetation of regulated sites in RFC 5 (e.g. sites on the Nerang River below Hinze Dam) has shifted in structure in response to flow regime alteration, and is now similar to that of low discharge sites in the Mary and Logan river catchments. Strongly regulated sites also have significantly lower riparian species density and basal area (ground covered by the bases of trees) per hectare than unregulated sites across all flow classes in the study region. Furthermore, densities of reeds, rushes and sedges are higher at regulated sites, probably as a result of reductions in high in-channel discharges and flood flows that would normally dislodge these plants.

   Aquatic vegetation structure differed between regulated and unregulated sites within two Reference flow regime classes: RFC 1, including Obi Obi Creek downstream of Baroon Pocket Dam, and RFC 5, which includes the Nerang River downstream of Hinze Dam. When the effects of local, within-site habitat variation are removed, total cover of submerged aquatic vegetation is higher in regulated sites than unregulated sites.

   Higher species richness of fish assemblages was found to be associated with regulated sites within the study region probably because low discharges below dams were elevated compared to their normally low levels during dry months. In one regulated site, Six Mile Creek, non-migratory fish species richness was almost double that of unregulated sites within the same Historic flow class despite relatively slight overall flow regime change. Densities of Duboulay's rainbowfish were also significantly higher in regulated sites compared with unregulated sites within RFC 2, while densities of Pacific blue-eye were significantly lower in regulated sites than in unregulated Reference sites in RFC 1.

5. **Ecologically important flow metrics in SEQ rivers range across a suite of hydrologic variables encompassing the magnitude, frequency, duration, timing and variability/ predictability of low to medium and high discharges:** Statistically significant hydro-ecological relationships have been quantified in this project. Relationships have been established between flow and each of the ecological assets explored – riparian vegetation, aquatic vegetation and fish. Many of these relationships have not been quantified previously for the study region.

   Variability in flows during the dry season was identified as a particularly important influence on riparian vegetation structure, probably because riparian vegetation is more reliant on stream discharge during the dry season when there is less rainfall and moisture levels of bank soils are lower. Furthermore, variation in flows at this time of year may result in frequent spells during which the streams cease to flow, resulting in a dropping of the local riparian groundwater table. These processes influence which riparian species can persist during dry periods. Bankfull discharge and duration were significantly related to the diversity of near-stream vegetation. High flows that reach the bankfull height along a stream channel, influence soil moisture levels, the vigour of riparian vegetation and its capacity for seed production, seedling growth rates and plant survival.

   Flow characteristics relating to the discharge required to mobilise the median size of stream substrate particles were identified as most significant for aquatic vegetation, as these high flows influence the periodic removal of plants. This normal process in an unregulated river with periods of natural high flows mobilises stream substrates and prevents the build-up of dominant aquatic plant species, some of which may be alien species. A more diverse and patchy plant assemblage tends to develop on suitable substrates. Diverse plant assemblages offer greater diversity of habitat for invertebrates and fish.

   Fish assemblage structure was most influenced by the occurrence and duration of zero and low discharges, daily hydrologic variability, high discharges (e.g. the number of floods greater than the median discharge) and seasonal patterns of mean monthly discharge. Duration of zero and low discharges and daily flow variability can influence fish habitat availability and fish survival. Some alien species are more tolerant of zero and low flows than native species. Seasonal patterns of monthly discharge and water temperature, drive seasonal patterns of fish breeding. Floods disturb stream substrates and aquatic vegetation and help to generate habitat diversity. They also provide hydrologic connectivity and movement pathways for fish throughout the channel network, allowing individuals at various life history stages, to access suitable habitat and food resources and to encounter reproductively active mates.

6. **The ecological importance of discharge patterns is situational and ecological responses to flow regime alteration therefore depend on climate and other catchment variables:** Considerable geological and climatic variation is present across the SEQ study area and the effects of environmental factors other than flow, especially climate, were apparent for all of the ecological assets examined in this study. With respect to riparian vegetation, significant climatic, catchment and land-use characteristics were identified including the coldest month mean temperature, catchment relief and upstream geological characteristics, as well as the effects of dryland agriculture and intensive land-uses. For aquatic vegetation, mean substrate particle size emerged as particularly significant with depth, bankfull shear stress, water quality, turbidity, riparian canopy cover as well as dryland agriculture having a significant influence. Fish assemblage patterns were associated with gradients in climatic factors (rainfall and temperature), catchment geology, channel morphology, stream habitat structure and hydrology.

Testing of ELOHA concepts

The key findings of this project with respect to testing the central concepts of the ELOHA framework can be summarised as follows:

1. **Rivers of a chosen region can be grouped into distinctive flow regime classes on the basis of ecologically relevant flow metrics, such as measures of magnitude, duration, timing, frequency and variability of flows:** Two flow regime classifications have been developed, a Reference flow classification describing modelled 'natural' conditions and an Historic flow classification describing the actual conditions measured at gauging stations. Flow regime classes within each classification are distinguished mainly with respect to aspects of flow magnitude, and to a lesser extent, flow variability. Flow alterations caused by dams have created a new Historic flow class (HFC1) with more variable seasonal flow peaks and elevated low flows in normally dry months.

2. **Ecological characteristics of rivers within each flow regime class will be relatively similar compared to those of other classes. Therefore these flow regime classes represent distinct management units or groups of streams that can be managed in similar ways in terms of environmental flows:** The project found mixed support for the concept that ecological characteristics of rivers within each flow regime class will be relatively similar compared to those of other classes. Fish assemblage structure displayed the greatest correlation with flow regime class, and a range of differences in fish assemblage diversity, density and composition were apparent between Reference flow classes. Riparian vegetation structure was a relatively poor predictor of Reference flow regime class although significant differences were evident amongst a range of metrics describing bankfull vegetation structure across the classes of both the Reference and Historic flow classifications. No consistent, statistically significant differences in aquatic vegetation were detected amongst Historic flow regime classes, with the exception that amphibious plant species – those with some tolerance of exposure and dessication – were found to dominate sites in one Historic flow regime class characterised by high flow variability.

3. **Rivers within each flow regime class that are 'regulated' in the same way by dams and other infrastructure will show similar ecological responses to flow regime change**: Mixed support was found for this concept. Since no two dams in the study area have produced the same types of hydrologic change, ecological effects also vary among sites below dams across the study region. Nevertheless, significant differences in riparian vegetation were apparent between regulated and unregulated sites in RFC 5 (containing the Nerang River downstream of Hinze Dam) and for aquatic vegetation within two Reference flow classes: RFC 5 and RFC1 (containing Obi Obi Creek downstream of Baroon Pocket Dam).

   Fish assemblage structure also differed between regulated and unregulated sites within RFCs 1 and 2, particularly with respect to densities of Pacific blue-eye and Duboulay's rainbowfish. Pacific blue-eye were significantly lower in regulated sites in RFC 1 and densities of Duboulay's rainbowfish were significantly higher in regulated sites in RFC 2.

4. **Increasing degrees of flow regime change will have increasing impacts on ecological response variables:** This concept was tested firstly by relating ecological changes to the overall gradient of hydrologic alteration across the study area (as measured by the Gower dissimilarity metric). There were no significant ecological response gradients associated with the gradient of overall hydrologic alteration. This was due to a number of contributing factors but particularly the relatively gentle gradient of overall hydrologic alteration present across the study region, with a relatively low level of maximum change (0.25 on a scale of 0-1) and the presence of only a few strongly regulated sites. Other factors were that the overall hydrologic alteration gradient did not account for ecological differences among hydrologic classes spread over the gradient. Significant ecological responses to hydrologic alteration within each hydrologic class were established for the ecological assets studied in this project. However they did not form linear or threshold relationships apparently because each dam altered downstream hydrology and ecology in a different way.

   In the second test of this ELOHA concept, overall gradients of change in individual hydrologic metrics were examined and several pronounced ecological responses to alteration of these metrics were discovered. These relationships can be presented as graphs (Figure 7.1) that summarise all of the positive and negative ecological changes associated with positive and negative changes in individual hydrologic metrics. These graphs summarise the unique ecological responses to each type of hydrologic alteration downstream of individual dams and weirs.

Overall, the findings from these tests support the ELOHA principle that it is necessary to classify the hydrologic regimes of a region and examine ecological responses to each type of hydrologic alteration within each flow class. The finding that each dam has altered hydrologic regimes in a different way and produced different ecological responses supports the third ELOHA concept as described above.

Recommendations

For management

- Management and monitoring of flow regimes and river ecosystems across SEQ should take into account regional hydrologic variation (e.g. by incorporating the hydrologic classifications developed in this project). For instance, monitoring programs might ensure adequate coverage of all flow classes, while assessment of monitoring information might be conducted within and across these classes to ensure monitoring objectives and outcomes are considered with respect to regional variation in hydrology.
- Management and monitoring of flow regimes and river ecosystems across SEQ should take landscape context, particularly climatic factors, into account. These factors are significant drivers of ecological patterns, both directly and indirectly, through their influence on patterns of stream discharge, particularly for riparian vegetation and fish assemblages.
- Management and monitoring of flow regimes and river ecosystems across SEQ should recognise the significance of local habitat characteristics for flow-ecology relationships, especially hydraulic characteristics (e.g. sheer stress, substrate particle composition) with respect to the structure of riparian and aquatic vegetation.
- To maintain the health of aquatic ecosystems and minimise the potential for further ecological impacts of hydrologic alteration, management of flow regimes and catchments in SEQ should endeavour to maintain the relatively low current levels of overall hydrologic alteration.
- None of the rivers or streams included in this study can be considered to have ‘pristine’ discharge regimes since hydrologic alteration in SEQ has been shown here to be geographically widespread.
- Flow regimes in SEQ should be managed with particular emphasis on characteristics identified as having ecological importance, including:
  - for riparian vegetation, the variability of flows during the dry season and bankfull discharge
  - for aquatic vegetation, flows that mobilise the median particle size of stream substrates and the frequency with which such flow events occur; and
  - for fish, the occurrence and duration of periods of zero and low flow as well as daily hydrologic variability, the number of high discharge events (e.g. floods greater than the median discharge, and seasonal patterns of mean monthly discharge).
- Different management approaches are required for each dam within the study area since each dam has altered the downstream flow regime in a different way. Dam management should take into account the specific landscape, catchment, hydrologic and ecological characteristics associated with each dam and stream/river. Some specific recommendations for dam management include:
  - consideration should be given to increasing the numbers of high in-channel flow events downstream of all dams in the study region to limit the encroachment of the active channel by aquatic vegetation and riparian reeds, rushes and sedges
  - flow management at Six Mile Creek Dam should aim to restore specific flow metrics (e.g. moving averages of the annual minimum 3-90 day flows and their duration) closer to Reference conditions since decreases in low flows are associated with relatively significant ecological impacts downstream (e.g. for fish).
- The flow-alteration-ecological response relationships established during this study can be presented as ELOHA graphs (Figure 7.1). These demonstrate the effects of altered stream hydrology by revealing measured relationships between altered flow metrics and particular ecological responses compared to unregulated conditions for each study site. These graphs can be used to guide levels of user-defined ‘acceptable change’ in ecological metrics and their associated flow metrics at each site studied, and therefore can inform environmental flow management in each impounded stream/river.

For policy development

- Hydrologic alteration should be considered as a probable current and future risk to river ecosystem health in SEQ based on the findings of this ELOHA trial. The ecological effects of hydrologic alteration across SEQ are likely to continue, particularly for longer lived riparian vegetation and fish that may still be responding to past as well as present changes in hydrologic regime.
- Priority for revisions of environmental flow arrangements should be given to dams that have had relatively strong impacts on flow regimes in the region, since the greatest ecological impacts of hydrologic alteration generally occur in association with moderate to strong flow regulation downstream of dams, particularly on the Nerang River downstream of Hinze Dam.
- Metrics describing the condition of riparian and aquatic vegetation, fish assemblages and fish species could be used as indicators of hydrologic alteration impacts in monitoring programs such as the Ecosystem Health Monitoring Program (EHMP) of SEQ. They provide strong signals of hydrologic alteration, catchment condition and climate variability across the study area, and may therefore be useful indicators of land-use and climate change as well as hydrologic alteration.
- Restoration of flow regimes in SEQ should be used to provide opportunities for further validation of the hydro-ecological relationships identified in this project.

For future research

- Further analyses of the ecological datasets produced by this project are recommended. Considerable knowledge with significant implications for management and policy could be extracted from the datasets developed in this ELOHA trial through further analyses, addressing different questions from those asked in this project. Some key analyses might include:
  - identification of threatened or refugial habitats and biotic assemblages in rivers and streams and their riparian zones across SEQ, and development of targets for restoration
  - development of population models for key riparian, aquatic plant and fish species to gain improved understanding of the importance of flow and other factors for life history stages, strategies and recruitment processes
  - modelling of climate change impacts under a range of scenarios for riparian and aquatic vegetation and fish assemblages of SEQ
  - synthesis of the outcomes of further data analyses to support improvements to the ELOHA framework by suggesting ecological metrics that provide deeper insight into the ecological impacts of hydrologic alteration.
- Further trials of the ELOHA framework are recommended across different types of aquatic ecosystems and along stronger gradients of hydrologic alteration than those present in the SEQ study region.
- Key requirements to conduct such a trial include the availability of a good hydrologic monitoring network and either existing hydrologic models of pre-regulation discharge patterns or project team members skilled in hydrologic modelling. Ecological components considered in future trials may differ from those examined here depending on skills, knowledge and values placed on ecological assets in the selected region. Considerable skills in statistical analyses of ecological datasets are essential as are strong collaborative relationships between researchers, managers and other stakeholders such as land owners and community groups.

1. Introduction

1.1 Project objectives and scope

This project (*Hydro-ecological relationships and thresholds to inform environmental flow management and river restoration*) represents the first attempt in Australia to explore the scientific implications of using the framework ELOHA (Ecological Limits of Hydrologic Alteration) as a means to understand how flow regime alterations affect rivers and their riparian and aquatic biota in many rivers at a regional scale, and to interpret findings in terms of water management.

The following extract from the original submission to the National Water Commission (NWC) conveys the intent of the project:

> 'This project will provide a synthesis of hydro-ecological relationships in unregulated rivers of coastal and inland Queensland, and undertake field studies comparing unregulated and regulated rivers to identify thresholds of habitat and ecological response to flow regime alteration that will inform environmental flow management in rivers with contrasting flow regime characteristics and particular human 'footprints' (such as extent/type of rural and urban; riparian degradation, water quality impairment, alien species). The outcomes of the project will inform environmental flow management in rivers that may be regulated in the future, as well as the restoration of rivers that have been/are still regulated.'

1.2 Background

Land use change, river impoundment, surface and groundwater abstraction and artificial inter/intra-basin transfers profoundly alter natural flow regimes. Globally, the modification of river flows is so pervasive that the ~45,000 dams above 15 m high are capable of holding back >6500 km^3 of water, or about 15% of the total annual river runoff globally (Nilsson et al. 2005). In addition, increasing numbers of rivers are so deprived of water that they no longer reach the ocean, permanently or for parts of the year (Postel and Richter 2003).

A recent synthesis of threats to the world's rivers has found that impoundments and depletion of river flows are the clearest sources of biodiversity threat by directly degrading and reducing river and floodplain habitat, with 65% of global river discharge and aquatic habitat under moderate to high threat (Vörösmarty et al. 2010). Given escalating trends in species extinction, human population growth, climate change, water use and development pressures, the new global synthesis predicts that freshwater systems, and dependent societies, will remain under threat well into the future.

According to the global synthesis, Australia's freshwater crisis is less serious than in many parts of the world, yet demands new solutions to the challenges of sharing limited and spatially uneven water resources to meet productive, environmental and social objectives.

The National Water Initiative (NWI) has made a commitment to identifying over-allocated water systems and restoring those systems to environmentally sustainable levels of extraction, with emphasis on the quality of the science underpinning water plans and clear articulation of the environmental outcomes for threatened water-dependent ecosystems. The NWC's 2007 First Biennial Assessment of Progress in the Implementation of the NWI concluded that the protection of threatened water-dependent ecosystems, including the recovery of over-allocated systems, continues to be a major challenge in implementing the NWI Agreement.

1.3 Sustainable levels of water extraction

What exactly is a sustainable level of water extraction? How much water does a river need, when and how often? Many scientists and water managers have provided answers to this question in the form of environmental flow recommendations for thousands of rivers and floodplain wetlands around the globe.

Australian scientists have been at the forefront of recent developments in the field of environmental flow assessment, proposing what is now universally accepted as the way forward, that is to recognise the importance of the entire flow regime for the structure, functioning and productive capacity of rivers from source to terminus (Arthington et al. 1992; Walker et al. 1995; Puckridge et al. 1998; Kingsford 2006).

With publication of the 'natural flow regime paradigm' as a template for river conservation and restoration (Poff et al. 1997), scientists and practitioners have increasingly recognized that the structure and functions of the 'riverine' ecosystem, and many adaptations of its biota, are dictated by patterns of temporal variation in river flows (Lytle and Poff 2004). There is now broad general agreement among scientists and many water managers that to protect freshwater biodiversity and maintain the ecosystem services provided by rivers, natural flow variability must be maintained.

The rapid acceptance of the natural flow regime concept has been accompanied by an expectation that ecologists can easily provide specific environmental flow prescriptions for riverine ecosystems. Unfortunately, translating general hydrologic–ecological principles and knowledge into specific management rules for particular river basins and reaches remains a daunting challenge. Of the 200 or so methods available, around 70% still focus almost entirely on the habitat requirements of a few species (usually fish of recreational or commercial value), with less than 10% actually attempting to consider the entire natural flow regime and its ecological correlates (Tharme 2003). Australian river scientists have made major contributions to the development of this small minority of methods, proposing and elaborating 'holistic ecosystem methods' in collaboration with colleagues from other countries, particularly South Africa (Arthington et al. 2003; King et al. 2003).

From the early foundations of a source-to-terminal ecosystems approach, based upon the natural flow regime paradigm and concepts of river restoration, several frameworks have emerged – the Building Block Methodology, Flow Restoration Methodology, Flow Events Method, Flows, Benchmarking Methodology, DRIFT (Downstream Response to Imposed Flow Transformation), and ELOHA (Ecological Limits of Hydrologic Alteration).

1.4 The ELOHA framework

The central objective of ELOHA is to develop environmental flow prescriptions for multiple rivers rather than taking a river-by-river approach, as most methods do. Arthington et al. (2006) suggested a process that aims to quantify flow alteration – ecological response relationships for different types of river system classified according to their natural hydrological characteristics (magnitude, timing, frequency, duration and variability).

This process extends the Benchmarking Methodology and DRIFT by proposing development of quantitative flow alteration – ecological response relationships determined by empirical measurement along gradients of flow regime alteration, rather than ranking the ecological condition of regulated river sites into categories aligned to the severity of their alteration from the natural ('Reference') ecological condition.

Methods based on ranking of ecological condition have fundamental limitations, no matter how carefully applied, and generally, are too coarse to identify thresholds of ecological response to flow alterations. The best they can achieve is to identify 'thresholds of potential concern using professional judgement' (Biggs and Rogers 2003). Whether or not these potential thresholds actually represent thresholds of ecological response to perturbation is seldom known; therefore they serve mainly as precautionary guides to management action.

As well as improving the quantification of hydro-ecological relationships, Arthington et al. (2006) wanted to study how streams of different hydrological character (stable predictable rainforest streams versus highly variable arid zone streams, to take an extreme example) might differ in their responses to flow alteration (or to restoration of a regulated flow regime). In so doing, this approach could address the needs of water managers for transferable hydro-ecological relationships and environmental flow guidelines, rather than managing for the 'uniqueness' of each river's flow regime.

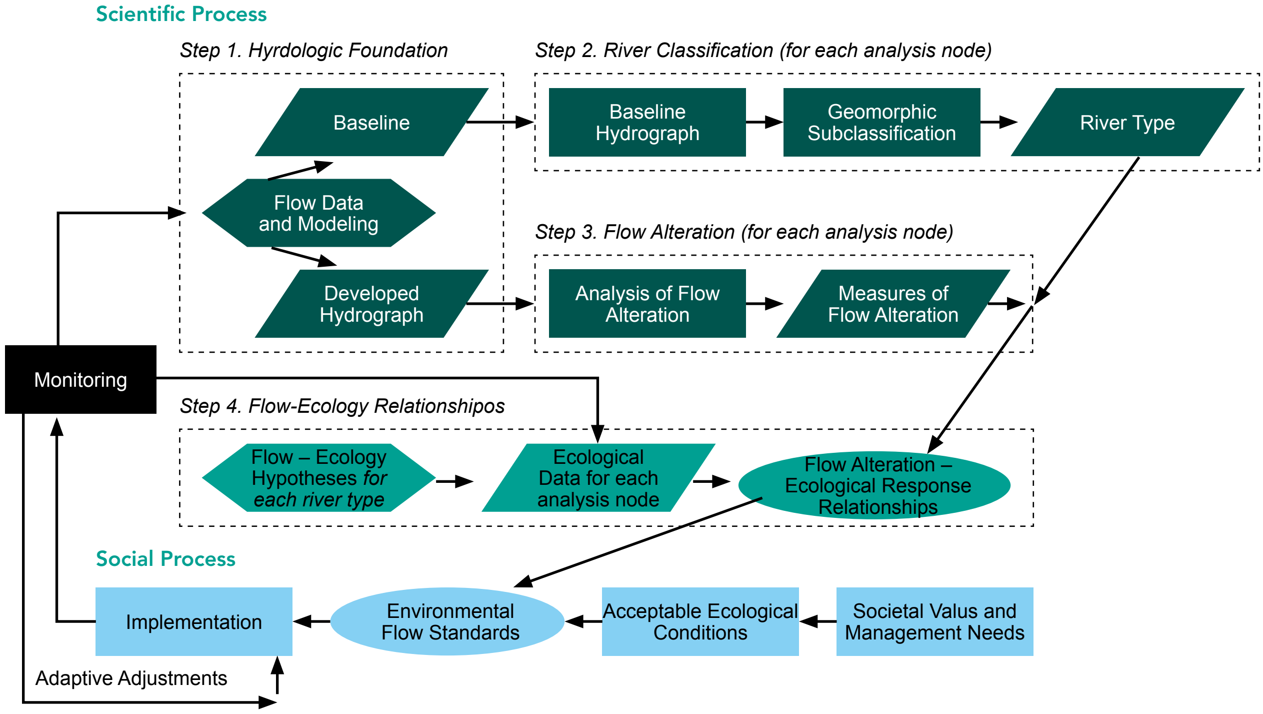
Within a region, the ecological characteristics of streams/rivers in each hydrological class are expected to be relatively similar compared to the ecological characteristics between the classes; therefore, these classes may represent distinct 'management units' (Arthington et al. 2006). By comparing ecological condition along flow–impairment gradients, it may be possible to develop and calibrate ecologically relevant flow standards for each hydrologic class of streams/rivers.

The aim is to develop empirical flow response curves for each natural asset of interest (e.g. habitat, aquatic and riparian vegetation, invertebrates, fish, other vertebrates, and ecosystem process rates, etc) and each ecologically relevant flow variable defining the stream class (e.g. low flow discharge, the magnitude, timing and frequency of flood flows, duration of low flow spells, and temporal variability etc.).

Attracted by these suggestions, The Nature Conservancy (TNC) invited 19 river scientists to develop a fully-fledged working environmental flow assessment framework now known as ELOHA (Figure 1.1; Poff et al. 2010). The ELOHA process consists of a biophysical and a social science module, with the major steps in the science module as follows:

1. Hydrologic modelling is used to build a 'hydrologic foundation' of baseline and current hydrographs for stream and river segments throughout the chosen study region.
2. Using a set of ecologically relevant flow variables, river segments within the region are classified into a few distinctive flow regime types that are expected to have different ecological characteristics.These river types can be further sub-classified according to important geomorphic features that define hydraulic habitat conditions for biota.
3. The deviation of current–condition flows from baseline–condition flows is determined for a suitable length of flow records.
4. Flow alteration – ecological response relationships are developed for each river type, based on a combination of existing ecological literature, field studies across gradients of hydrologic alteration, and expert knowledge; ideally, a parsimonious suite of flow metrics will emerge that collectively depicts the major facets of the flow regime and explains much of the observed variation in ecological response to particular kinds of flow alteration in each river flow type.
5. Interpretation of these hydro-ecological relationships and thresholds occurs in a consensus context where stakeholders and decision-makers explicitly evaluate acceptable risk as a balance between the perceived value of the ecological goals (and ecosystem services), the economic costs involved and the scientific uncertainties in functional relationships between ecological responses and flow alteration (Poff et al. 2010).
6. Implementation of ELOHA studies within an adaptive management context, where the objective is to formalize ongoing collection of monitoring data and targeted field sampling, to test and fine-tune the hypothesised flow alteration – ecological response relationships.

Figure 1.1: The ELOHA framework (reproduced from Poff et al. 2010)



The SEQ trial of the ELOHA framework set out to test the major tenets of the framework and to develop quantitative relationships between flow variables and three biotic communities of streams in the region—riparian and aquatic vegetation and fish. The project objectives, field methods, analyses and results are described in the following sections of this document, together with a summary of implications for management of environmental flows and monitoring the health of rivers and streams in the study region.

1.5 Objectives of the project

Environmental Objectives

1. To provide an analysis of the hydrological regimes of unregulated (Reference) river basins in Queensland that are relevant to the study areas selected for this project.
2. To provide a quantitative assessment of how the flow regimes of regulated rivers in these study areas have been altered by water infrastructure and the array of types/degrees of flow regulation. Flow metrics of relevance to ecological responses to flow alteration will be included in these analyses.
3. To provide a synthesis of knowledge of ecological responses to flow regime alteration in selected rivers within the study areas selected for the project. This will be based on past research supported by LWRRDC, Queensland DNRM, Queensland Fisheries, CRCFE, monitoring studies, Ph D theses, benchmarking studies for WRPs and consultancies.
4. To design a field research program that will identify how existing flow regime alterations in the study areas have impacted on habitat structure/heterogeneity, and the structure, dynamics and productivity of biological assemblages, life history strategies (aquatic plants, invertebrates, fish) and food web structure.
5. To implement the research field program in selected rivers of contrasting flow regime type and range/degree of hydrological alteration by studying habitat and ecological responses to gradients of change in each flow characteristic.
6. To identify thresholds (if there are any) or linear relationships of habitat and ecological response to flow regime alteration, with emphasis on responses of riparian vegetation, aquatic vegetation, and fish or other ‘indicators’ of ecosystem ‘health’ (structure, productivity, resilience).
7. To identify a limiting suite of flow variables that together govern the condition or ‘health’ of each river system (or river zone, or set of rivers in a bioregion) and thresholds levels of ecological response to flow regime alteration for the whole suite of flow variables.
8. To assess the relative influence of flow regime alteration *versus* other pressures (e.g. land use extent/type; riparian degradation, water quality impairment, presence of alien species) on habitat condition and ecological condition/health.
9. To provide information and guidelines on the relative influence of flow and other pressures on river ecosystems and practical advice on how to manage particular combinations of flow alteration and the other pressures so as to achieve healthier rivers.
10. To show how the findings of this study can be related to rivers and flow regime types beyond the geographic scope of this research project.

Educational Objectives

1. To provide opportunities for young scientists to participate in a large, multi-disciplinary field research program and to gain experience in field studies and the analysis and communication of its results.

1.6 Outline of the report

The project is reported in four parts: an Executive Summary, a Waterlines Report, a Scientific Report (this report) and Appendices to the Scientific Report (i.e. four literature reviews that underpin hypothesis development and design of field studies). The Scientific Report is set out according to the following sections and subject material:

Chapter 2: Field study area

The ELOHA field study was conducted in the Gold Coast, Logan–Albert, Brisbane, Pine–Caboolture, Maroochy, Noosa and Mary River catchments of coastal SEQ, Australia. This region was chosen for has several reasons. The ecology of streams and rivers in the region has been investigated by staff of the Australian Rivers Institute. It has a relatively dense stream gauging network and a variety of flow regime types.

The water resource needs of the region have been investigated (Moreton Basin Water Resource Plan; Logan–Albert Water Resource Plan; Mary Basin Water Resource Plan), thus the results of the project will be of direct relevance to the management of these catchments and specifically, to the monitoring and review of Water Resource Plans over the next 5–10 years.

Chapter 3: Flow regime classification

This chapter presents flow classifications for river catchments in SEQ, the study area for this trial of the ELOHA framework. Flow regime classification represents the first step in the ELOHA framework – the building of the 'hydrological foundation' to underpin analysis of hydro-ecological relationships in rivers with natural flow regimes and in rivers altered by dams and weirs.

It is essential to understand the natural patterns of variation in flow regimes of rivers across bioclimatic regions, and how dams and other types of water infrastructure can alter each different type of flow regime. These flow classifications are therefore central to the experimental design of the ELOHA field trial.

Individual flow classifications are presented for two flow scenarios: a modelled pre-development (more or less natural) flow scenario (the Reference flow classification, based on IQQM pre-development data) and an Historic flow scenario based on gauged records of flow over the past decades (the Historic flow classification).

The first classification allows the characteristics and variability of natural flow regimes within the study area to be described, while the second classification is intended to reveal changes in the flow regimes of localities affected by dams and weirs, and possibly, land use change. Both classifications are then used in the analysis of ecological responses to flow variability and flow alteration by dams and other factors.

Chapter 4: Field study design

This chapter presents an outline of the approach and methods used to design the fieldwork component of the ELOHA project in relation to the project aims (Chapter 1). Firstly, it describes the approach taken to select study sites, drawing on the results of the Reference flow regime classification (based on IQQM pre-development data) and the Historic flow classification (based on gauge data) identified in Chapter 3.

It then considers the requirements / constraints associated with surveying each of the focal biotic components (riparian vegetation, aquatic vegetation, fish) at suitable locations across a range of streams and rivers. The selection of specific field sampling locations or sites within reaches was made using aerial photos, satellite imagery, discussion with landholders, previous field experience and field visits. The final list of field sites is listed and located on a map of the study region.

Chapter 5: Land use

Whilst stream flows are recognised as one of the principal influences on stream ecology (Poff et al. 1997; Bunn and Arthington 2002), many other catchment characteristics not directly related to stream flows are also important drivers of stream habitat structure and ecological processes.

A key question for the ELOHA field trial, and for the ELOHA methodology as a whole, is whether the influences of flow can be extricated from the influences of landscape-scale environmental variability and anthropogenic disturbances. This has to be achieved in order to develop the generalised flow – ecology response models for the distinctive river flow classes fundamental to the ELOHA method.

This chapter is divided into two principal sections. The first section describes the methods used for the collation of landscape environmental variables and land use and management data. The second section investigates how landscape and land use variables that are potentially relevant to ecological responses co-vary with each other, with hydrological alteration and across the RFCs (based on IQQM pre-development data) and HFCs (based on gauge data) identified in the hydrological analysis (Chapter 3).

Chapter 6: Riparian vegetation

This chapter documents the riparian vegetation component of the ELOHA field trial in SEQ. It presents the objectives of the study and the major hypotheses tested during the field trial, followed by description of field, laboratory and statistical methods. The results of statistical analyses are presented and interpreted in relation to the major concepts of the ELOHA framework and hypotheses tested during the field trial.

Four main themes are discussed: relationships between catchment and in-stream environmental variables, land use, flow and vegetation structure; the importance of flow as an influence on riparian vegetation patterns; vegetation differences between regulated and unregulated sites; and vegetation responses to gradients of flow variability and flow alteration by dams.

The relevance and utility of the field results as guides to water management, ecological health and monitoring in SEQ are discussed, and the implications for future research to test, strengthen and refine the ELOHA framework and the flow–ecology relationships are outlined.

Chapter 7: Aquatic vegetation

This chapter documents the aquatic (in-stream) vegetation component of the ELOHA field trial in SEQ. It presents the objectives of the study and the major hypotheses tested during the field trial, followed by description of field, laboratory and statistical methods. The results of statistical analyses are presented and interpreted in relation to the major concepts of the ELOHA framework and hypotheses tested during the field trial.

Four main themes are discussed: relationships between catchment and in-stream environmental variables, land use, flow and vegetation structure; the importance of flow as an influence on aquatic vegetation patterns; vegetation differences between regulated and unregulated sites; and vegetation responses to gradients of flow variability and flow alteration by dams.

The relevance and utility of the field results as guides to water management, ecological health and monitoring in SEQ are discussed, and the implications for future research to test, strengthen and refine the ELOHA framework and the flow–ecology relationships are outlined.

Chapter 8: Fish

This chapter documents the fish component of the ELOHA field trial in SEQ. It presents the objectives of the study and the major hypotheses tested during the field trial, followed by description of field, laboratory and statistical methods. The results of statistical analyses are presented and interpreted in relation to the major concepts of the ELOHA framework and the hypotheses tested during the field trial.

Three main themes are discussed: relationships between catchment and in-stream environmental variables and fish assemblage structure; the importance of flow as an influence on fish assemblages and species; differences in fish metrics between regulated and unregulated sites; and fish responses to gradients of flow variability and flow alteration by dams.

The utility and relevance of the field results as guides to water management in SEQ are discussed, and the implications for future research to test, strengthen and refine the ELOHA framework and the flow–ecology relationships are outlined.

Chapter 9: Synthesis

The findings of this ELOHA trial in SEQ are synthesized in terms of project objectives in three categories – environmental, economic and social, and educational. These objectives are set out in section 1.5 above.

Chapter 10: Key recommendations

Key outcomes and recommendations for management, policy and research are sumamrised in this chapter.

Chapter 11: Bibliography

All literature cited in each chapter is presented in one block at the end of the Scientific Report.

Appendices

Appendices to the Scientific Report present three literature reviews on flow–ecology response relationships for riparian and aquatic vegetation and fish in SEQ streams and rivers, as follows:

- Appendix 1: Riparian vegetation–flow relationships and responses to flow regime alteration: a review of evidence from south-east Queensland streams. C. S. James and A. Barnes.
- Appendix 2: Aquatic vegetation–flow relationships and responses to flow regime alteration: a review of evidence from south-east Queensland streams. S.J. Mackay.
- Appendix 3: Freshwater fish–flow relationships and responses to flow regime alteration: a review of evidence from south-east Queensland streams. A.H. Arthington and D. Sternberg.

From these reviews, international literature and knowledge derived from personal research, 13 hypotheses have been developed as the main focus for field research during the trial of the ELOHA framework. Material contained in these review will be incorporated into the final Scientific Report, as appropriate, and into several future publications.

An additional component of the ELOHA study, which involved preparing a synthesis of past research on the ecology of fish in relation to flow regime in arid-zone floodplain rivers with Cooper Creek as a case history, has been published.

Arthington, A.H. and S.R. Balcombe (2011). Extreme hydrologic variability and the boom and bust ecology of fish in arid-zone floodplain rivers: a case study with implications for environmental flows, conservation and management. *Ecohydrology* 4: 708–720.

The intent of this review was to inform DERM's review of the Cooper Creek Water Resource Plan, and to support the proposal to designate Cooper Creek as a Wild River.

2. Study area

The ELOHA field study was conducted in the South Coast, Logan–Albert, Brisbane, Pine–Caboolture, Maroochy, Noosa and Mary River catchments of coastal SEQ, Australia (Figure 2.1). The chosen study area has several advantages in that the area has been well investigated by staff of the Australian Rivers Institute, it has a relatively dense stream gauging network and a variety of flow regime types, and the water resource needs of the region have been investigated (Moreton Basin Water Resource Plan; Logan–Albert Water Resource Plan; Mary Basin Water Resource Plan). The results of the project will be of direct relevance to the management of these catchments, and specifically to the monitoring and review of Water Resource Plans over the next 5–10 years.

The topography of the region is highly varied (Murphy et al. 1976; Beckman et al. 1987; Bridges et al. 1990; Malcolm et al. 1998). The main physiographic features are coastal lowlands of varying width, characterised by gently undulating terrain, often less than 30 m in elevation (Murphy et al. 1976; Young and Dillewaard 1999; Loi et al. 1998). The coastal lowlands give way to foothills and plateaux over 300 m above sea level (a.s.l.) to the west, north and south of the study region (Figure 2.1).

The Brisbane and Mary River catchments comprise approximately 72% of the total study area of 32 000 km^2 (Table 2.1). However, the Noosa and Maroochy catchments have higher mean annual runoff per unit of catchment area (560.8 and 782.8 Ml.year^{-1}.km^{-2}) than the Brisbane (82 Ml.year^{-1}.km^{-2}) and Mary (213 Ml.year^{-1}.km^{-2}) (Table 2.1). This reflects rainfall gradients across the region (Section 2.1). The volume of water held in storages varies considerably between catchments. The greatest volume of water is held in the Brisbane River catchment due to the presence of Wivenhoe and Somerset Dams (Chapter 3). The storage capacity of dams and weirs in the study area is approximately 38% of the mean annual runoff (Table 2.1).

2.1 Land use and vegetation

Land use throughout the region reflects topography, soils and distance from the coast. Agriculture is the dominant land use within the region, with approximately 40% of the total area used for grazing and 4% for cropping (Queensland DPI 1993). Extensive urbanisation has also occurred within the region, particularly along the coastal corridor.

SEQ is the most intensely populated region of Queensland and has one of the highest rates of population increase in Australia. The population of the region is predicted to increase by approximately 40% from 2004 to 2026 (Queensland Government 2004). The largest urban centre is Brisbane (population approximately 900 000, Queensland DNRM 2005a), but major urban centres occur immediately north and south of Brisbane.

The lower Brisbane River, Maroochy and South Coast catchments are the river catchments most impacted by urbanisation. The extent of urbanisation and agriculture within the region has resulted in widespread clearing of native vegetation (Young and Dillewaard 1999). Approximately 20% of the study area is National Park or State Forest reserve (Queensland DPI 1993) but remnant native vegetation is often located in steeper areas or on soils unsuitable for agriculture (Beckman et al. 1987).

Figure 2.1: Location and principal river catchments of the study area

Plots show maximum (filled circle) and minimum (open circle) mean daily temperatures and mean monthly rainfall (filled square).

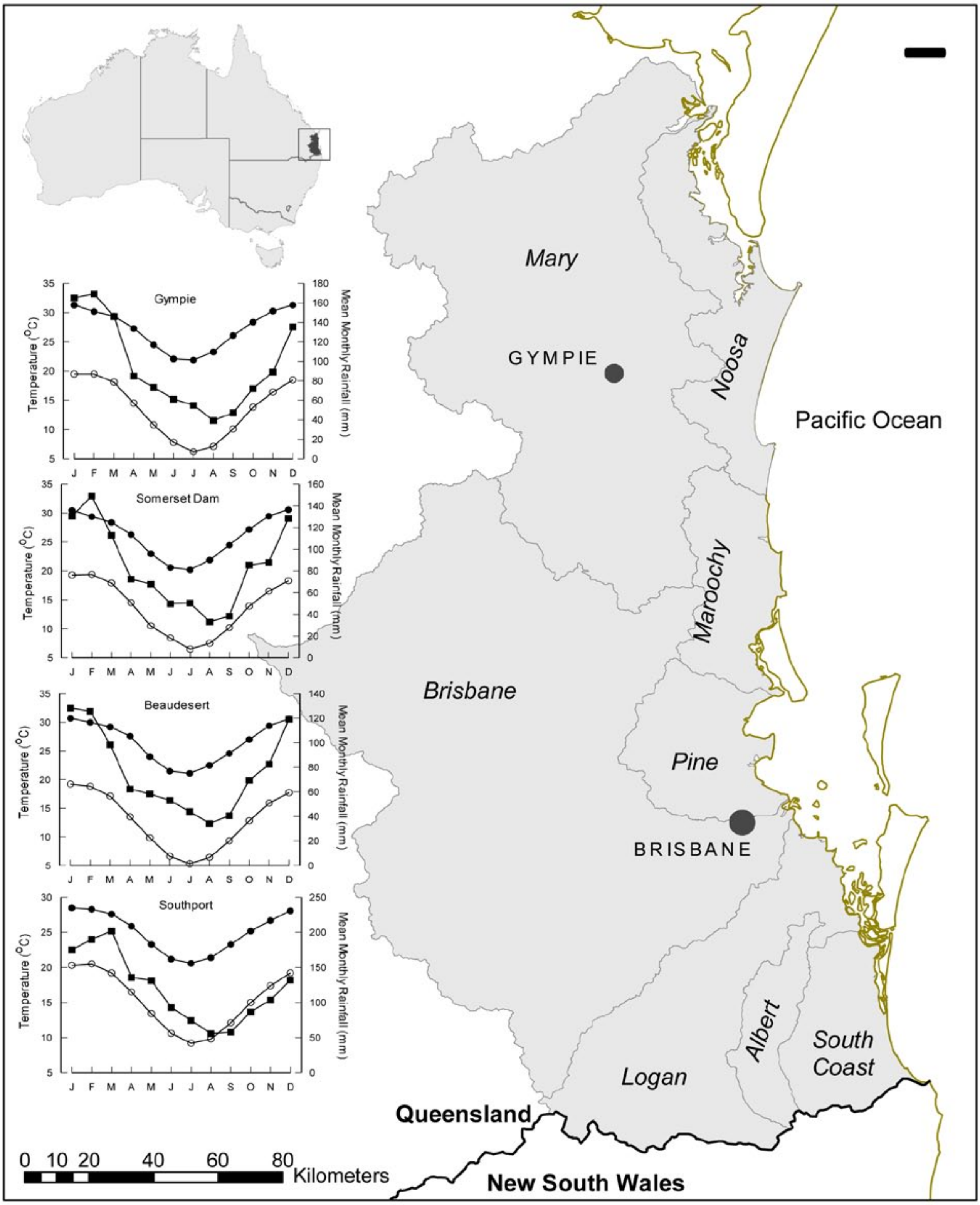


Table 2.1: Characteristics of principal river catchments in the study area

Sources: Queensland DPI 1993; Long and Lloyd 1997; Queensland DNRM 2001, 2002, 2005a,b. Mean annual runoff from Australian Natural Resources Atlas (http://www.anra.gov.au/topics/water/availability/qld).

Catchment	Catchment area (km^2)	Mean annual runoff (Ml per year, natural)	Approximate volume of water in storages (Ml)
Mary River	9 595	2 042 000	136 419
Noosa River	1 915	1 074 000	0
Maroochy-Mooloolah Rivers	861	674 000	30 031
Pine-Caboolture Rivers	1282	380 000	232 035
Brisbane River	13 560	1 112 933	1 870 260
Logan River	3 073	389 000	44 745
Albert River	782	185 700	30
South Coast[1]	1 302	658 000	173 141

[1] *Comprising Coomera River, Nerang River, Little Nerang Creek, Back Creek, Pimpama River, Mudgeeraba Creek, Currumbin Creek and Tallebudgera Creek.*

The study area is wholly contained within the SEQ Bioregion. This bioregion is characterised by high floral and faunal diversity (Young and Dillewaard 1999). Threats to biodiversity in the region include population expansion and associated land use changes, weeds and feral animals (Young and Dillewaard 1999).

Prior to European colonisation the vegetation of the region was dominated by rainforest (especially along the coast and adjacent hinterland at higher elevations), dry open sclerophyll forest characterised by *Eucalyptus* and *Angophora* spp. and wallum heathlands and *Melaleuca* swamps on coastal flats with sandy soils (Coaldrake 1961; Beckman 1967).

The composition of the riparian vegetation varies with elevation and hydrology but *Callistemon* spp. (Bottlebrush), *Castanospermum australe* (Black Bean), *Syzygium floribundum* (River Myrtle) and *Casuarina* spp. (River Oak) are common riparian tree species within freshwater reaches of the study area (Paton 1971; Arthington et al. 2000). Extensive weed invasion has occurred in some riparian zones impacted by agriculture and urban development (Arthington et al. 2000).

2.2 Climate

The climate of the region is subhumid and subtropical and is influenced by tropical and temperate weather patterns (Bridges et al. 1990; Pusey et al. 2004). The climate is classified as 'Cfa' under the Koeppen–Geiger climate classification system, that is the minimum temperature of the coldest month is between -3–18°C, rain occurs in all months, and the maximum temperature of the warmest month exceeds 22°C (Linacre and Hobbs 1977). Average maximum and minimum temperatures do not vary substantially throughout the study area and differ by approximately 20°C at any given location (Figure 2.1).

Rainfall patterns are more variable. A distinct longitudinal (east-west) rainfall gradient exists across the study area, with average annual rainfall varying from 1400 mm on the coast to 800 mm in the western part of the study area (Bridges et al. 1990; Young and Dillewaard 1999). However, rainfall can be high on the western ranges that border the region (Cosser 1989). Most rainfall occurs from January to March (Figure 2.1) and is often associated with thunderstorms (Bridges et al. 1990). Consequently streams and rivers of the region generally have late summer–early autumn discharge regimes (Musgrove 2003), with periods of low discharge occurring from August–November (Pusey et al. 2004).

However, temperate weather systems that produce winter rain in southern Australia may also produce significant rainfall in the study area from autumn to mid-winter (Bridges et al. 1990; Pusey et al. 2004). As the occurrence and intensity of summer and autumn–winter rainfall is irregular, discharge regimes of rivers and streams in the region are highly variable (Pusey et al. 2004; Chapter 3). The incidence of flooding in summer and autumn in SEQ is unpredictable and consequently the coefficient of variation of mean daily discharge is relatively high (Pusey et al. 2004).

3. Flow regime classification

3.1 Introduction

The flow regime describes the temporal or seasonal pattern of river flows (Gordon et al. 2005). The flow regime is typically characterised by five attributes: discharge *magnitude*, the *frequency*, *timing* and *duration* of discharge events (e.g. floods, droughts), and discharge *variability/predictability* (Poff et al. 1997).

These individual flow regime attributes can be described by numerous *flow* or *hydrologic metrics*, which can be calculated for a variety of time scales (e.g. days, months, years). Flow classification identifies streams and rivers with similar flow regime characteristics, using hydrologic metrics as the basis for classification. An extensive review of issues associated with calculating flow metrics and classifying flow regimes is presented in Kennard et al. (2010a,b).

This chapter presents flow classifications for river catchments in SEQ, the study area for this trial of the ELOHA framework (Poff et al. 2010). Individual flow classifications are presented for two flow scenarios: a modelled pre-development (more or less natural) flow scenario and an Historic flow scenario based on gauged records of flow over the past decades.

These classifications allow the characteristics and variability of natural flow regimes within the study area to be described, while the second classification is intended to reveal changes in the flow regimes of some localities affected by dams and weirs. These flow classifications represent the first step in the ELOHA framework – the building of the 'hydrological foundation' to underpin analysis of hydro-ecological relationships in rivers with natural flow regimes and in rivers altered by dams and weirs.

These flow classifications are therefore central to the experimental design of the ELOHA project. River catchments within the SEQ study area are shown in Figure 3.1.

3.1.1 Review of existing flow classifications

The flow regimes of SEQ river catchments have been the subject of several studies (Milton and Arthington 1983, 1984, 1985; Pusey et al. 1993; Arthington and Long 1997; Arthington and Lloyd 1998; Arthington et al. 2000; Pusey et al. 2000; Bunn and Arthington 2002; Mackay et al. 2003; Kennard et al. 2007; Mackay 2007; Stewart-Koster et al. 2007). These studies related aspects of flow regime, habitat structure and biotic response (e.g. species richness, community structure, life history strategy and timing) in unregulated and regulated streams and rivers of the region.

Flow regime classifications relevant to the present study are those completed by Pusey et al. (1993), Mackay (2007) and Kennard et al. (2010a). Pusey et al. (1993) described flow regime attributes for six stream gauges in the Mary River catchment. The flow regime of the Mary catchment was found to be of low to moderate predictability. High and low discharge events could occur within any month due to the influence of tropical and temperate weather patterns within the region. Flow predictability was related to position in the catchment. Downstream sites had more predictable discharge attributes compared with upstream sites.

Mackay (2007) classified hydrologic metrics calculated from mean daily discharge data for 45 stream gauges in SEQ. Some stream gauges included in the classification were influenced by flow regime alteration. The classification identified five flow regime classes in SEQ, four classes being dominated primarily by unregulated sites and a fifth flow class representing altered flow regimes. The high spatial variability in flow regimes across SEQ was attributed in part to the steep east-west rainfall gradient (difference in average annual rainfall approximately 600 mm) across the region.

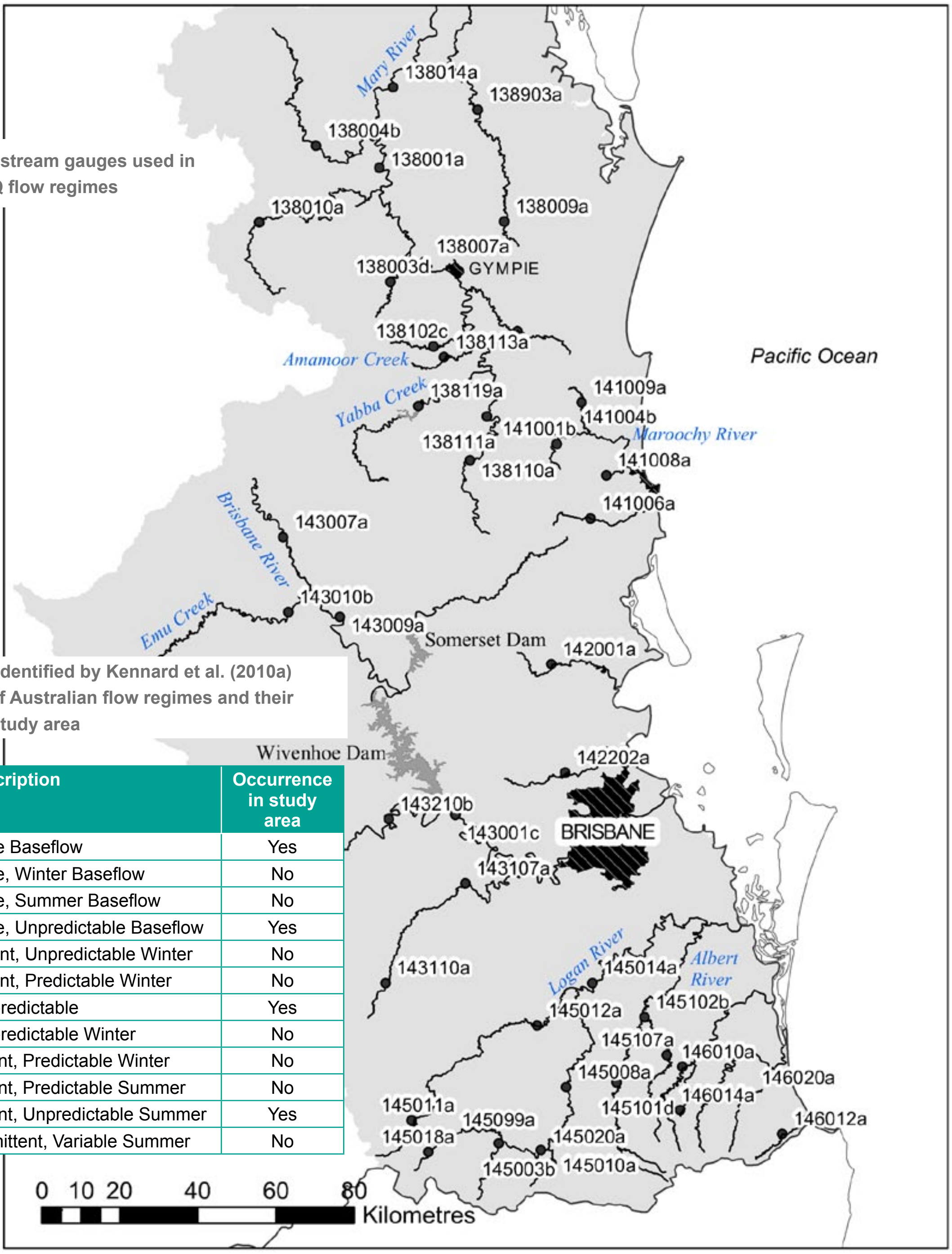


Figure 3.1: Locations of stream gauges used in the classification of SEQ flow regimes

Table 3.1: Flow classes identified by Kennard et al. (2010a) from the classification of Australian flow regimes and their occurrence in the SEQ study area

Flow class	Flow class description	Occurrence in study area
1	Perennial, Stable Baseflow	Yes
2	Perennial, Stable, Winter Baseflow	No
3	Perennial, Stable, Summer Baseflow	No
4	Perennial, Stable, Unpredictable Baseflow	Yes
5	Rarely Intermittent, Unpredictable Winter	No
6	Rarely Intermittent, Predictable Winter	No
7	Intermittent, Unpredictable	Yes
8	Intermittent, Unpredictable Winter	No
9	Highly Intermittent, Predictable Winter	No
10	Highly Intermittent, Predictable Summer	No
11	Highly Intermittent, Unpredictable Summer	Yes
12	Extremely Intermittent, Variable Summer	No

Kennard et al. (2010a) completed the first continental-scale classification of Australian riverine flow regimes. This classification, based on 830 minimally disturbed gauges and 120 hydrologic metrics, identified 12 flow regime classes across Australia and four flow regime classes within SEQ: Perennial–Stable Baseflow; Perennial–Unpredictable Baseflow; Intermittent–Unpredictable; and Highly Intermittent–Unpredictable Summer Dominated. Perennial–Stable Baseflow streams were characterised by high magnitude runoff, high baseflow constancy with relatively low discharge seasonality.

Perennial–Unpredictable Baseflow streams also had a weak seasonal discharge pattern but daily and annual flows were more variable than in Perennial–Stable Baseflow streams. Intermittent–Unpredictable streams were characterised by low constancy and predictability of flows and a low number of zero flow days per year (median approximately < 20 days per year).

Highly Intermittent–Unpredictable Summer (Dominated) streams were characterised by a summer-dominated discharge pattern with low discharge predictability and a high number of zero flow days per year (median approximately 150 days per year). The most common flow regime types were Perennial–Unpredictable Baseflow (mostly in the Logan–Albert and other southern catchments) and Intermittent–Unpredictable (spread throughout SEQ).

Kennard et al. (2010a) suggested that reclassification may be necessary where examination of hydrologic patterns at relatively fine spatial scales (e.g. well defined geographic areas such as SEQ) is required. In this instance it was recommended that the continental classification scheme be used to describe hydrologic variability within the context of continental-wide variability, and then classification using the 120 metrics be undertaken to describe flow variability within the desired study area. Table 3.1 shows flow classes identified by Kennard et al. (2010a).

3.1.2 Flow regime alteration in south-east Queensland

Anthropogenic alteration of natural flow regimes can be due to a number of activities, including the construction and operation of dams and weirs, unsupplemented extraction (extraction of water from natural river flows, as opposed to supplemented extraction, where explicit releases are made from storages for extraction downstream), interbasin and intrabasin water transfers and land use changes (Letcher et al. 2001; Siriwardena et al. 2006).

Dams and weirs are widespread throughout the study area (Queensland Department of Natural Resources and Mines (DNRM) 2001, 2002a; Australian National Commission on Large Dams 2002; Table 3.2). The extent to which dams and weirs influence downstream flow regimes varies considerably, depending on storage capacity, operating procedures and the frequency of spilling (Arthington et al. 2000; Poff et al. 2007).

Most dams occurring in the study area have capacities at full supply level of less than 50 000 Ml and were constructed by the mid 1970s (Table 3.2). Two significant dams in terms of storage volume in the study area (Wivenhoe Dam and Hinze Dam) were completed in the late 1980s. The storage capacity of dams and weirs in the study area is approximately 38% of the mean annual runoff (Table 2.1).

Table 3.2: Major dams and weirs in SEQ

Superscripts show storage function: S-storage, I-irrigation, H-hydroelectricity, C- flood control. Source: Queensland DNRM (2001, 2002a); ANCOLD (2002); Queensland Water Infrastructure Pty Ltd (http://www.qldwi.com.au); Seqwater (http://www.seqwater.com.au).

Catchment	Location	Storage	Storage Volume (ML)	Year Completed
Mary	Obi Obi Creek	Baroon Pocket Dam[S]	**61 000**	1989
	Yabba Creek	Borumba Dam[S,I]	**42 600**	1964
	Mary River	Mary Barrage	**11 700**	1983
	Six Mile Creek	Six Mile Creek Dam[S]	**9 300**	1964
	Tinana Creek	Tinana Barrage	**4 770**	1980
	Tinana Creek	Teddington Weir	**3 590**	1951
	East Deep Creek	Cedar Pocket Dam[I]	**730**	1984
Maroochy	Rocky Creek	Cooloolabin Dam[S]	**13 500**	1979
	South Maroochy River	Wappa Dam[I]	**4 550**	1961
Mooloolah	Addlington Creek	Ewen Maddock Dam[S]	**9 100**	1973
Pine	North Pine River	North Pine Dam[S]	**202 000**	1976
	Sideling Creek	Sideling Creek Dam[S]	**15 500**	1969
Brisbane	Brisbane River	Wivenhoe Dam[S,C,H]	**1 150 000**	1985
	Stanley River	Somerset Dam[S,C,H]	**369 000**	1953
	Reynolds Creek	Moogerah Dam[S,I]	**92 500**	1961
	Cressbrook Creek	Cressbrook Dam[S]	**81 842**	1983
	Atkinson Lagoon	Atkinson Dam[I]	**31 300**	1970
	Perseverance Creek	Perseverance Dam[S]	**30 940**	1965
	Pryde Creek	Split-yard Creek Dam[H]	**28 600**	1980
	Cabbage Tree Creek	Lake Manchester[S]	**26 217**	1916
	Offstream	Clarendon Dam[I]	**23 300**	1992
	Offstream	Bill Gunn Dam[I]	**7 520**	1987
	Enoggera Creek	Enoggera Reservoir[S]	**4 500**	1866
	Brisbane River	Mt Crosby Weir	**3 430**	1892
	Gold Creek	Gold Creek Dam[S]	**1 595**	1885
	Sandy Creek	Enoggera Army Dam[I]	**200**	1971
Tingalpa	Tingalpa Creek	Leslie Harrison Dam[S]	**24 800**	1968
Logan–Albert	Burnett Creek	Maroon Dam[I]	**38 400**	1974
	Offstream	Bromelton Dam	**8 210**	2008
	Logan River	Cedar Grove Weir	**1 139**	2007
	Logan River	Bromelton Weir	**410**	1996
South Coast	Nerang River	Hinze Dam[S]	**165 000**	1989
	Little Nerang Creek	Little Nerang Dam[S]	**8 390**	1961

The most widespread form of flow regime alteration in SEQ is arguably unsupplemented extraction whichmay extend or exacerbate low spell durations but has little effect on high flow spell magnitude and duration (Brizga et al. 2005a).

Several interbasin and intrabasin water transfer schemes exist in the study area. In the Mary River catchment interbasin transfer schemes exist between Yabba Creek and the Noosa River (via extraction from the Mary River downstream of the Yabba Creek confluence), between the Mary Barrage and Teddington Weir and the Tinana Barrage on Tinana Creek, and water from Baroon Pocket Dam enters the Maroochy River estuary via reticulated water supplies and urban water returns (Brizga et al. 2005a). Several schemes exist in the Brisbane River catchment.

The Moogerah Dam and Warrill Valley Water Supply Scheme have extensive impacts upon the hydrology of Warrill Creek and Reynolds Creek. The Buaraba Creek–Atkinson Dam scheme involves the transfer of water between waterbodies that have natural connectivity and therefore impacts associated with the scheme are relatively minor (Brizga et al. 2006a). Additional water transfer schemes in the Brisbane catchment include the Kholo pipeline and Wivenhoe Dam–Tarong Powerstation in the Burnett River catchment (Brizga et al. 2006a).

3.1.3 Aims

The aims of this chapter are to:

1. identify flow regime classes in the study area (i.e. groups of streams with similar flow regime characteristics), based on the classification of flow metrics representing the five facets of the flow regime
2. identify a subset of flow metrics that best discriminate the flow classes identified
3. characterise flow regime alteration that has arisen from the construction and operation of dams and weirs in the study area.

The flow classifications presented in this chapter underpin the experimental design for the project and provide a more detailed classification than presented by Kennard et al. (2010a) for SEQ.

3.2 Methods

3.2.1 Data sources

The flow regime classifications were based on two datasets; modelled pre-development flow data derived from an ''IQQM (Simons et al. 1996) and stream gauge data. IQQM pre-development data represents Reference (pre-European) conditions and stream gauge data represents Historic flow conditions (i.e. the actual discharge recorded at an individual stream gauge through time, which may be influenced by changes to land use through time, water resource development and unsupplemented extraction).

IQQM data are modelled for nodes in the river network, such as the junctions of tributaries or at dams or weirs, and they often coincide with currently operating or decommissioned gauging stations. Discharge data for a total of 87 IQQM nodes and 72 gauges were obtained from the Queensland DERM (Attachment 3.1). Flow data for Obi Obi Creek at Kidaman (138104a) was obtained from Water Quality Accounting (Queensland DERM) as modelled gauge data derived from a calibration model for the Mary River catchment.

Hydrologic metrics calculated from IQQM pre-development flow data are designated *Reference metrics* and metrics calculated from gauged flow data are designated *Historic metrics*. Similarly, the classification based on Reference flow metrics is termed the *Reference flow classification* and the classification based on Historic flow metrics is termed the *Historic flow classification*.

It is acknowledged that several sources of error exist with measurement and modelling of stream flow data, but that these are largely beyond the control of this project (Kennard et al. 2010b). The metrics calculated in this report are based on the concept that the flow data provided are the best available, given potential errors in gauging and modelling flows.

3.2.2 Period and length of flow record used for metric calculation

The length of discharge record and time period of discharge records used to calculate flow metrics were determined by the period of discharge record available for Reference (IQQM) and Historic (gauge) data and the protocols of Olden and Poff (2003) and Kennard et al. (2010a,b). The periods of record available for IQQM nodes were 1890–1999 for the Mary and Maroochy catchments, 1889–2000 for the Brisbane River and Pine–Caboolture catchments, 1890–2003 for the Logan–Albert catchment and 1890–2000 for the South Coast catchments (small coastal catchments extending from the Pimpama River to the New South Wales border – Figure 3.1).

The period of record available for gauge data varied according to the period of operation of individual gauges (Attachment 3.1) and whether periods of missing data (if present) could be infilled adequately (Section 3.2.3). The period 1975–2000 was chosen for calculation of flow metrics as a compromise between the periods of record available and the need to maximise the length of record and number of gauges available for analysis.

The flow records chosen were not concurrent (i.e. start and end dates of the flow records varied between gauges). Concurrent flow records are not essential as long as all flow records are contained within a discrete temporal window (Kennard et al. 2010b). The advantage in using non-concurrent hydrologic records is that it may increase the number of stream gauges available for classification.

A minimum record length of 15 years within the period 1975–2000 was chosen as being sufficient to calculate hydrologic metrics representative of the true long-term value (Kennard et al. 2010b). Discharge data were arranged by water years to avoid splitting the flood season across consecutive calendar years (Gordon et al. 2005). The month with the lowest mean monthly discharge (October) was selected as the start of the water year (Gordon et al. 2005). The maximum period of record used to calculate flow metrics was therefore 1 October 1975–30 September 2000, with a minimum record length of 15 years. Sixty gauges fulfilled these criteria (Attachment 3.1).

3.2.3 Data screening and infilling

Historic flow data were screened for periods of missing data prior to flow metric calculation. Reference (IQQM) data did not contain missing data. Periods of missing record were infilled by linear interpolation or multiple regression. Linear interpolation was used to infill missing periods of up to 15 days (Nature Conservancy 1997).

Multiple regression was used where an adjacent stream gauge was available to predict stream discharge for the gauge being infilled. The Reynolds Creek gauge (143112a) had three substantial periods of missing data (34, 97 and 134 days) in the time period chosen for calculation of flow metrics. These gaps could not be infilled by regression as the flow regime at this gauge is altered by Moogerah Dam. The discharge record selected for calculation of flow metrics for Reynolds Creek was shortened to 1/10/1981–30/9/1991 to exclude the two longest periods of missing data.

The shortest remaining period of missing data (34 days) was infilled by linear interpolation. While this was not an ideal method of infilling it was the only method available to provide a sufficient period of record for this gauge. This gauge was not included in the Historic flow classification but was assigned to a HFC using random forests (Section 3.2.5).

Other stream gauges with periods of missing record that could not be infilled appropriately (i.e. linear interpolation was not appropriate or adjacent stream gauges were not available for development of multiple regression models) were excluded from the Historic flow classification.

3.2.4 Selection of flow metrics

A large suite of metrics has been used to characterise flow regimes (Growns and Marsh 2000; Olden and Poff 2003). Given the relatively small size of our study area (approximately 32 400 km^2) a relatively small suite of ecologically relevant flow metrics was selected to characterise the five key facets of the flow regime (i.e. *magnitude*, *frequency*, *timing* and *duration* of discharge events, and discharge *variability/predictability*) whilst minimising metric redundancy (Olden and Poff 2003; Poff et al. 2010). The metrics included in the Indicators of Hydrologic Alteration (IHA) software package (Nature Conservancy 2007) have been identified as describing key facets of the flow regime while minimising metric redundancy (Olden and Poff 2003). The IHA package calculates 33 hydrologic metrics and measures of variability for each metric. The IHA metrics used in this study are listed in Table 3.3.

An issue with the IHA metric set is that high flow conditions are not represented adequately (Olden and Poff 2003). To compensate for this the Median of Annual Maximum Flows and Specific Mean Annual Maximum Flows were also included (Olden and Poff 2003). The magnitude of floods with Average Recurrence Intervals (ARIs) of 1, 2 and 10 years were also included as potentially important indicators of the frequency of inundation of banks (e.g. for riparian vegetation), and Colwells Indices (Colwell 1974) were included as indicators of flow predictability, constancy and seasonality.

Since it is not necessary to use the entire IHA metric set for flow classification (Olden and Poff 2003) the number of metrics describing discharge magnitude was reduced by excluding mean monthly discharge metrics for February, April, June, August, October and December (Table 3.3).

3.2.5 Statistical methods

Hydrologic metrics were calculated using the River Analysis Package (RAP, Marsh et al. 2003) and the IHA package (The Nature Conservancy 2009). Magnitude metrics were standardised by upstream catchment area to downweight the influence of these metrics on the Reference and Historic classifications. Principal components analysis (PCA) was used to investigate redundancy in Reference and Historic flow metric datasets. The PCA was based on the correlation matrix (Olden and Poff 2003). Retention of components was determined by inspection of scree plots, eigenvalues and metric loadings on components. Metrics with loadings within the range ±0.5 were excluded from classification (Tabachnik and Fidell 1989).

Table 3.3: IHA parameters used in the classification of SEQ flow regimes

Parameters in bold text are additional parameters thought to be of relevance to aquatic biota in the study area. Magnitude metrics were standardised by upstream catchment area (except for Specific Mean Annual Maximum Flow, Sp_MeanAnnMax) to reduce the influence of discharge magnitude metrics on classification. Ecological relevance partially based on Richter et al. (1998) and Graf (2006).

Flow regime component	Metrics	Acronym	Ecological relevance
Magnitude (23 metrics)	Mean daily flow (January, March, May, July, September, November) [1]	MDF_Jan, MDF_Mar, etc.	Maintenance of channel form, habitat heterogeneity, dispersal of biota, germination of seeds, availability of water for riparian and in-stream vegetation, availability of habitat for fish
	Annual minima, 1 day mean	MA1dayMin	
	Annual minima, 3 day means	MA3dayMin	
	Annual minima, 7 day means	MA7dayMin	
	Annual minima, 30 day means	MA30dayMin	
	Annual minima, 90 day means	MA90dayMin	
	Annual maxima, 1 day mean	MA1dayMax	
	Annual maxima, 3 day means	MA3dayMax	
	Annual maxima, 7 day means	MA7dayMax	
	Annual maxima, 30 day means	MA30dayMax	
	Annual maxima, 90 day means	MA90dayMax	
	Baseflow Index (ratio of baseflow to total flow)	BFI	
	Mean number of zero flow days per year	MeanZeroDay	
	Magnitude of floods with average recurrence intervals of 1, 2, 10 years	ARI_1yr, ARI_2yr, ARI_10yr	
	Specific Mean Annual Maximum Discharge [2]	Sp_MeanAnnMax	
	Median of Annual Maximum Discharge	MedAnnMax	
Timing (5 metrics)	Julian date of each annual 1 day maximum discharge	JDMax	Seasonal cues for reproduction and migration, availability of habitat for spawning
	Julian date of each annual 1 day minimum discharge	JDMin	
	Predictability of mean daily discharge	PREDICT	
	Constancy of mean daily discharge	CONSTAN	
	Seasonality of mean daily discharge (Contingency/Predictability)	SEASON	
Frequency and Duration (4 metrics)	Number of high pulses within each year [3]	HSNum	Availability of floodplain habitats, soil moisture stress, frequency of substrate mobilisation, completion of life cycles
	Number of low pulses within each year [3]	LSNum	
	Duration of high pulses within each year	HSDur	
	Duration of low pulses within each year	LSDur	
Rate and frequency of change (3 metrics)	Mean rate of discharge rise	RateRise	Erosion of banks, bars; habitat stability, frequency of inundation and exposure
	Mean rate of discharge fall	RateFall	
	CV of mean daily discharge	CVDaily	

Mean monthly discharge metrics reduced from 12 to 6 to downweight the influence of magnitude metrics on flow classifications.
Calculated as the mean annual maximum flow divided by catchment area.
Based on 25^{th} (low spell threshold) and 75^{th} percentiles (high spell threshold).

Classification was undertaken using the Mclust package for R (Fraley and Raftery 2008; R Development Core Team 2010). Mclust is a model-based hierarchical agglomerative clustering procedure based on Gaussian finite mixture models. Model-based clustering assumes that the observed data come from a population comprised of several subpopulations (Raftery and Dean 2006).

Each subpopulation is modelled separately and hence the entire population is comprised of a mixture of models. In the context of flow classification each subpopulation represents a flow class. Individual models are probability density functions, usually multivariate normal distributions, parameterised by their means and covariances (Fraley and Raftery 2007). Covariance properties determine cluster geometry, i.e. shape, volume and orientation (Fraley and Raftery 2002).

These attributes can be fixed or allowed to vary between clusters. Models appropriate for multidimensional data (where the number of cases exceeds the number of variables) were used following the recommendation of Fraley and Raftery (2008) and cluster geometry was limited to spherical models (options 'EII', 'VII') and diagonal models (options 'EEI', 'VEI', 'EVI', 'VVI').

Clustering is undertaken by merging pairs of clusters so that the classification likelihood is maximised, as summarised below (Fraley and Raftery 2002):

1. Determine the maximum number of clusters (*M*) and set of mixture models to consider (in Mclust the maximum number of clusters is 9).
2. Perform hierarchical agglomeration to provide an approximate classification likelihood for each model and obtain the corresponding classifications for a maximum of *M* groups.

3. Apply the EM (expectation–maximisation) algorithm to estimate parameters for each model and each number of clusters, using the classification likelihood obtained from step 2 to provide starting values for the EM algorithm.
4. Compute Bayes Information Criterion (BIC) using optimal parameters determined from the EM algorithm for 2-*M* clusters. The model and number of clusters that maximises BIC is the optimal classification.

Several methods were used to identify the flow metrics best discriminating the flow classes identified in the Reference and Historic flow classifications. The clustvarsel package for R (Dean and Raftery 2009) was used to identify a subset of flow metrics in which all flow metrics contain classification information (Raftery and Dean 2006).

The clustvarsel algorithm identifies a flow metric to add to the flow metric subset that best improves the classification and then determines whether an existing flow metric in the subset can be dropped (Raftery and Dean 2006). The algorithm begins by identifying the flow metric that has the most evidence of univariate clustering, next identifies the second clustering variable that has the most evidence of bivariate clustering and then selects the next clustering variable as the one that shows the best evidence for multivariate clustering (whilst including the first two variables). The algorithm then searches for a flow metric to drop from the subset, based on change in Bayes Information Criterion. The procedure is repeated until no metric can be found to include, and no metric can be found to drop (Raftery and Dean 2006).

The Reference and Historic flow classifications were validated using *random forests*, a decision tree method similar to classification and regression tree methods (Breiman 2001). The random forests method produces many classification trees and combines the predictions from all trees, which reduces the error of prediction that may occur from a single classification tree (Cutlet et al. 2007).

In summary, a bootstrap sub-sample of the dataset (approximately two thirds of the observations) is used to construct a classification tree. Each tree is constructed using a randomly selected subset of the available predictor variables to determine splitting rules at each node (Cutlet er al. 2007). These trees are used to predict class membership for cases in the remaining one third of the dataset (the out-of-bag or OOB sample).

Class membership (in this case, membership of IQQM nodes and stream gauges in flow classes) is determined as the class with the highest number of votes (Cutler et al. 2007). Flow classifications were validated by inspection of the confusion matrix from each random forests model. The confusion matrix is a summary of misclassification rates and compares the flow class membership for each case (i.e. IQQM node and gauge) as determined by the Mclust classification with the flow class predicted by the random forests models.

The number of flow metrics to try at each node was determined using the tuneRF function in the randomForest package for R (Liaw and Weiner 2009). This function constructs random forests and compares the out-of-bag (OOB) error rate when the number of flow metrics used at each node is varied. One thousand trees were constructed for each hydrologic classification to ensure that random forests models were not overfit (Breiman 2001).

Variable importance (a measure of misclassification rate when the values for an individual predictor are permuted) was assessed using mean decrease in prediction accuracy and the Gini index (Liaw and Weiner 2009). Mean decrease in accuracy for individual predictors (flow metrics) is assessed using the OOB samples. Values for each predictor variable are randomly permuted for the OOB samples and passed down each tree to get new predictions. The difference in the misclassification rate between the original and modified OOB sample is averaged over all trees and normalised by the standard error (Cutler et al. 2007; Liaw and Weiner 2009).

The Gini index is a measure of node impurity and is calculated as the sum of the proportion of cases belonging to each class. A terminal node has high purity when all the cases in that node belong to a single class. For an entire tree the Gini index is determined as the sum of the values of the Gini index for all terminal nodes, weighted by the number of cases in each node (Cutler et al. 2007).

Finally, relationships between flow classes were examined using non-metric multidimensional scaling (nMDS) as implemented in the vegan package for R (Oksanen et al. 2010). Separate ordinations were completed for Reference and Historic metric sets. The association matrix was generated using the Gower (dissimilarity) metric. Since the Gower metric standardises by the range of each variable (Gower 1971) flow metrics were not standardised prior to calculation of the association matrix. Fifty random starts were used to find the configuration that minimised stress (goodness of fit between observed dissimilarities and ordination distances).

The *metaMDS* function rotates the best solution (rotation to principal components) so that maximum variation is displayed on the first ordination axis (Oksanen et al. 2010). The *envfit* function was used to fit environmental vectors to the ordination space. This function finds the maximal correlation between intrinsic variables and the ordination space. Significance was assessed using a randomisation procedure and 999 permutations. The results from clustvarsel and ordination were used to describe RFCs and HFCs in terms of the classification scheme of Kennard et al. (2010a) for continental Australia.

3.2.6 Assessing flow regime alteration in the study area

Several techniques were used to assess the extent of flow alteration within the study area. Firstly, the Gower metric (Gower 1971) was used to compare Reference and Historic flow regimes. Random forests were used to confirm changes in the hydrologic regimes of individual sites as identified with the Gower metric. The Reference random forests model (Section 3.2.5) was used to allocate gauges to the RFCs determined by Mclust. For this analysis gauges were allocated to RFCs according to the class membership of the corresponding IQQM node.

It was assumed that if there was little or no hydrologic change between Reference and Historic flow regimes then the Reference random forests model should be able to allocate individual gauges to the same RFC as the corresponding IQQM node.

The Range of Variability Approach (RVA, Richter et al. 1997) was used to investigate changes in individual flow metrics (i.e. differences in individual metrics between the Reference and Historic flow regimes). The RVA approach can be used to compare flow records for two distinct time periods, such as pre- and post-dam periods. The availability of pre-dam discharge data varied between dams.

The two year period either side of the construction date was not used in defining pre-dam and post-dam flow regimes (Table 3.4). Using this criterion sufficient pre-dam discharge data were available for Baroon, Six Mile Creek and Hinze Dams only.

The pre-impact or Reference time period for each flow metric is divided into three categories. The default RVA percentiles were used to define the categories (defaults are 17 percentiles from the median, i.e. 33rd and 66th percentiles). This produces three categories with an equal number of values. The low category contains values up to the 33rd percentile, the middle category contains values in the 34–67th percentile range, and the high category contains values greater than the 67th percentile. Next, the expected frequency with which post-dam values for individual flow metrics should fall within each of the three categories is calculated. If there was no flow regime impact then the low, middle and high categories should each contain 33% of flow metric values in the post-dam flow record.

The expected frequency is equal to the number of values in each of the low, middle and high categories during the pre-dam period multiplied by the ratio of post-dam years to pre-dam years of flow record (The Nature Conservancy 2009). The actual number of values in each of the three categories is then calculated for the post-dam period.

Hydrologic alteration for each metric is measured by a Hydrologic Alteration factor which is calculated for each of the three categories as (pre-dam frequency – post-dam frequency) / post-dam frequency (Nature Conservancy 2009). A positive Hydrologic Alteration factor indicates that the frequency of values in the category has increased in the post-dam period (maximum value infinity) and a negative value indicates that the frequency of values in the category has decreased (maximum decrease possible is -1) (The Nature Conservancy 2009).

Table 3.4: Pre and post-dam periods of flow record used for the RVA

Insufficient pre-dam data were available for Borumba Dam (Yabba Creek) and Maroon Dam (Burnett Creek).

Dam	Construction date	Pre-dam record	Post-dam record
Six Mile Creek Dam (Six Mile Creek)	1964	1947–1962	1966–2005
Baroon Pocket Dam (Obi Obi Creek)	1989	1979–1987	1991–1999
Moogerah Dam (Reynolds Creek)	1961	1931–1959	1980–2009[2]
Hinze Dam (Nerang River)	1989[1]	1956–1974	1991–2009

[1] *The dam was first constructed in 1976 (capacity 42 400 ML) but substantially upgraded in 1989. Pre-dam record taken to end in 1974*

[2] *65 days of missing record in this interval*

3.3 Results

3.3.1 Reference flow classification

PCA and flow metric redundancy

PCA of Reference flow metrics explained 88.0% of the variation in the dataset with five components (Table 3.5). As all metrics had loadings less than -0.5 and greater than 0.5 on at least one component, all metrics were retained for classification. While component 5 only had two metrics loading on it (MedAnnMax and JDMax) these loadings were high (>0.7) and it was therefore retained (Tabachnik and Fidell 1989). Five metrics (MeanZeroDay, CVDaily, LSDur, HSDur, SEASON) had high negative loadings (i.e. less than -0.5) on individual components but did not have high positive loadings (i.e. greater than 0.5) on any components (Table 3.5).

Component 1 explained 48% of the total variance and was associated with metrics describing high discharge magnitude (e.g. mean monthly discharge, ARIs, moving averages of flow maxima). Component 2 explained approximately 19% of the total variance and was associated with low discharge magnitude (moving average of flow minima, MDF_Sep [generally a month of low flow]), baseflow index (BFI) and negatively associated with discharge seasonality (SEASON).

Thus IQQM nodes with high baseflow tend to be less seasonal than IQQM nodes with low baseflow. Components 3–5 each explained less than 10% of the total variance and represent flow spell number (component 3), timing of flow minima and flow stability (component 4) and timing of flow maxima (component 5). Metric loadings on component 3 suggest that spell frequency is inversely related to spell duration (i.e. high spell frequency is associated with short spell duration).

Flow metric loadings on component 4 suggest that sites with a high mean number of zero flow days per year have high discharge variability (CVDaily) with the annual flow minimum occurring relatively late in the year.

Table 3.5: Rotated component matrix of variable loadings from PCA of Reference flow metrics

Percentage variation explained by each component is shown in brackets. See Table 3.3 for flow metric acronyms. Loadings ≤-0.5 and ≥0.5 shown in bold text.

Flow metrics	Component and variation explained				
	1 (48.0%)	2 (19.1%)	3 (9.7%)	4 (6.6%)	5 (4.6%)
MDF_Jan	.967	.174	.047	.081	.005
MDF_Mar	.907	.092	.038	.197	.150
MDF_May	.944	.195	.045	.169	.034
MDF_Jul	.910	.331	.048	.147	.070
MDF_Sep	.812	.521	-.025	.176	-.035
MDF_Nov	.909	.275	.159	.034	-.016
ARI_1yr	.948	.056	.177	.027	.118
ARI_2yr	.975	.040	.093	.092	.089
ARI_10yr	.969	.050	.056	.056	.063
MedAnnMax	.298	-.056	.127	-.146	.722
MA1dayMin	.351	.887	-.093	.134	.008
MA3dayMin	.390	.877	-.091	.148	.010
MA7dayMin	.432	.860	-.088	.160	.001
MA30dayMin	.549	.797	-.053	.188	.001
MA90dayMin	.754	.610	.005	.194	.008
MA1dayMax	.976	.022	.101	.061	.123
MA3dayMax	.977	.057	.075	.071	.052
MA7dayMax	.976	.070	.060	.097	.043
MA30dayMax	.975	.100	.041	.124	.049
MA90dayMax	.971	.131	.038	.139	.056
RateRise	.861	-.050	.209	.005	.242
RateFall	.886	-.020	.226	.029	.211
CVDaily	-.343	-.296	-.194	-.673	-.179
BFI	.074	.685	-.220	.508	.016
MeanZeroDay	-.070	-.144	.026	-.811	.374
Sp_MeanAnnMax	.965	-.013	.098	.074	.138
LSNum	.003	-.020	.897	-.160	.175
LSDur	-.101	-.015	-.782	-.276	-.001
HSNum	.227	-.027	.880	.017	.142
HSDur	-.240	-.021	-.871	-.098	-.057
PREDICT	-.057	.893	.195	.136	.116
CONSTAN	-.188	.945	.125	-.027	-.006
SEASON	.298	-.708	-.051	.266	.238
JDMin	.198	.105	.206	.608	.290
JDMax	.102	.035	.168	.147	.718

Classification

Classification of Reference flow metrics identified six RFCs (Table 3.6). Classification uncertainty (one minus the probability of class membership, Figure 3.2) was zero for all IQQM nodes except Laidley Creek at 301 (uncertainty 0.025), Mary at Miva (uncertainty <0.005), Tinana Creek at Tagigan Road (uncertainty <0.005) and Munna Creek at Marodian (uncertainty <0.005).

RFC 1 consisted of 26 nodes from all river basins in the study area. Approximately one third of the nodes in RFC 1 were from the Mary catchment, including tributary nodes and nodes from the upper and middle regions of the main channel (i.e. upstream of Dagun Pocket). The Noosa and Maroochy catchments were represented by single nodes (Kin Kin Creek at Mouth and Mooloolah River at Addlington Creek, respectively). All nodes from the Pine–Caboolture catchment (five nodes) were allocated to RFC 1.

The Brisbane catchment was represented by two nodes (Warrill and Laidley Creeks) and the Logan–Albert catchment by all Albert River nodes, the Canungra Creek node, and one upper Logan River node (Logan River at Forest Home). The South Coast catchment was represented by two nodes (Coomera River and Tallebudgera Creek).

RFC 2 consisted of 17 nodes from the Mary, Brisbane and Logan–Albert catchments (Table 3.6). Several of the nodes in this group are now influenced by flow regulation. The Mary catchment nodes in this flow class were located on tributaries. Two of the Mary catchment nodes were located on Yabba Creek, now regulated by Borumba Dam. Brisbane catchment nodes included Bremer River and Buaraba, Laidley, Reynolds and Warrill Creeks.

Of the Brisbane catchment nodes in this class, Reynolds Creek is now regulated by Moogerah Dam and Buaraba Creek is part of an interbasin transfer scheme with Atkinson Dam, off-stream storage in the Brisbane River catchment. Weirs also now exist on Warrill and Laidley Creeks in the Brisbane River catchment.

Figure 3.2: Plot of classification uncertainty (1–probability of class membership) for the Reference flow classification

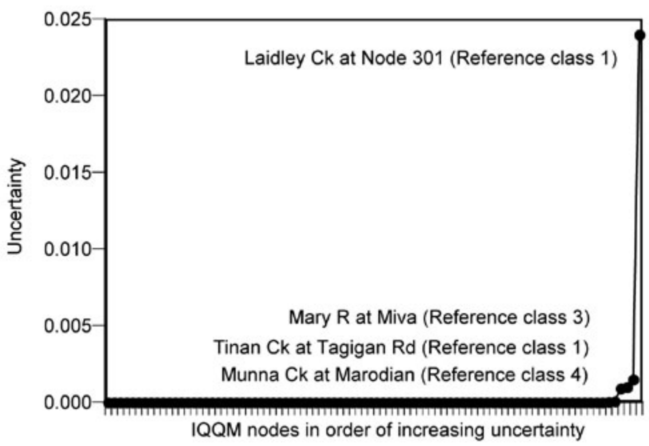


Table 3.6: Membership of 88 IQQM nodes in RFCs

Continental flow class membership (Kennard et al. 2010a) for individual IQQM nodes is shown in brackets. '?' indicates assumed continental flow class based on adjacent IQQM nodes.

Reference class 1 (n=26) – Intermittent Unpredictable	Reference class 2 (n=17) – Intermittent	Reference class 3 (n=5) – Perennial
Amamoor Creek at Zachariah Lane (7)	Glastonbury Creek at Glastonbury (7)	Mary River at Miva (7)
Kandanga Creek at Hygait (7)	Tinana Creek at Bauple East	Teewah Creek at Coops Corner (1)
Mary River at Bellbird Creek (7)	Yabba Creek at Borumba Dam	Christmas Creek at Rudds Lane (4)
Mary River at Dagun Pocket (7?)	Yabba Creek at Imbil	Logan River at Round Mountain (4)
Mary River at Fishermans Pocket (7?)	Bremer River at Adams Bridge (11)	Logan River at Teviot Brook junction
Mary River at Kenilworth (7?)	Bremer River at Walloon (11)	
Mary River at Moy Pocket (7?)	Buaraba Creek at 15.8km	
Obi Obi Creek at Kidaman	Buaraba Creek at Vineyards	
Six Mile Creek at Cooran	Buaraba Creek at Lockyer Creek junction	
Tinana Creek at Tagigan Road (7)	Laidley Creek at Lockyer Creek junction	
Kin Kin Creek at Mouth	Reynolds Creek at Moogerah Dam tailwater	
Mooloolah River at Addlington Creek	Warrill Creek at Kalbar	
Caboolture River at Upper Caboolture (7)	Teviot Brook at The Overflow (7)	
North Pine River at North Pine Dam Tailwater	Burnett Creek at Maroon Dam tailwater	
North Pine River at Youngs Crossing	Logan River at Rathdowney	
North Pine River at Laceys Creek	Logan River at Yarrahappini	
South Pine River at Drapers Crossing (7)	Teviot Brook at Logan River junction	
Warrill Creek at Toohills Crossing		
Laidley Creek at Node 301		
Albert River at Bromfleet (7)		
Albert River at Glendower		
Albert River at Lumeah No. 2 (7?)		
Canungra Creek at Main Road Bridge (4)		
Logan River at Forest Home (7)		

Table 3.6 *(continued)*: Membership of 88 IQQM nodes in RFCs

Continental flow class membership (Kennard et al. 2010a) for individual IQQM nodes is shown in brackets. '?' indicates assumed continental flow class based on adjacent IQQM nodes.

Reference class 4 (n=17) – Intermittent	Reference class 5 (n=18) – Intermittent	Reference class 6 (n=5) – Perennial
Coomera River at Maybury (7)		
Tallebudgera Creek at Tallebudgera Creek Road		
Munna Creek at Marodian (7)	Obi Obi Creek at Baroon Pocket	Obi Obi Creek at Gardners Falls
Wide Bay Creek at Brooyar (7)	Six Mile Creek at Cooroy	South Maroochy River at Kureelpa
Wide Bay Creek at Kilkivan (7)	Eudlo Creek at Kiels Mountain	Stanley River at Woodford Weir inflows
Brisbane River at Avoca Vale (7)	Mooloolah River at Mooloolah (7)	Logan River at Bromelton Rocks
Brisbane River at Gregors Crossing	North Maroochy River at Eumundi (7)	Mudgeeraba Creek at Nerang
Brisbane River at Linville	Petrie Creek at Warana Bridge	
Brisbane River at Savages Crossing	Rocky Creek at Cooloolabin Dam	
Brisbane River at Watts Bridge	South Maroochy River at Kiamba	
Brisbane River at Wivenhoe Dam	South Maroochy River at Wappa Dam	
Cressbrook Creek at Cressbrook Dam	Stanley River at Peachester (7)	
Cressbrook Creek at Rosentretters Crossing	Running Creek at Deickmanns Bridge (4)	
Cressbrook Creek outflows to Brisbane River	Back Creek at Beechmont (4)	
Emu Creek at Boat Mountain (11)	Currumbin Creek at Nicholls Bridge (7?)	
Emu Creek outflows to Brisbane River	Little Nerang Creek at 4km	
Lockyer Creek at Helidon	Little Nerang Ck at Neranwood	
Lockyer Creek at Rifle Range Road	Mudgeeraba Creek at Springbrook Road	
Murphy Creek at Alice Creek junction	Nerang River at Glenhurst	
	Nerang River at Whipbird	

RFC 3 included five nodes from three catchments. Three nodes were located in the Logan–Albert catchment (Christmas Creek and two Logan River nodes) and the remaining nodes were located on the lower Mary River (Miva) and Teewah Creek (Noosa catchment). RFC 4 consisted of 17 IQQM nodes from the Mary and Brisbane catchments. The Mary catchment nodes included three IQQM nodes from the north-western part of the catchment (Wide Bay and Munna Creeks, see Figure 3.1).

The Brisbane catchment nodes included all main channel Brisbane River nodes, all Cressbrook Creek nodes, Emu Creek nodes and nodes from Lockyer Creek (including Murphy Creek, the headwaters of Lockyer Creek). The flow regime characteristics of nodes allocated to this flow class suggests that the group may be characterised by high coefficient of variation of mean daily flows and frequent spells of low to zero discharge, especially for the Mary catchment nodes, and the Brisbane River at Avoca Vale, Linville and Gregors Crossing (Mackay 2007).

RFC 5 consisted of 18 IQQM nodes from the Mary, Maroochy, Brisbane and Gold Coast catchments (Table 3.6). Seven of the nine Maroochy catchment nodes in the Reference flow classification were allocated to RFC 5. The only Brisbane catchment node was the Stanley River at Peachester.

The only Logan catchment node was Running Creek at Deickmanns Bridge. Seven Gold Coast nodes were allocated to this flow class, including two systems now influenced by dams (Nerang River and Little Nerang Creek). RFC 6 consisted of five nodes from five different catchments. Three of these nodes (South Maroochy River, Stanley River and Obi Obi Creek) have headwaters on the Maleny plateau and are potentially influenced by similar weather patterns.

Flow metrics discriminating Reference flow classes

Clustvarsel identified six flow metrics that best discriminated between the RFCs (Figure 3.3). These were MDF_Mar, MDF_Sep, MA1dayMin, MA1dayMax, MedAnnmax and MeanZeroDays. All flow metrics identified by clustvarsel were associated with discharge magnitude. Ordination of Reference flow metrics (Figure 3.4) also shows the prominence of the discharge magnitude gradient. Associated with the discharge magnitude gradient is a flow variability gradient, as shown by CVDaily (Figure 3.4).

RFCs 4 and 6 represent the extremes of the discharge magnitude gradient (Figures 3.3 and 3.4), with RFC 4 representing relatively low discharge magnitude per unit catchment area and RFC 6 high discharge magnitude (Figure 3.4). RFC 6 is widely scattered in ordination space when compared with the other flow classes, although forming a discrete group in this space. RFCs 2 and 4 included nodes with a high mean number of zero flow days per year (mean of approximately 20 and 60 days respectively). RFC 6 and Teewah Creek (RFC 3) had relatively low values for CVDaily (Figure 3.4).

A second gradient, representing spell number and duration and baseflow, is also present but this gradient is small compared with the discharge magnitude gradient. With the exception of flow class 5 all RFCs were arranged perpendicular to this gradient. Teewah Creek is unusual in having a high baseflow index and low numbers of high and low flow spells, suggesting a relatively stable flow regime influenced by groundwater (Brizga et al. 2005b).

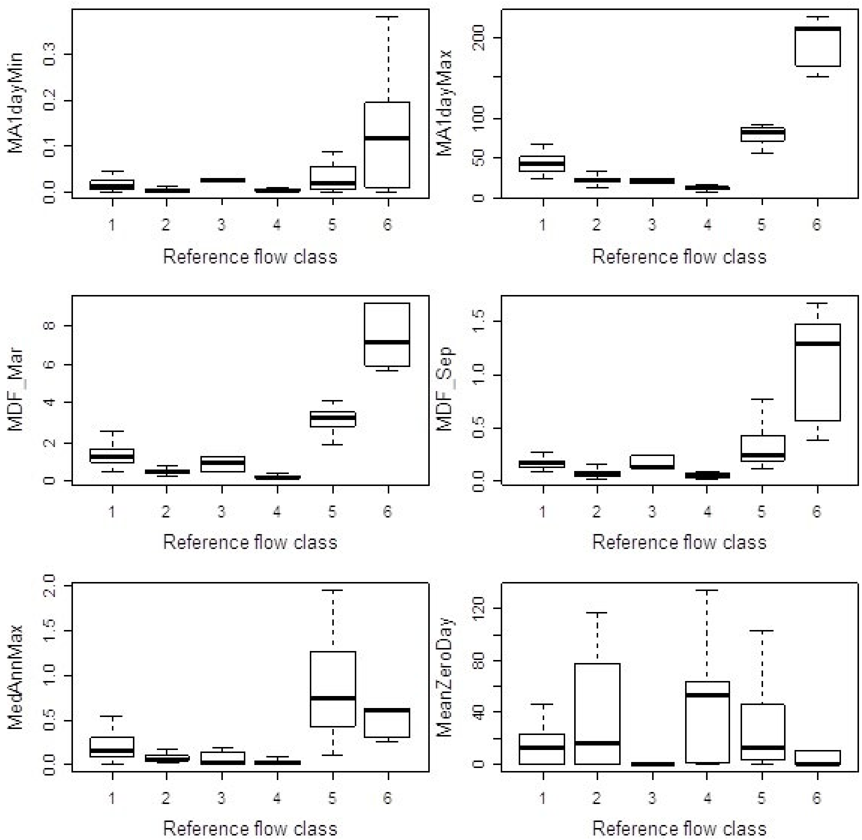
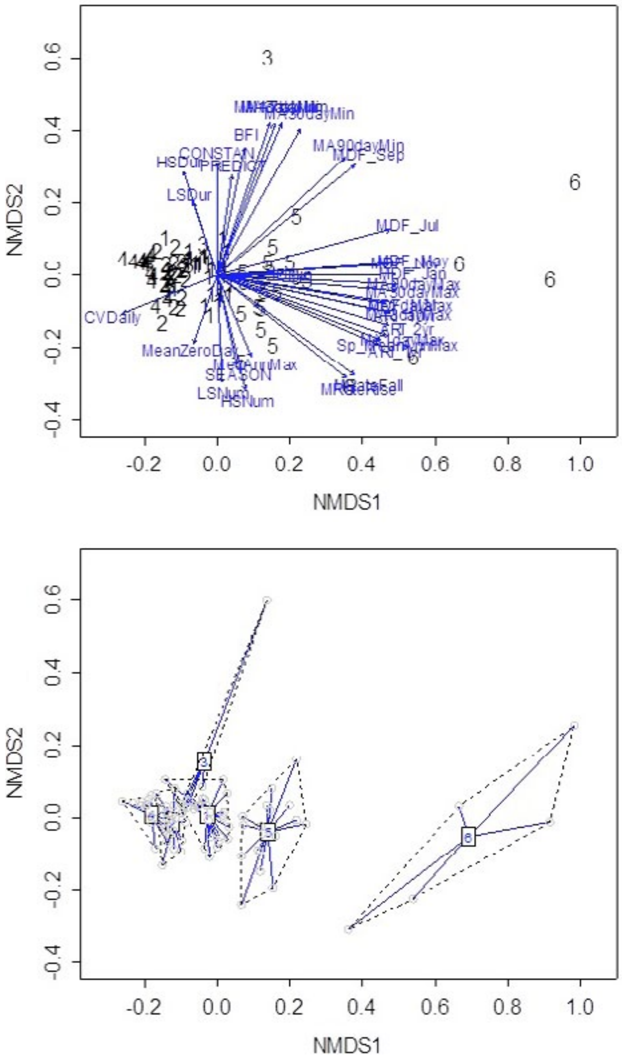


Figure 3.3: Box and whisker plots of flow metrics selected by the clustvarsel function as best discriminating between the Reference flow classes

The box represents the upper and lower quartiles, the line within the box is the median, and the whiskers represent the data range. See Table 3.3 for metric acronyms.

Figure 3.4: Two-dimensional nMDS ordination of IQQM nodes based on Reference flow metrics

Stress = 0.073. (a) Distribution of IQQM nodes in ordination space as shown by Reference flow class membership. Vectors are flow metrics significantly correlated with the ordination space (p<0.01). (b) Distance of individual IQQM nodes to flow class centroids (numbered).



The two major flow regime types identified by Kennard et al. (2010a) for the continental flow classification were perennial and intermittent flow regime classes. It is difficult to align the RFCs along a gradient of flow perenniality–intermittency as the principal discharge gradient in the Reference flow classification is discharge magnitude. Of the RFCs identified, RFC 3 can be defined as perennial as only one node ceased to flow (Christmas Creek, mean of 3.9 zero flow days per year). In addition, three of the nodes in RFC 3 (Teewah Creek, Christmas Creek and Logan River at Round Mountain) were classified as perennial by Kennard et al. (2010a).

RFC 6 is also defined as perennial, although one IQQM node in this class (South Maroochy River at Kureelpa) had a MeanZero Day of 103 days per year. The remaining IQQM nodes in this class had MeanZeroDay values ranging from 0–10.8 days per year. None of the IQQM nodes in this class had an equivalent stream gauge in the continental classification.

The remaining RFCs are defined as intermittent although all RFCs contained nodes that never ceased to flow. RFCs 1, 2, 4, and 5 are highly variable in terms of MeanZeroDay. Of the intermittent classes of Kennard et al. (2010a) it is unlikely that any of the IQQM nodes would be considered Extremely Intermittent (continental flow class 12) as this flow class tended to occur in Drainage Division 4 (Murray–Darling Division). RFC 1 is dominated by IQQM nodes belonging to continental flow class 7 and therefore this flow class is designated as *unpredictable intermittent*.

The IQQM nodes in RFCs 2, 4 and 5 are poorly represented in the continental classification but most IQQM nodes in these classes with equivalent stream gauges in the continental classification tend to belong to continental flow class 7. MeanZeroDay values within each of these flow classes are highly variable although HFCs 2 and 5 have similar distributions for this metric (Figure 3.3). Due to the difficulty in aligning these flow classes with the continental flow classification these classes are designated as intermittent only.

Reference classification validation

The Reference random forests model (all flow metrics included) correctly allocated IQQM nodes to two of the six Mclust-defined RFCs (Table 3.7). The OOB estimate of the error rate was 7.95% (seven nodes not correctly allocated) but the error rate for RFC 3 was 20% (Table 3.7).

Seven nodes were incorrectly allocated to RFCs by the random forests model. These were Tinana Creek at Tagigan Road (allocated to RFC 2); Tinana Creek at Bauple East (allocated to class 4); Yabba Creek at Imbil (allocated to RFC 1); Laidley Creek at node 301 (allocated to RFC 2); Burnett Creek at Maroon Dam tailwater (allocated to RFC 3); Teewah Creek at Coops Corner (allocated to RFC 1) and Munna Creek at Marodian (allocated to RFC 2).

Three of the misclassified nodes (Tinana Creek at Tagigan Road, Laidley Creek at node 301 and Munna Creek at Marodian) were identified as having relatively high classification uncertainty (Figure 3.2).

Table 3.7: Confusion matrix for the Reference random forests model using 35 Reference flow metrics to allocate IQQM nodes to Mclust-defined RFCs

OOB estimate of error rate 7.95%. Number of flow metrics used at each split = 5. Total n = 88.

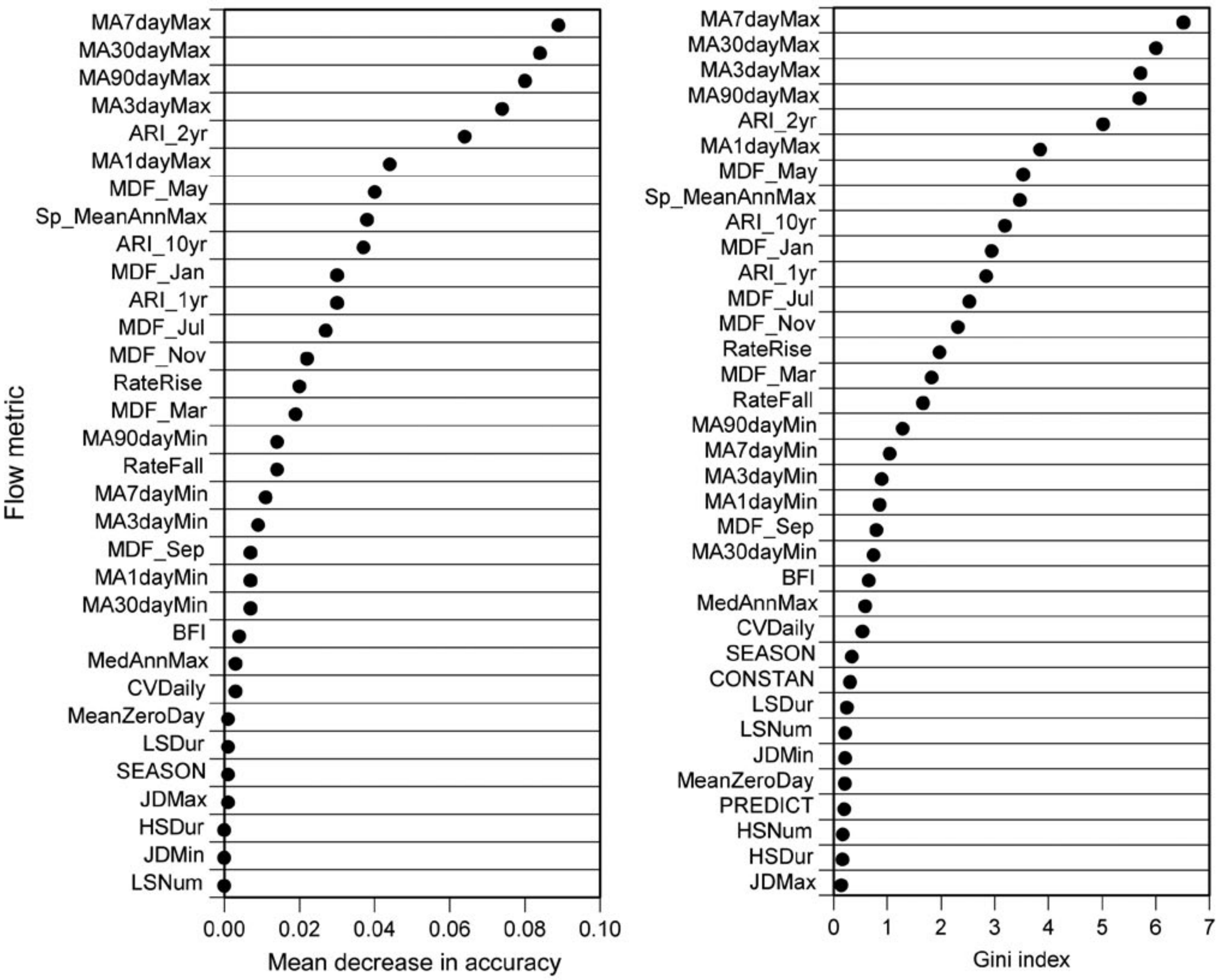
Mclust-defined flow class	RFC predicted by random forests						Error (%)
	1	2	3	4	5	6	
1 (*n* = 26)	24	2	0	0	0	0	7.7
2 (*n* = 17)	1	14	1	1	0	0	17.6
3 (*n* = 5)	1	0	4	0	0	0	20.0
4 (*n* = 17)	0	1	0	16	0	0	5.9
5 (*n* =18)	0	0	0	0	18	0	0
6 (*n* = 5)	0	0	0	0	0	5	0

Variable importance for the Reference random forests model identified flow metrics related to discharge maxima as being important in discriminating between RFCs, in contrast to *clustvarsel* which identified discharge maxima and minima as important in discriminating between flow classes (Figure 3.5).

The five most important metrics identified by the random forests model (using mean squared error and Gini index as importance criteria) were the moving averages of the 1, 3, 7, 30 and 90 day maximum discharge (MA1-90dayMax) and the 2-year ARI (ARI_2yr). The metrics selected by *clustvarsel* are relatively unimportant using the mean decrease in accuracy and Gini index as selection criteria (compare Figures 3.3 and 3.5). Thus the random forests model identified metrics describing discharge maxima as more important than discharge minima, whereas clustvarsel selected metrics describing discharge minima as being more important to flow class discrimination.

Figure 3.5: Variable importance for the Reference random forests model based on 35 Reference flow metrics and 1000 trees

See Table 3.3 for flow metric acronyms.



3.3.2 Historic flow classification

PCA and metric redundancy

Preliminary PCA indicated that Oxley Creek at New Beith was an outlier in terms of low spell duration (a single low flow spell equivalent to the length of the flow record used to calculate metrics) and this gauge was subsequently excluded from the Historic flow classification.

PCA of Historic flow metrics extracted six components with eigenvalues greater than 1, explaining 89.8% of the variation in the dataset (Table 3.8). However, only one metric (JDMin) loaded on component 5 and only two metrics (JDMax and CVDaily) loaded on component 6. These components were therefore considered unreliable (Tabachnik and Fidell 1989) and not included in the Historic flow classification.

The first two components of the Historic metric PCA were associated with high and low discharge magnitude. Component 3 was associated with high and low spell frequency and duration and component 4 was associated with Colwells measures of discharge predictability, constancy and seasonality (Table 3.8). The results of the principal components analyses for Reference and Historic flow metrics were broadly similar (compare Tables 3.5 and 3.8). Most components of the Reference PCA had a similar component in the Historic PCA, although the ordering of components varied.

Table 3.8: Rotated component matrix of metric loadings from PCA of Historic flow metrics

Percentage variation explained by each component shown in brackets. See Table 3.3 for metric acronyms. Loadings ≤-0.5 and ≥0.5 shown in bold text.

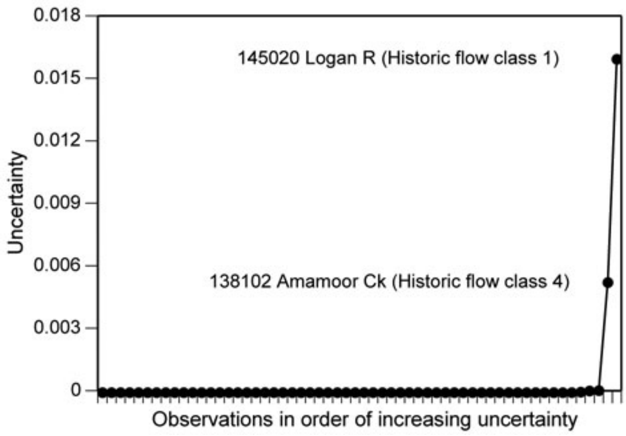
Flow metrics	Component and Variance Explained					
	1 (39.7%)	2 (22.3%)	3 (10.6%)	4 (8.0%)	5 (4.8%)	6 (4.4%)
MDF_Jan	**.761**	.343	.238	-.212	-.021	.116
MDF_Mar	**.880**	.189	.173	-.162	-.194	.034
MDF_May	**.855**	.314	.168	-.013	-.076	-.020
MDF_Jul	**.581**	**.720**	.168	-.124	-.113	.110
MDF_Sep	.406	**.853**	.180	-.022	-.138	.136
MDF_Nov	**.580**	**.561**	.335	-.132	.167	.293
ARI_1yr	**.868**	.054	.250	-.165	.149	.137
ARI_2yr	**.952**	-.008	.098	-.139	.039	.006
ARI_10yr	**.937**	-.054	-.031	-.054	-.045	-.143
MedAnnMax	**.561**	-.027	.178	-.010	.439	.300
MA1dayMin	-.029	.970	.030	.178	.081	.012
MA3dayMin	-.027	.970	.037	.183	.074	.012
MA7dayMin	-.023	.972	.049	.186	.062	.013
MA30dayMin	.010	.976	.082	.172	.022	.028
MA90dayMin	.140	.966	.142	.108	-.035	.062
MA1dayMax	.977	-.023	.075	-.100	-.014	-.077
MA3dayMax	.972	.000	.115	-.130	-.023	-.061
MA7dayMax	.967	.030	.135	-.127	-.059	-.049
MA30dayMax	.952	.114	.165	-.164	-.109	-.008
MA90dayMax	.924	.217	.187	-.181	-.130	.036
RateRise	.963	-.069	-.023	.015	.074	.018
RateFall	.940	-.055	.077	.019	.057	.071
CVDaily	-.082	-.428	-.395	.199	.431	-.521
BFI	-.180	.706	.260	.080	-.469	.218
MeanZeroDay	-.035	-.301	-.676	.403	.379	.125
Sp_ MeanAnnMax	.977	-.025	.072	-.100	-.016	-.080
LSNum	.146	.093	.804	.227	-.235	.296
LSDur	-.162	-.194	-.813	.220	.297	-.021
HSNum	.405	.111	.729	.147	.297	.169
HSDur	-.272	-.104	-.833	-.019	-.162	.066
PREDICT	-.187	.375	.091	.861	.091	.014
CONSTAN	-.306	.383	.046	.851	.011	-.022
SEASON	.479	-.191	.219	-.755	.058	.071
JDMin	.152	-.078	.098	-.022	-.700	.030
JDMax	-.120	.155	.063	-.003	.013	.880

Historic flow classification

Classification identified five HFCs (Table 3.8). Fifty-six of the 58 gauges used in the classification had a probability of class membership of 1.0 (Figure 3.6). The remaining two gauges had a probability of class membership greater than 0.985 (Figure 3.6). Comparison of the Reference and Historic flow classifications (Tables 3.6 and 3.8) shows several similarities in classification structure despite differences in the number of nodes/gauges used in each classification and the number of classes identified.

Each classification includes a class containing sites in drier parts of the study area (RFC 2 and HFC 4) and a class containing mostly Maroochy catchment, South Coast and Stanley River sites (RFC and HFC 5). RFC 1 resembles HFC 3 in containing mostly Mary and Logan–Albert catchment sites. RFCs 2 and 6 have no directly comparable classes in the Historic flow classification.

Figure 3.6: Plot of classification uncertainty (1–probability of class membership) for the Historic flow regime classification



HFC 1 consisted of six gauges located in Teewah Creek, Burnett Creek downstream of Maroon Dam, the Brisbane River downstream of Wivenhoe Dam and the Logan River downstream of the Burnett Creek confluence (Table 3.9).

Thus the flow regimes of five of the six gauges in this flow class were influenced directly or indirectly by dams. HFC 2 consisted of 14 gauges located mostly in the Mary and upper Brisbane River catchments (Table 3.9). A single gauge (145012 Teviot Brook) was located in the Logan River catchment. This class roughly corresponded to RFC 4, that is gauges located in the drier parts of the study area (Table 3.6).

Table 3.9: Historic flow class membership for 58 gauges as determined by classification of Historic flow metrics using model-based clustering

Gauges directly influenced by dams and weirs shown in bold text. Continental flow class membership (Kennard et al. 2010a) for individual gauges shown in brackets. '?' indicates assumed continental flow class membership based on adjacent gauges.

Class 1 (n=6) (Perennial)	Class 2 (n=13) (Highly Intermittent–Unpredictable Summer)	Class 3 (n=18) (Rarely Intermittent–Unpredictable)
140002 Teewah Creek at Coops Corner (1)	138002 Wide Bay Creek at Brooyar (7)	138001 Mary River at Miva (7)
143001 Brisbane River at Savages Crossing	138004 Munna Creek at Marodian (7)	138003 Glastonbury Creek at Glastonbury (7)
143035 Brisbane River at Wivenhoe Dam tailwater	138010 Wide Bay Creek at Kilkivan (7)	138007 Mary River at Fishermans Pocket (7?)
145008 Logan River at Round Mountain (4)	138119 Yabba Creek at Borumba Dam release	138009 Tinana Creek at Tagigan Road (7)
145020 Logan River at Rathdowney	143007 Brisbane River at Linville	138014 Mary River at Home Park (7)
145099 Burnett Creek at Maroon Dam tailwater	143009 Brisbane River at Gregors Crossing	138109 Mary River at Dagun Pocket (7?)
	143010 Emu Creek at Boat Mountain (11)	138111 Mary River at Moy Pocket (7?)
	143015 Cooyar Creek at dam site	138903 Tinana Creek at Bauple East
	143107 Bremer River at Walloon (11)	143108 Warrill Creek at Amberley
	143110 Bremer River at Adams Bridge (11)	145003 Logan River at Forest Home (7)
	143210 Lockyer Creek at Rifle Range Road	145010 Running Creek at Deickmann Bridge (4)
	143306 Reedy Creek upstream of Byron Creek (11)	145011 Teviot Brook at Croftby (7)
	145012 Teviot Brook at The Overflow (7)	145014 Logan River at Yarrahappini
		145018 Burnett Creek upstream of Maroon Dam (7)
		145101 Albert River at Lumeah No. 2 (7)
		145102 Albert River at Bromfleet (7)
		145107 Canungra Creek at Main Road Bridge (4)
		146002 Nerang River at Glenhurst

Table 3.9: *(continued)* HFC membership for 58 gauges as determined by classification of Historic flow metrics using model-based clustering

Gauges directly influenced by dams and weirs shown in bold text. Continental flow class membership (Kennard et al. 2010a) for individual gauges shown in brackets. '?' indicates assumed continental flow class membership based on adjacent gauges.

Class 4 (n=12) – Intermittent–Unpredictable	Class 5 (n=9) – Rarely Intermittent–Unpredictable
138102 Amamoor Creek at Zachariah Lane (7)	141001 South Maroochy River at Kiamba
138107 Six Mile Creek at Cooran	141003 Petrie Creek at Warana Bridge
138110 Mary River at Bellbird Creek (7)	141004 South Maroochy River at Yandina
138113 Kandanga Creek at Hygait (7)	141006 Mooloolah River at Mooloolah (7)
142001 Caboolture River at upper Caboolture (7)	141008 Eudlo Creek at Kiels Mountain
142202 South Pine River at Drapers Crossing (7)	141009 North Maroochy River at Eumundi (7)
143028 Ithaca Creek at Jason Street	143303 Stanley River at Peachester (7)
143032 Moggill Creek at upper Brookfield	146012 Currumbin Creek at Nicholls Bridge
143307 Byron Creek at causeway	146014 Back Creek at Beechmont (4)
145103 Cainbable Creek at dam site (7)	
146010 Coomera River at Maybury (7)	
146020 Mudgeeraba Creek at Springbrook Road	

HFC 2 included four gauges influenced to varying extents by flow regime alteration. However only gauge (138119 Yabba Creek) is directly influenced by an upstream dam. HFC 3 consisted of 19 gauges located mostly in the Mary and Logan–Albert River catchments. The flow regimes of three gauges in this class are influenced by flow regime alteration, but only 146002 Nerang River has a dam (Hinze Dam) immediately upstream of the gauge.

HFC 4 consisted of 12 gauges from five different catchments. The flow regime of one gauge (138107 Six Mile Creek) was influenced by flow regime alteration (Six Mile Creek Dam). HFC 5 consisted of nine gauges and was broadly similar to RFC 5 in terms of composition. HFC 5 included gauges from three catchments but was dominated by gauges from the Maroochy River catchment (six gauges). The flow regimes of two gauges in this class are influenced by flow regime alteration (Table 3.9).

Flow metrics discriminating Historic flow classes

The clustvarsel function identified six flow metrics that best discriminated between the HFCs (Figure 3.7). These flow metrics described discharge magnitude, the timing of high and low spells and discharge variability (CONSTAN). Ordination of gauges suggests that the principal gradient is high discharge magnitude and the secondary gradient is related to spell duration and low discharge magnitude (Figure 3.8).

HFC 5 is on the positive extreme of the high magnitude discharge gradient (i.e. high values for high discharge magnitude metrics) and HFCs 1 and 2 are on the negative extreme of this gradient (i.e. low values for high discharge metrics, Figure 3.8b).

Gauges in HFC 1 had relatively high values for MA1dayMin and CONSTAN but relatively low values for ARI_1yr and ARI_10yr. This may be due to the influence of flow regime alteration on five of the six gauges in this flow class. Water is released from Wivenhoe Dam (Brisbane River) at a high constant rate to supply water to Mt Crosby weir (Arthington et al. 2000). Irrigation releases are made from Maroon Dam and hence Burnett Creek and Logan River gauges would be subjected to elevated flows.

Teewah Creek is the only gauge in this flow class not subjected to flow alteration but has a relatively high baseflow component (Brizga et al. 2005b). The Logan River gauges in HFC 1 are downstream of the Burnett Creek confluence and are indirectly subjected to flow alteration by Maroon Dam on Burnett Creek. The Logan River at Rathdowney (145020) had the lowest probability of class membership (Figure 3.6) and is upstream of the Running Creek confluence, whereas 145008 (Logan River at Round Mountain) is downstream of the Running Creek confluence.

Although the Logan River at Round Mountain was not included in the classification of Kennard et al. (2010b), it can be assumed that since Running Creek is perennial (continental flow class 4) that gauge 145008 is also perennial and belonging to Kennard flow class 4. Teewah Creek is an outlier in HFC 1, especially for the very low CVDaily (approximately 125%).

CONSTAN is highly variable in this flow class. Teewah Creek and Brisbane River have high CONSTAN (approximately 0.5) and Logan River and Burnett Creek low constancy (0.1). Predictability (PREDICT) also follows a similar trend. HFC 1 is designated as *Perennial*.

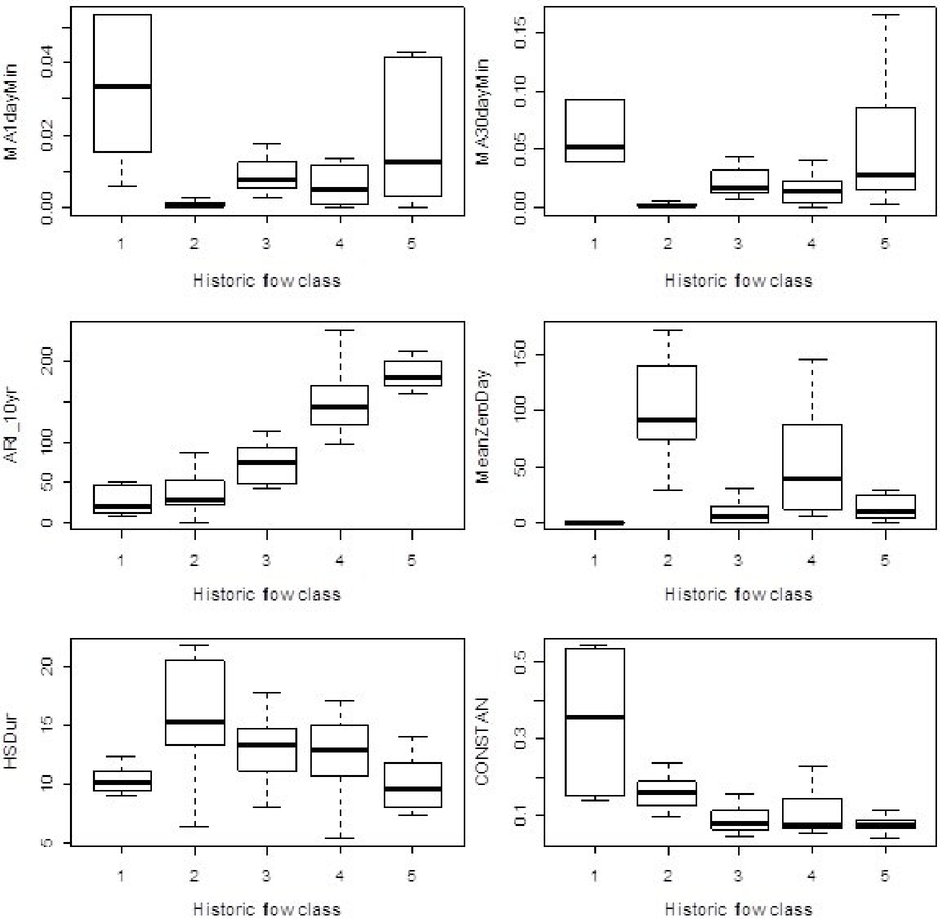
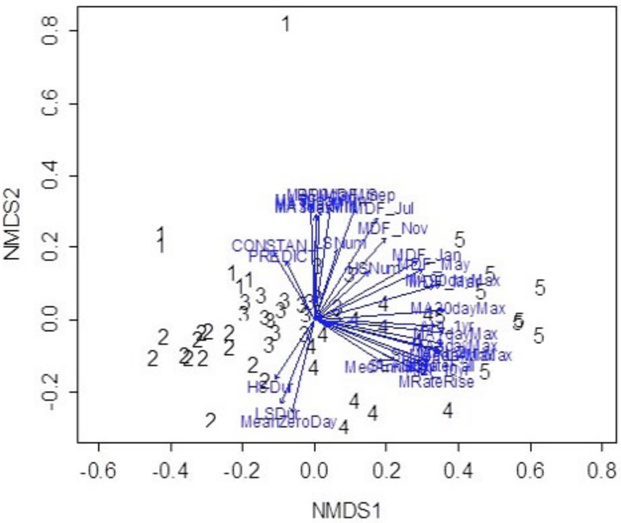


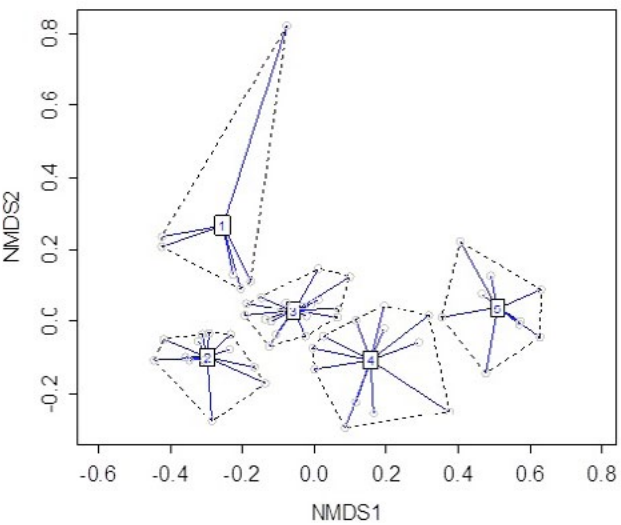
Figure 3.7: Box and whisker plot of important metrics identified by *clustvarsel* as distinguishing between HFCs

The box represents the upper and lower quartiles, the line within the box is the median, and the whiskers represent the data range. See Table 3.3 for flow metric acronyms.

Figure 3.8: Non-metric multidimensional scaling (nMDS) of gauges based on Historic flow metrics. Stress = 0.060, two dimensions

(a) Distribution of flow classes in ordination space. Vectors represent flow metrics significantly correlated with the ordination space (p<0.01). See Table 3.3 for variable acronyms. (b) Group centroids for each of the five flow classes and distance to centroid for each gauge.





The majority of gauges in HFC 2 belong to continental flow classes 7 (*Intermittent–Unpredictable*) and 11 (*Highly Intermittent–Unpredictable Summer Dominated*) (Table 3.9). HFC 2 was characterised by long periods of low discharge (median value for LSDur 50 days, median value for MeanZeroDay 100 days) and had highly variable discharge regimes (range for CVDaily 509%–1625%). The gauges in HFC 2 have very low values for low discharge magnitude metrics (MA1dayMin and MA90dayMin in Figure 3.7). HFC 2 corresponds to continental flow class 11 on the basis of MeanZeroDay and this class is designated as *Highly Intermittent–Unpredictable Summer*.

The majority of gauges in HFCs 3 and 4 belong to continental flow class 7 and therefore could be designated as *Intermittent–Unpredictable*. HFC 3 gauges have far fewer days of zero discharge (median value for MeanZeroDay approximately 10 days per year) and have shorter low flow spells (median value for LSDur approximately 15 days) compared with HFC 2.

Although the majority of gauges in this flow class correspond to continental flow class 7 it is difficult to align this flow class with continental flow class 7 on the basis of the relatively low value for MeanZeroDay (median value 10 days per year), and relatively high value for MA1dayMin (median approximately 0.01 $Mlday^{-1}.km^{-2}$) (Figure 3.7). HFC 3 is designated as *Rarely Intermittent – Unpredictable*. HFC 4 was highly variable for MedAnnMax, MeanZeroDay but was similar to HFC 3 in terms of MA1 and 90dayMin, HSDur and CONSTAN (Figure 3.7). Due to the higher number of mean zero days for HFC 4 (compared with HFC 3) this class is designated *Intermittent–Unpredictable*.

HFC 5 included gauges with high discharge (and flood) magnitude per unit catchment area and low discharge constancy (Figures 3.7 and 3.8). This flow class was associated with low CONSTAN (Figure 3.7) but relatively high discharge magnitude per unit of catchment area. Low flow spells were of low duration and the mean number of zero flow days per year was low. Discharge tended to peak in autumn. This flow class included members of continental flow classes 4 and 7 but five gauges in the class were not included in the continental flow classification. HFC 5 is designated as *Rarely Intermittent–Unpredictable*.

Historic flow classification validation

The random forests model based on the 33 flow metrics used for the model-based classification had an OOB estimate of the overall classification error rate of 8.5% (Table 3.10). Gauges were allocated to two of the five Mclust-defined flow classes without error. HFC 1 had the highest overall misclassification rate (33.3%). The following gauges were misclassified by the random forests model:

- 140002 Teewah Creek and 145020 Logan River at Rathdowney (HFC 1) were allocated to HFC 3 by the random forests model
- 138111 Mary River at Moy Pocket (HFC 3) was allocated to HFC 4 by the random forests model
- 145103 Cainbable Creek and 146020 Mudgeeraba Creek (HFC 4) were allocated to HFCs 3 and 5 respectively by the random forests model.

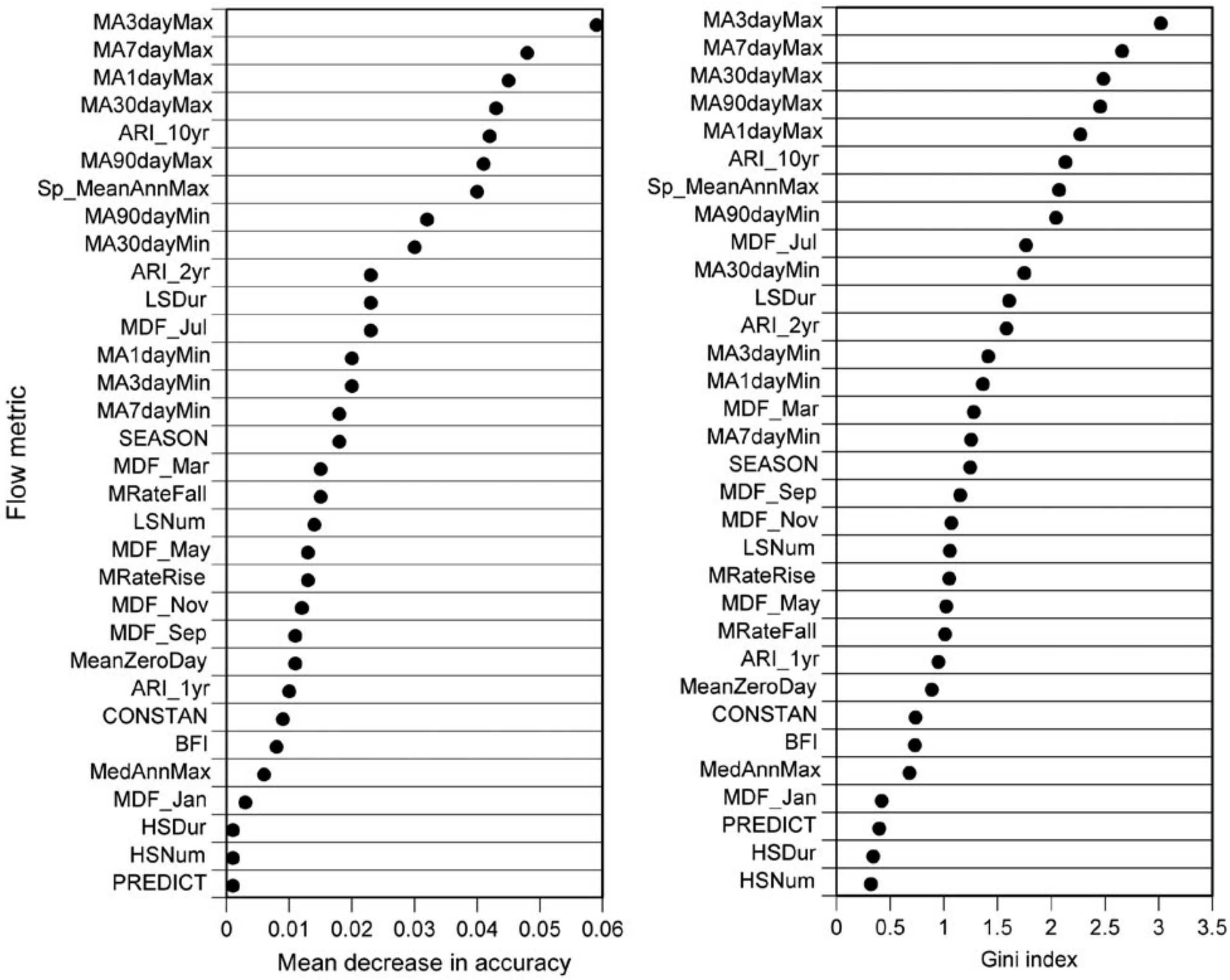
Variable importance as determined by the Historic random forests model indicated flow metrics associated with discharge maxima were more important than discharge minima in allocating gauges to HFCs (Figure 3.9). This contrasts with the results obtained from *clustvarsel* (Figure 3.7) but is supported by the ordination (Figure 3.8).

Table 3.10: Confusion matrix from the random forests model based using 32 Historic flow metrics to allocate gauges to Mclust-defined Historic flow classes

Out-of-bag (OOB) estimate of error rate 8.6%. Number of metrics tried at each split = 3. Total n = 58.

Mclust-defined flow class	HFC predicted by random forests model					Error (%)
	1	2	3	4	5	
1 (*n* = 6)	4	0	2	0	0	33.3
2 (*n* = 13)	0	13	0	0	0	0
3 (*n* = 18)	0	0	17	1	0	5.6
4 (*n* = 12)	0	0	1	10	1	16.7
5 (*n* = 9)	0	0	0	0	9	0

See Table 3.3 for flow metric acronyms.



The Historic random forests model was used to allocate gauges 138104 Obi Obi Creek at Kidaman and 143112 Reynolds Creek at Moogerah Dam tailwater to HFCs. The Obi Obi Creek gauge was allocated to HFC 3 (probability of class membership 0.736) and the Reynolds Creek gauge was allocated to HFC 1 (probability of class membership 0.623) (Table 3.11).

Table 3.11: Probability of HFC membership for gauges 138104 Obi Obi Creek at Kidaman and 143112 Reynolds Creek at Moogerah Dam tailwater as determined by the Historic random forests model

Predicted HFC for each gauge shown in bold text.

Gauge	HFC 1	HFC 2	HFC 3	HFC 4	HFC 5
138104a Obi Obi Creek	0.164	0.079	**0.736**	0.020	0.001
143112a Reynolds Creek	**0.623**	0.158	0.160	0.044	0.015

Comparison of Reference and Historic flow classifications

The adjusted Rand Index was used to compare (a) the Reference and Historic Flow classifications, and (b) the Historic classification with the classification of Kennard et al. (2010a) (Table 3.12). The adjusted Rand Index was 0.382 for the comparison of Reference and Historic flow classifications and 0.124 for the comparison of the Historic flow classification with the continental classification (Kennard et al. 2010a). The Reference classification was more similar to the continental classification than the Historic classification (Table 3.12).

Table 3.12: Comparison of Reference, Historic and continental flow classifications using the adjusted Rand Index

Comparison	Reference flow classification	Historic flow classification
Continental flow classification	0.160	0.124
Reference flow classification	—	0.382

The Reference and Historic flow classifications share a similar classification structure, although the adjusted Rand Index is relatively low. Changes in class membership for study reaches are relatively few (Table 3.13). Study sites in the Stanley River, Eudlo Creek, North Maroochy River, Currumbin Creek, Wide Bay Creek and Munna Creek did not change class membership.

Many study reaches in RFC 2 (Glastonbury Creek, Tinana Creek at Bauple East, Yabba Creek at Borumba Dam, Reynolds Creek, Logan River at Rathdowney) have effectively changed flow class but these changes are hard to describe. Many IQQM nodes in this RFC have changed class membership but have been grouped in the Historic classification with other IQQM nodes in RFC 2 (e.g. Glastonbury Creek, Logan River at Yarrahapinni, Tinana Creek at Bauple East have been grouped in the Historic flow classification).

The gauges that have undergone the greatest change in class membership are those in HFC 1, as this HFC has no analogue in the Reference flow classification (Table 3.13).

Table 3.13: Qualitative comparison of RFC and HFC membership of SEQ study sites

Study reach	RFC	HFC	Comment on flow class change
Stanley River	5	5	No change
Nerang River	5	3	Change from Maroochy catchment group to Mary–Logan group
Coomera River	1	4	Some similarities with Reference classification but split from Logan–Albert River gauges
Amamoor Creek	1	4	Similar to Reference classification but split from Logan–Albert River gauges
Yabba Creek	2	2	Similar to Reference but split from Brisbane River gauges upstream of Wivenhoe Dam and Munna and Wide Bay Creeks
Study reach	RFC	HFC	**Comment on flow class change**
Obi Obi Creek	1	3	In general still grouped with similar sites
Mary River at Moy Pocket	1	3	In general still grouped with similar sites
Six Mile Creek	1	4	Similar to Reference classification but split from Logan–Albert River gauges
Glastonbury Creek	2	3	Has changed association, but other nodes in RFC 2 have changed association
Eudlo Creek	5	5	No change
Reynolds Creek	2	1	Change to a new flow class with no comparable RFC
Burnett Creek downstream of Maroon Dam	2	1	Change to a new flow class with no comparable RFC
Currumbin Creek	5	5	No change
Wide Bay Creek	4	2	No change
Munna Creek	4	2	No change
North Maroochy River	5	5	No change
Mary River at Miva	3	3	Change from a small group of IQQM nodes to a large group including other Mary River gauges, and Logan and Albert River gauges
Tinana Creek at Bauple East	2	3	Changed association, but other IQQM nodes in RFC 2 have changed association
Logan River at Rathdowney	2	1	Change to a new flow class with no comparable RFC

3.3.3 Flow regime alteration within the study area

The Gower metric was used to compare Reference and Historic flow regimes on the basis of the 35 flow metrics used to undertake the Reference and Historic flow classifications (Figure 3.10). These comparisons were limited to IQQM nodes with a corresponding stream gauge (*n* = 48).

All gauges in the study area have been subject to flow regime change (Figure 3.10) based on the flow data available for this study. The extent of flow regime change in the study area is relatively minor (maximum Gower metric is one for total dissimilarity).

In general the greatest flow regime changes in the study area have occurred downstream of dams. However, three of the 11 gauges with the greatest flow regime change are not downstream of dams.

The gauge with the greatest flow regime change (145010 Running Creek) is not subject to flow regime alteration by dams or weirs. Furthermore, the presence of dams does not necessarily imply extensive flow regime change based on the Gower metric (138107 Six Mile Creek and 138104 Obi Obi Creek in Figure 3.10).

As a further measure of the extent of flow regime alteration in the study area the Reference random forests model was used to predict RFC membership for stream gauges. It was assumed that if the flow regime of an individual stream gauge was unmodified then that gauge should be allocated to the same RFC as the corresponding IQQM node. Probabilities of class membership for gauges allocated to the correct RFC varied between 0.598 and 0.995 (Table 3.14).

Eight of the 49 gauges included in this analysis were not allocated to the appropriate RFC (Table 3.14). Two of these gauges (Logan River gauges 145008 and 145020) had relatively high classification uncertainty under the Historic flow classification (Figure 3.6). Three misclassified gauges (138119 Yabba Creek, 145099 Burnett Creek and 146002 Nerang River) are downstream of dams. Despite flow regime alteration by Wivenhoe Dam the two gauges downstream of Wivenhoe Dam on the Brisbane River (143001 and 143005) were allocated to the correct RFC.

Prediction error (1–probability of class membership, where membership is determined by the flow class with the highest probability of membership) varied between flow class and catchment (Figure 3.11). Prediction error was relatively low for HFCs 2 and 5. HFC 2 includes gauges in the drier parts of the study area (i.e. lower rainfall) and HFC 5 includes gauges mostly in the Maroochy catchment (Figure 3.11a).

Prediction error was highest and most variable for HFC 1 (most gauges in the class subjected to flow regime alteration) and class 3 (large number of gauges located across the study area). Prediction error for the Maroochy catchment gauges was relatively low (maximum error of prediction approximately 0.15, Figure 3.11b).

Prediction error for the Logan–Albert, Mary and South Coast catchments was varied, suggesting a range of flow regime changes in these catchments. The Mary and Logan–Albert catchments include four of the five HFCs, and the South Coast catchments include three of the five HFCs (Table 3.8). In contrast, all Maroochy catchment gauges belong to HFC 5.

Figure 3.10: Comparison of Reference and Historic flow regimes using the Gower metric

Higher values for the Gower metric indicate greater divergence of the Historic flow regime from the Reference flow regime. Only IQQM nodes with a currently operating stream gauge shown. Names in bold text indicate gauges where study sites were established.

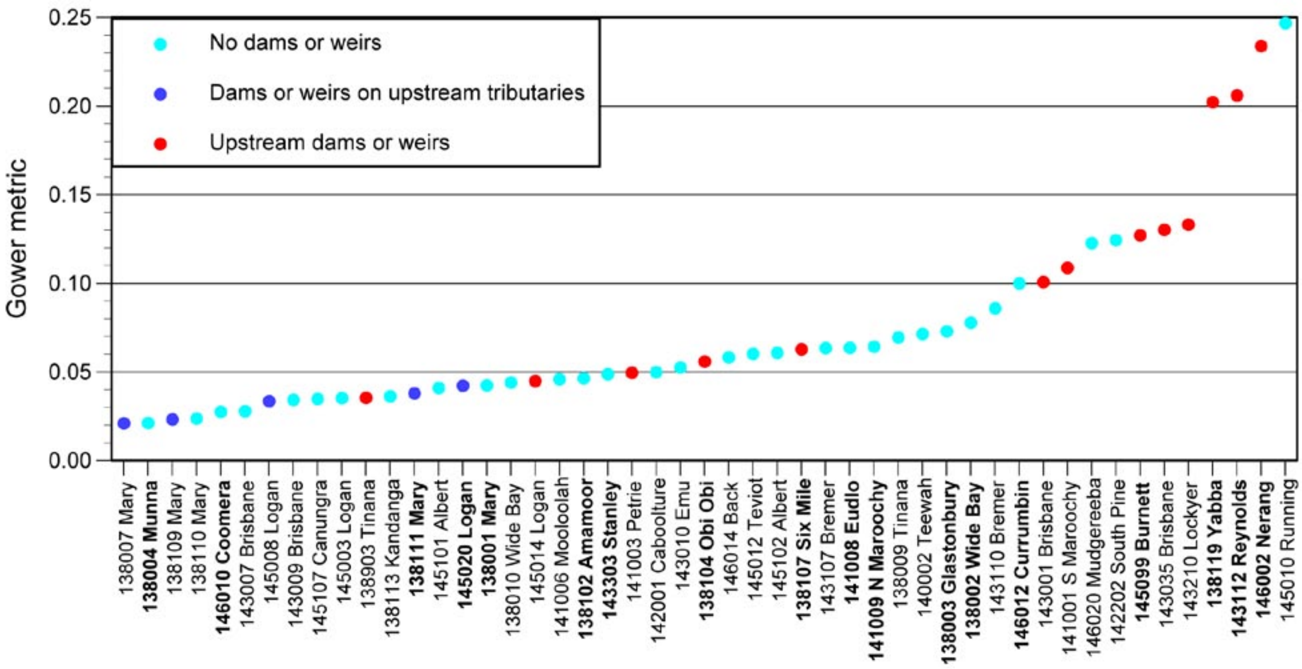


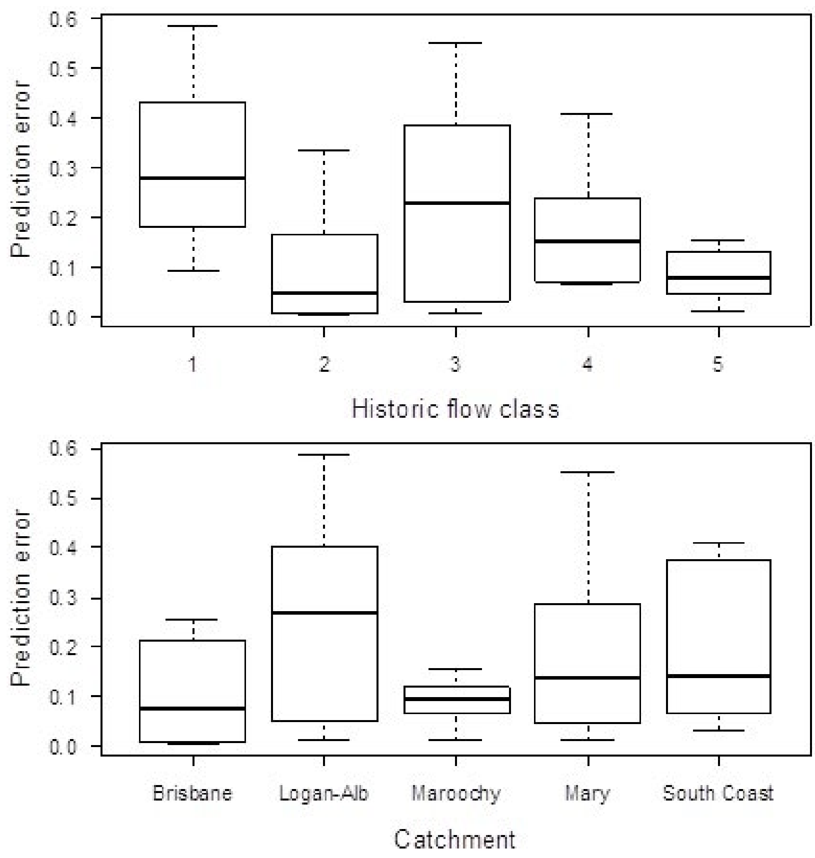
Table 3.14: Probability of RFC membership for gauges in RFCsas determined by the Reference random forests model

For each gauge the predicted RFC membership is shown in bold text. Bold green text indicates correct allocation to a RFC and bold red text indicates erroneous allocation to a RFC (the correct RFC is underlined). Zero probabilities are not shown.

Gauge	Reference flow class					
	1	2	3	4	5	6
138001 Mary River	0.135	**0.716**	0.019	0.019		
138002 Wide Bay Creek	0.009	0.066	0.008	**0.917**		
138003 Glastonbury Creek	0.229	**0.591**	0.020	0.160		
138004 Munna Creek	0.005	0.109	0.008	**0.878**		
138007 Mary River	**0.709**	0.258	0.032		0.001	
138009 Tinana Creek	**0.823**	0.148	0.004	0.020	0.005	
138010 Wide Bay Creek	0.005	0.040	0.003	**0.952**		
138102 Amamoor Creek	**0.836**	0.108	0.007	0.048	0.001	
138104 Obi Obi Creek	0.235	**0.450**	0.040	0.273	0.002	
138107 Six Mile Creek	**0.928**	0.010			0.062	
138109 Mary River	**0.957**	0.025	0.018			
138110 Mary River	**0.987**	0.008	0.001	0.001	0.003	
138111 Mary River	**0.990**	0.004	0.003		0.003	
138113 Kandanga Creek	**0.847**	0.132	0.006	0.008	0.007	
138119 Yabba Creek	0.001	0.039	0.004	**0.956**		
138903 Tinana Creek	0.191	**0.567**	0.095	0.147		
140002 Teewah Creek	0.176	0.073	**0.682**	0.035	0.031	0.003
141001 South Maroochy River	0.040	0.003		0.001	**0.881**	0.075
141003 Petrie Creek	0.002				**0.989**	0.009
141006 Mooloolah River	0.022				**0.936**	0.042
141008 Eudlo Creek	0.144	0.011		0.001	**0.844**	
141009 North Maroochy River	0.053	0.001	0.001		**0.905**	0.040
142001 Caboolture River	**0.929**	0.009			0.062	
142202 South Pine River	**0.688**	0.170	0.009	0.028	0.104	0.001
143001 Brisbane River	0.002	0.030	0.060	**0.908**		
143007 Brisbane River		0.005		**0.995**		
143009 Brisbane River		0.005		**0.995**		
143010 Emu Creek		0.008		**0.992**		
143035 Brisbane River	0.006	0.030	0.068	**0.896**		
143107 Bremer River	0.120	**0.786**	0.043	0.051		
143110 Bremer River	0.159	**0.773**	0.029	0.028	0.011	
143112 Reynolds Creek	0.015	0.122	0.116	**0.745**	0.002	
143210 Lockyer Creek		0.011		**0.989**		
143303 Stanley River	0.049		0.002		**0.940**	0.009
145107 Canungra Creek	**0.982**		0.015		0.003	
145003 Logan River	**0.950**	0.017	0.004		0.029	
145008 Logan River	0.016	0.165	0.405	**0.414**		
145010 Running Creek	**0.738**	0.019	0.234		0.009	
145012 Teviot Brook	0.039	**0.665**	0.023	0.273		
145014 Logan River	0.004	**0.601**	0.016	0.379		
145020 Logan River	0.038	0.408	**0.449**	0.104	0.001	
145099 Burnett Creek	0.025	0.137	0.117	**0.721**		
145101 Albert River	**0.986**	0.004	0.007		0.003	
145102 Albert River	**0.802**	0.185	0.008	0.005		
146002 Nerang River	0.054	**0.626**	0.069	0.255	0.001	
146010 Coomera River	**0.935**	0.003	0.004		0.058	
146012 Currumbin Creek	0.033				**0.967**	
146014 Back Creek	0.130	0.002	0.008		**0.858**	0.002
146020 Mudgeeraba Creek	**0.592**	0.003	0.004		0.401	

Figure 3.11: Box and whisker plots of prediction error from the Reference random forests model used to allocate gauges to RFCs.

(a) Prediction error across HFC. (b) Prediction error arranged by river catchment. Noosa and Pine–Caboolture catchments are not shown due to the low number of sites in these catchments. Plots based on data in Table 3.13.



Changes in individual flow metrics associated with flow regulation

Figure 3.12 is a heatmap (colour image) showing percentage difference in individual flow metrics under the Reference and Historic flow regimes, expressed as (Historic value –Reference value)/ Reference value for each gauge with a corresponding IQQM node. The colour of each grid represents the magnitude of the difference. Fifty-seven percent of the comparisons showed a change in value of 20% or less, and 37% of comparisons showed a change in metric value of 10% or less (Figure 3.12).

The extent to which individual flow metrics have changed from Reference condition varies between gauges. LSDur (low spell duration) is the flow metric that has undergone the greatest change in value relative to the Reference value, as shown by the yellow, orange and red colours in Figure 3.12. Most gauges have experienced an increase in low spell duration. Mean rates of rise and fall have also increased substantially when compared to the Reference value. In contrast, moving averages of the 3–90 flow minima (MA3-90dayMin) have decreased in value relative to the Reference flow value, as have mean monthly discharge metrics.

Magnitude metrics have undergone the greatest change across the study area (Figure 3.12), although this is skewed by the proportion of magnitude metrics in the dataset. In contrast, metrics associated with frequency, duration, variability and timing have undergone relatively minor change from Reference condition (Figure 3.12; Figure 3.13).

Figure 3.12: Heat map showing the percentage change in hydrologic metrics between reference and historic hydrologic regimes, expressed as (historic value – reference value) / reference value.

MeanZeroDay is expressed as the difference between reference and historic values due to division by zero. Negative values indicate that the reference metric value is higher than the historic metric value and positive values indicate that the historic metric value is greater than the reference metric value (see legend). Yellow and light blue cells indicate a change of 10% or less. Histogram shows absolute percentage change in flow metrics for changes of 0—125% only (total n*=1715). Numbers above bars are* n *for each category.*

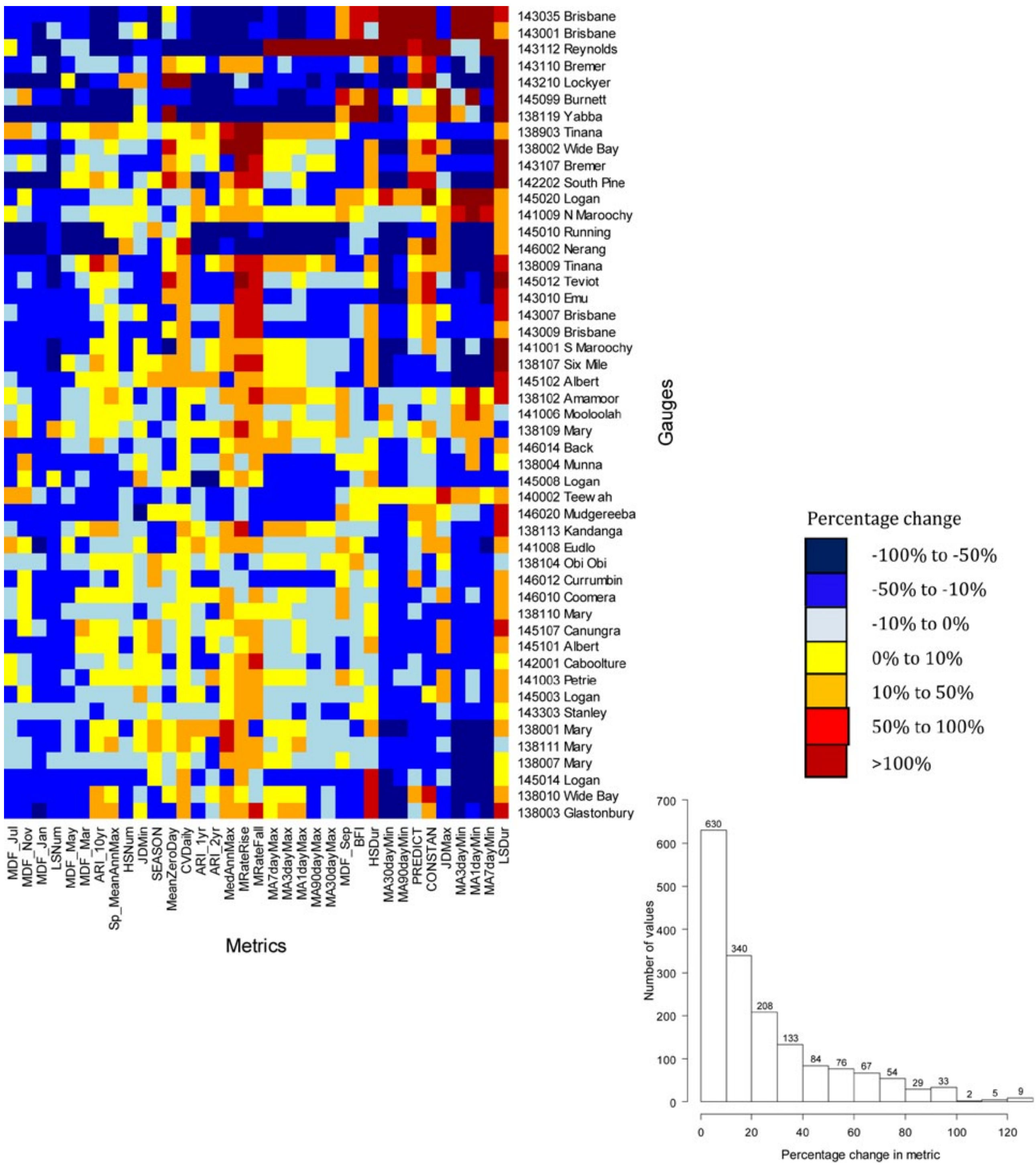


Figure 3.13: Box and whisker plot of the difference between Reference and Historic values for flow metrics, arranged by metric type

Difference between Reference and Historic flow metrics calculated as (Historic value–Reference value)/Reference value.

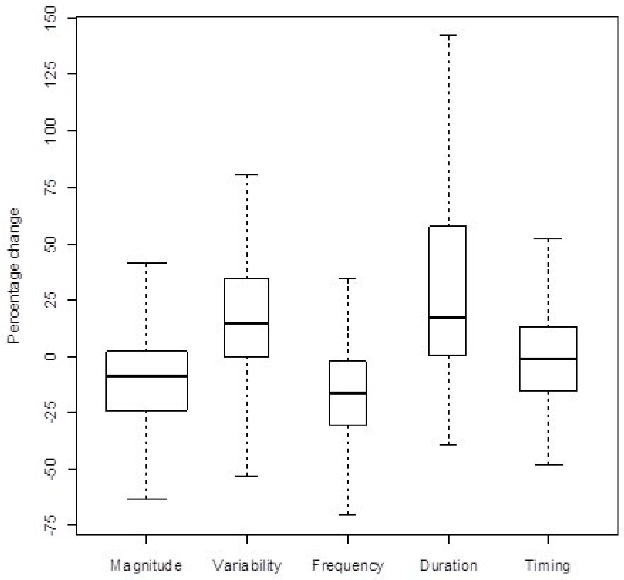


Figure 3.14 shows the percentage change in flow metrics between Reference and Historic flow regimes for those metrics identified by *clustvarsel* as best discriminating between HFCs (Figure 3.7). These plots represent gradients of flow regime change for individual metrics. In general the gauges on the extremes of the gradients represented in Figure 3.14 have flow regimes influenced by dams or weirs. An exception is ARI_10yr.

Many of the gauges with a positive change in ARI_10yr (i.e. Historic value exceeds Reference value) are not downstream of dams and weirs, and may suggest factors such as land use changes are influencing this metric. Changes in MA1dayMin and Ma30dayMIn have been negative (Reference value exceeds Historic) but the extreme changes in these metrics have been increased.

MA1dayMin has undergone the greatest change (shown by Brisbane River and Burnett Creek gauges but the North Maroochy River has also undergone a substantial increase in MA1dayMin despite not being subject to flow regime alteration by dams or weirs. HSDur and CONSTAN have tended towards substantial increases in these metrics. CVDaily has decreased in value for most gauges. For most metrics, it is evident that Reference values exceed Historic values.

Figure 3.14: Percentage change in metric values for metrics identified by clustvarsel as discriminating between HFCs.

MeanZeroDay is calculated as the difference between Historic and Reference values due to zero values. A positive difference indicates the Historic value is greater than the Reference value.

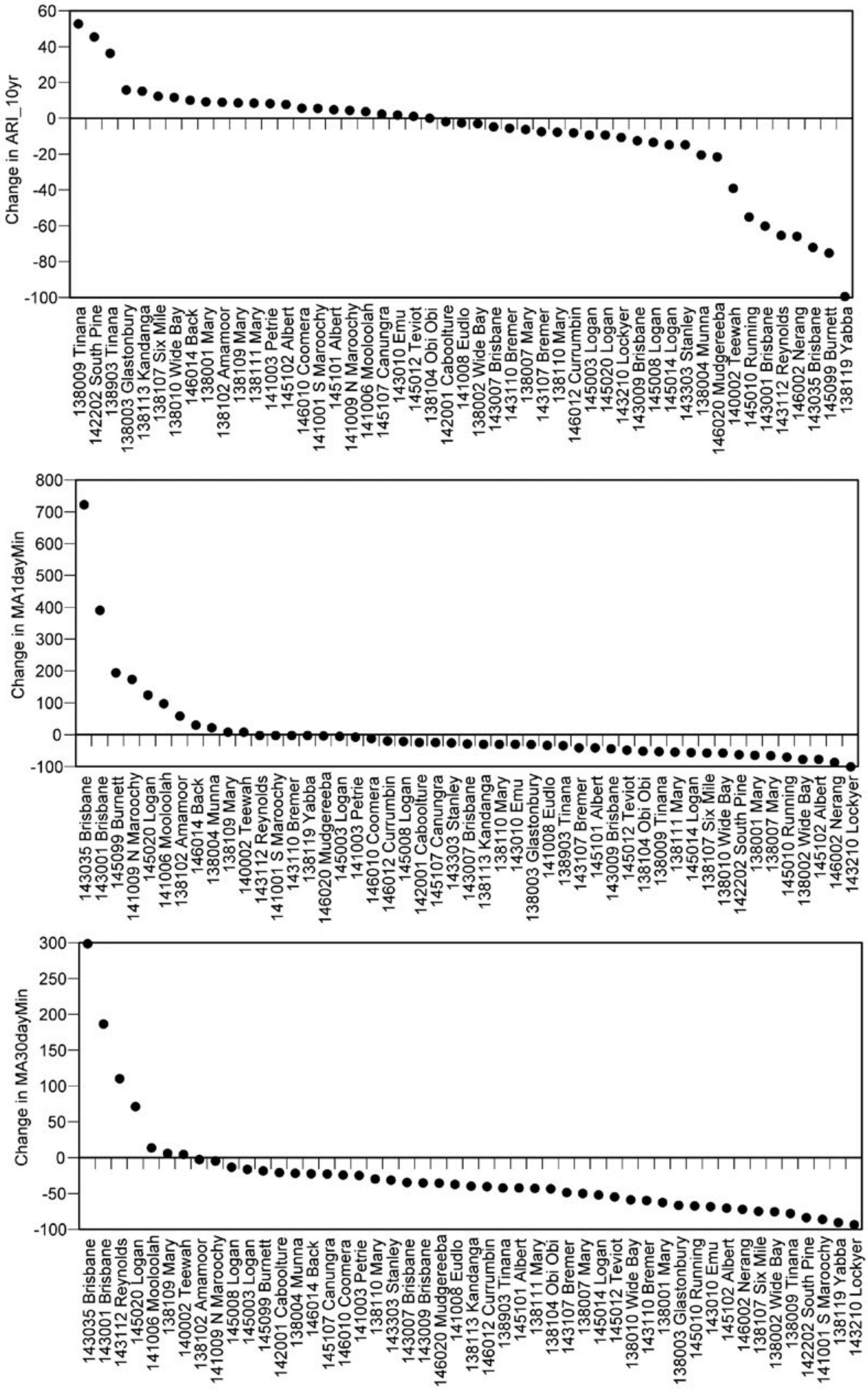
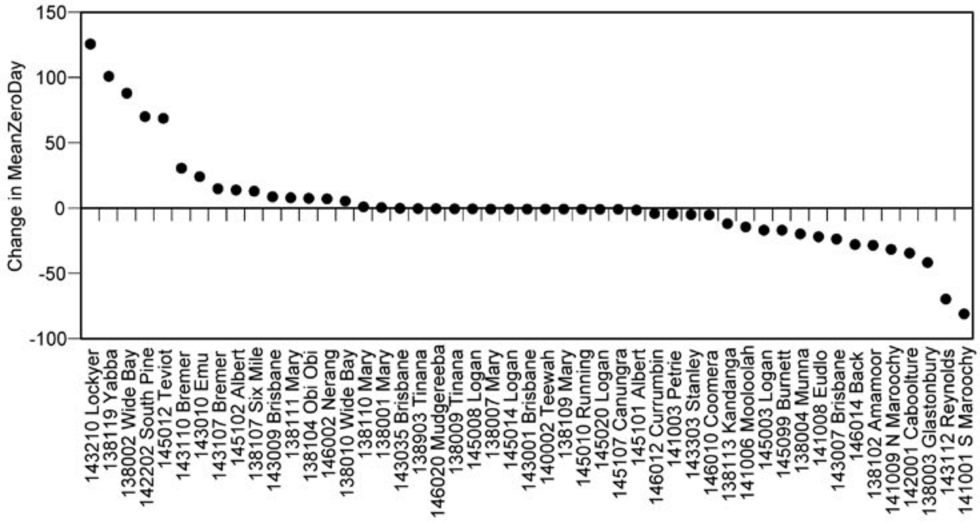
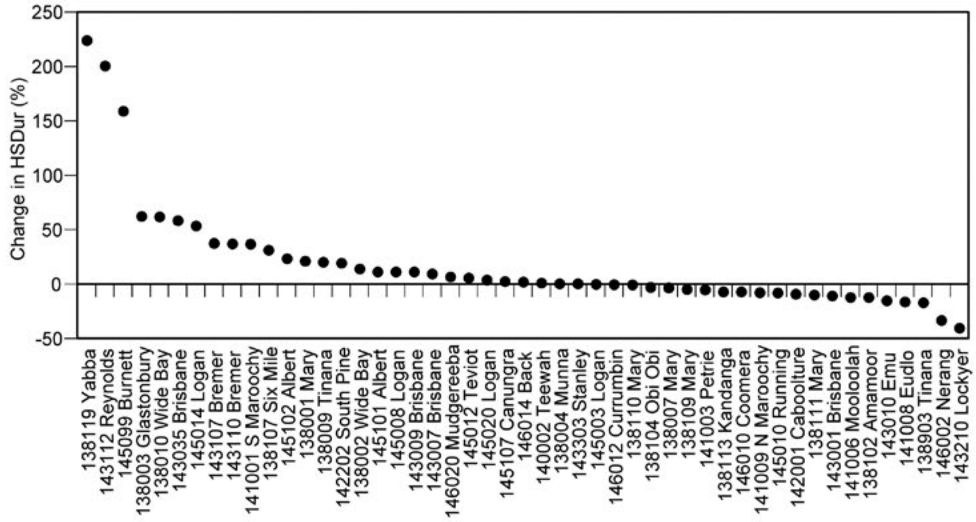


Figure 3.14 ***(continued)***





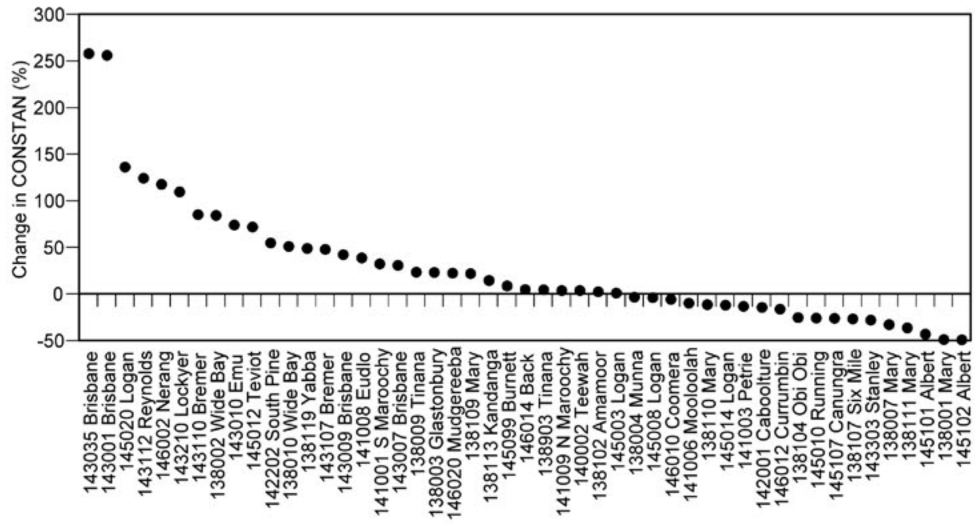
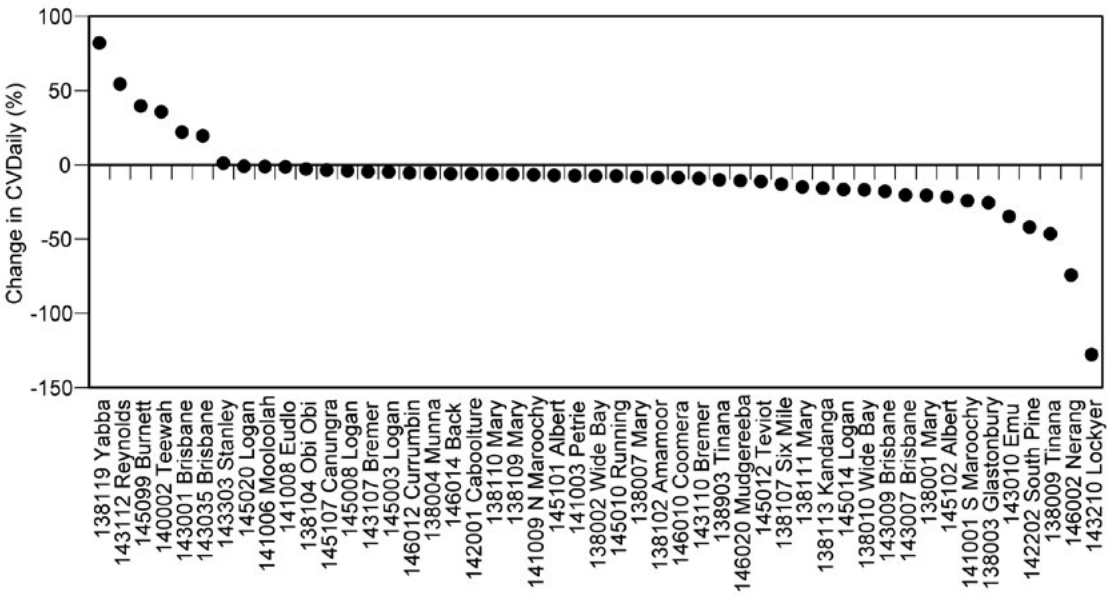


Figure 3.14 *(continued)*



Flow regime changes downstream of dams

Figure 3.15 shows differences between Reference and Historic flow regimes downstream of dams, and does not represent changes solely attributable to dams. Changes in flow regimes downstream of dams in the study area have been variable (Figure 3.15). Thirty-one percent of the metrics differed by less than 20% or less (compared with 57% for all gauges in Figure 3.12, see Histogram in Figure 3.15).

The dendrogram in Figure 3.15 indicates that there are two broad classes and four sub-classes of flow regulation by dams in the study area. The first broad class of flow regulation has resulted in substantial increases (mostly >100% but also in the range 50–100%) in some metrics (i.e. Historic values are larger than Reference values).

This group includes the Brisbane River downstream of Wivenhoe Dam (but upstream of the Lockyer Creek junction), Brisbane River at Savages Crossing (downstream of the Lockyer Creek junction) (collectively sub-class 1) and Reynolds Creek (sub-class 2). The flow metrics that have increased in this class are related to discharge magnitude (MA7-90dayMin, BFI), spell duration and variability PREDICT and CONSTAN (Figure 3.15). Specific flow regime changes downstream of Wivenhoe Dam include substantial increases in MA1-3dayMin and moderate increases in spell durations (but not evident further downstream at Savages Crossing, gauge number 143001).

Specific flow regime changes downstream of Moogerah Dam include substantial increases in MA1-90dayMax, MDF_Sep and JDMax, and moderate increase in spell durations. Gauges in this class have also undergone large decreases (50–100%) in the average recurrence intervals (ARIs) of one, two and 10 year floods (note that Reference values are still greater than Historic values in this case).

The second class of flow regulation has resulted in (mostly) small increases in some flow metrics (10–50%) but predominantly decreases of varying magnitude (mostly 50–100%) in flow metrics. This group includes Maroon Dam, Borumba Dam, Baroon Pocket Dam, Six Mile Creek Dam and Hinze Dam.

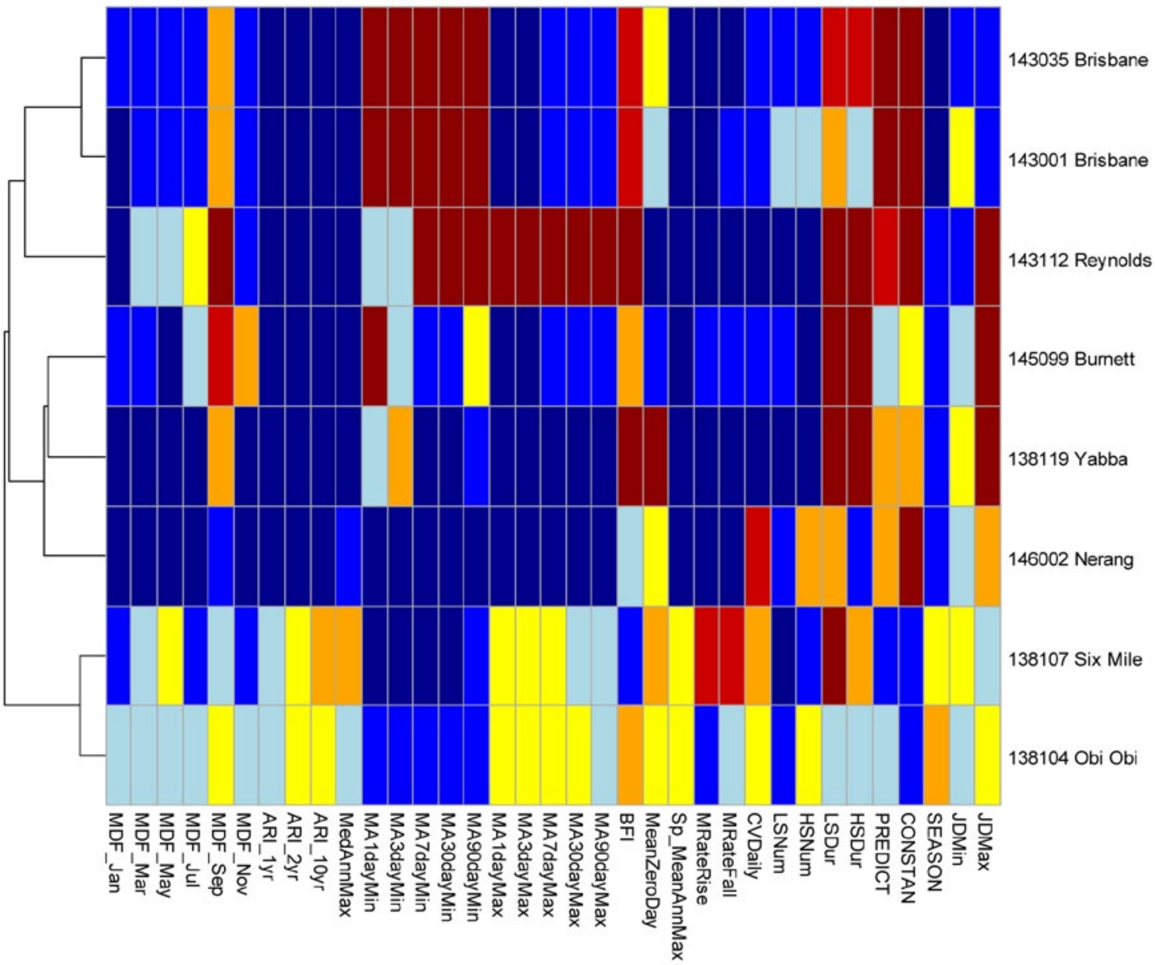
There is some variation in the magnitude of the change in individual flow metrics within this class. The dams that have caused the least extent of change in downstream flow regimes are Baroon Pocket Dam and Six Mile Creek Dam (Figure 3.15). Sub-class 3 includes Borumba Dam (Yabba Creek) and Maroon Dam (Burnett Creek). The flow regimes downstream of these dams include substantial changes in spell durations (LSDur and HSDur) and JDMax. Additional flow regime changes downstream of Borumba Dam include substantial increases in MeanZeroDay and BFI. Additional flow regime changes downstream of Maroon Dam include a substantial increase in MA1dayMin and a moderate increase (50–100%) in MDF_Sep (Figure 3.15).

Sub-class 4 includes Baroon Pocket Dam and Six Mile Creek Dam, which have undergone relatively minor flow regime change. The smallest extent of flow regime change has occurred downstream of Baroon Pocket Dam (Obi Obi Creek). Many of the flow metrics (25 of 35 metrics) have changed by less than 10% and the greatest extent of flow metric change is less than 50% (Figure 3.15).

Downstream of Six Mile Creek Dam 14 of 35 flow metrics have changed by 10% or less. Rates of rise and fall have increased to a moderate extent and LSDur has increased substantially (>100%). Hinze Dam (Nerang River) is grouped with Barron Pocket and Six Mile Creek Dams. Flow metric changes for Hinze Dam were generally in the range of -10 to -50 % (i.e. Reference value greater than Historic value) but substantial increases in CVDaily and CONSTAN have occurred (i.e. Historic value greater than Reference value).

Figure 3.15: Heat map showing the percentage change in hydrologic metrics between reference and historic hydrologic regimes, expressed as (historic value – reference value) / reference value.

MeanZeroDay is expressed as the difference between reference and historic values due to division by zero. Negative values indicate that the reference metric value is higher than the historic metric value and positive values indicate that the historic metric value is greater than the reference metric value (see legend). Yellow and light blue cells indicate a change of 10% or less. The dendrogram groups gauges downstream of dams with similar flow regime characteristics and was calculated using the Gower metric and hierarchical agglomerative clustering.

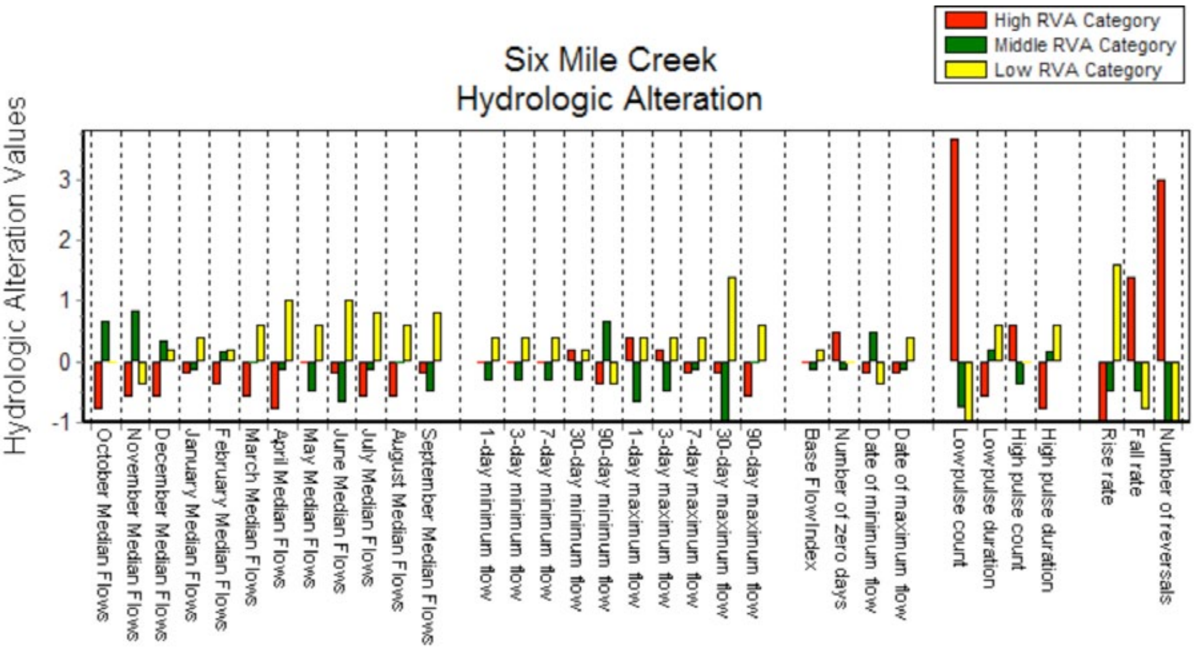


The Range of Variability Approach (RVA) was used to assess flow alteration downstream of the principal dams in the study area by comparison of pre and post-dam values for individual flow metrics. Due to the absence of pre-dam (gauge) data, Borumba Dam and Moogerah Dam were not included in this analysis. Results from the RVA analysis (Figure 3.16) support the conclusions drawn from the heatmap (Figure 3.15) concerning the extent of flow regime changes downstream of dams in the study area.

RVA plots suggest that flow regime changes downstream of Baroon Pocket Dam and Six Mile Creek dam have been relatively minor. Changes downstream of Hinze Dam (Nerang River) have tended towards an increase in most metrics, especially for the low RVA category (i.e. 0–33 percentiles). Changes downstream of Moogerah Dam (Reynolds Creek) have been more substantial, as shown by the Hydrologic Alteration Factor (y-axis).

Figure 3.16: Plots of change in IHA metrics identified by comparison of pre-dam and post-dam flow regimes

Yellow bars indicate the low RVA category (0–33 percentiles), green bars indicate the middle RVA category (34–66 percentiles) and red bars indicate the high RVA category (>66 percentile). The Hydrologic Alteration Value is calculated as (O–E)/E. Note differences in y-axis ranges.



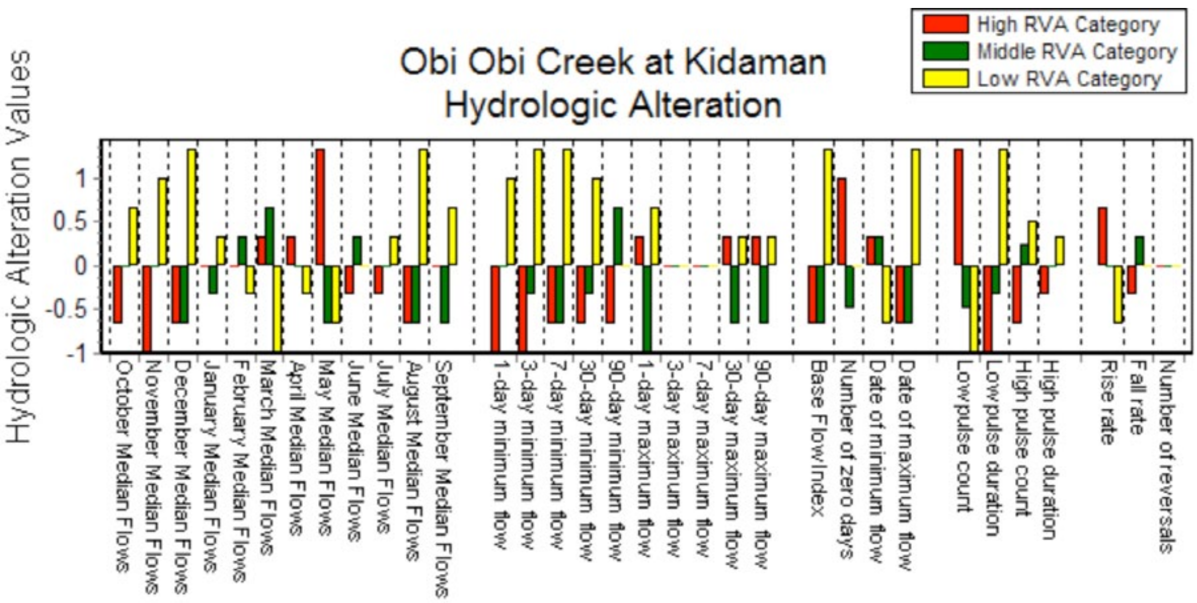
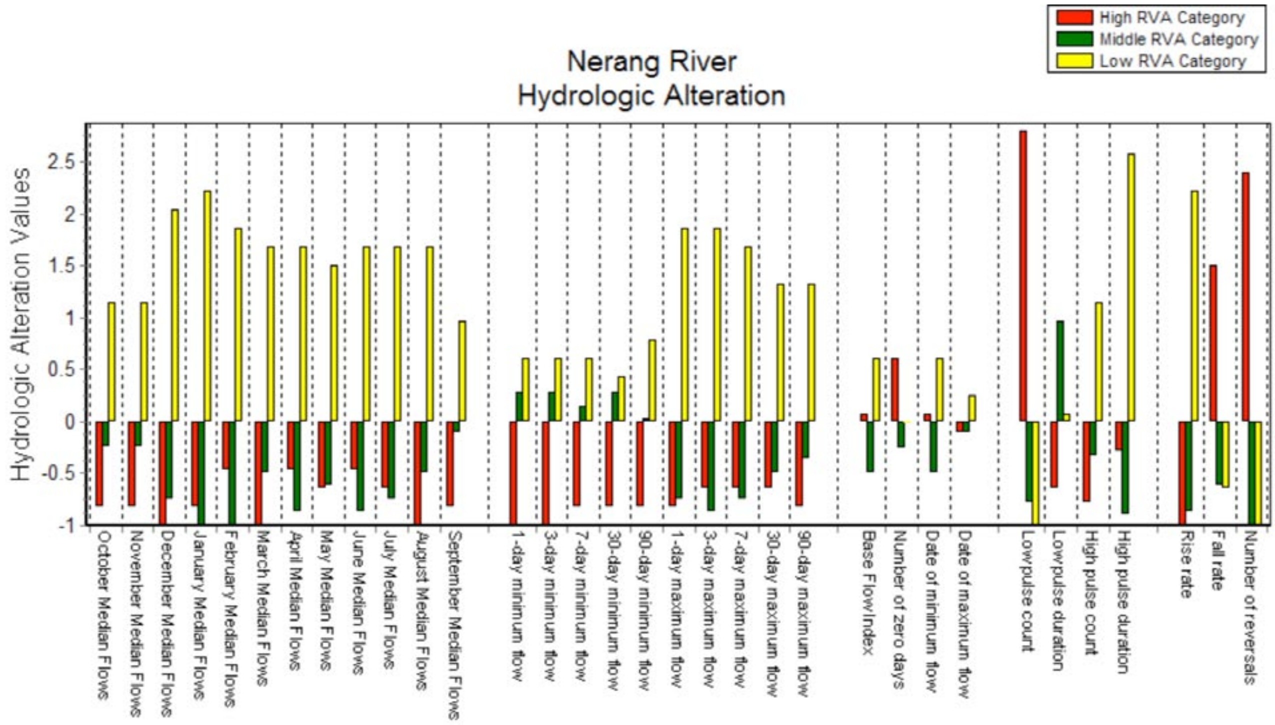
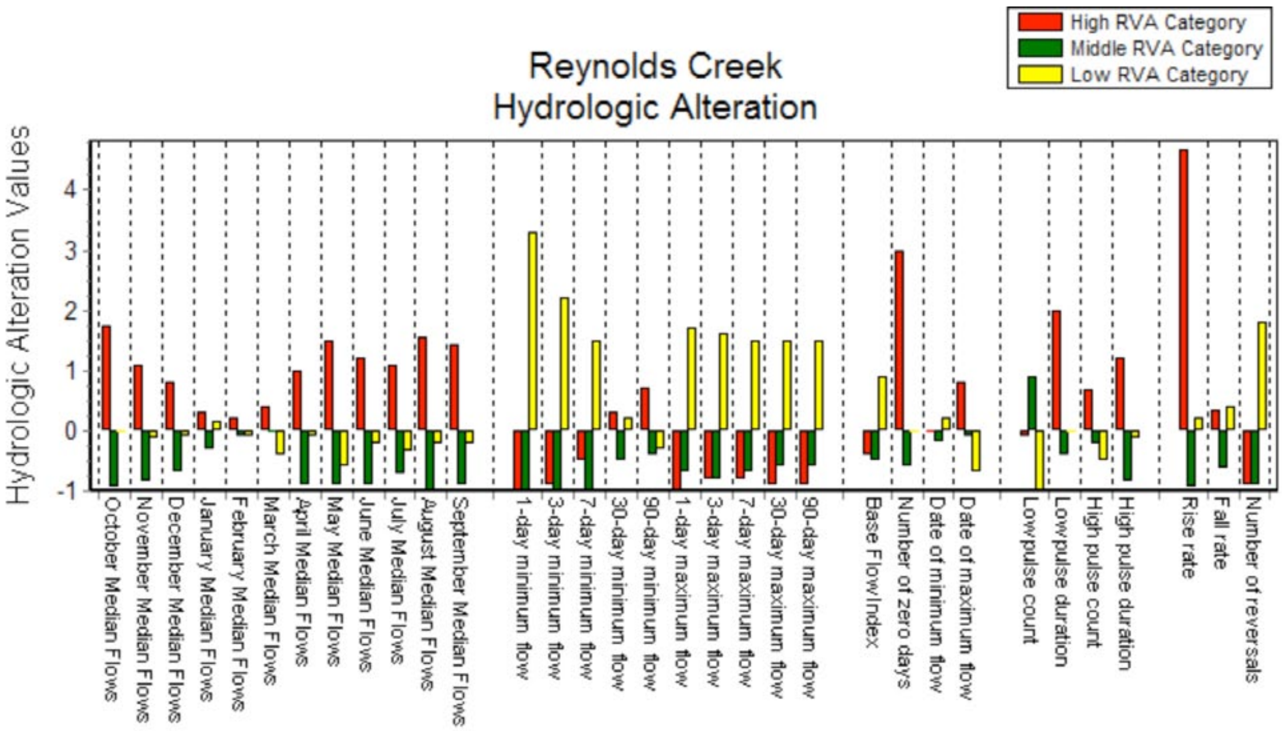


Figure 3.16: ***(continued)*** **Plots of change in IHA metrics identified by comparison of pre-dam and post-dam flow regimes**

Yellow bars indicate the low RVA category (0–33 percentiles), green bars indicate the middle RVA category (34–66 percentiles) and red bars indicate the high RVA category (>66 percentile). The Hydrologic Alteration Value is calculated as (O–E)/E. Note differences in y-axis ranges.





3.4 Discussion

The continental flow classification (Kennard et al. 2010a) identified four flow regime classes within the SEQ study area – two perennial flow classes and two intermittent classes. The flow classifications undertaken for the SEQ study area identified six RFCs and five HFCs.

The flow regime classes identified in this study display little concordance with the continental classification scheme, as shown by the relatively low adjusted Rand Index values (0.160 for the Reference classification, 0.124 for the Historic classification). One perennial flow class was identified under the Historic flow classification (comprised of sites influenced by flow regime alteration) and two perennial flow classes existed under Reference conditions. Flow regime change in the study area includes loss of perenniality and an increase in discharge intermittency.

There are similarities between the flow classification schemes in that flow classes identified in the Reference flow classification have analogues in the Historic flow classification. These flow classes included a flow class comprised of mostly Maroochy catchment sites (RFC and HFC 5) and a flow class comprised of sites in the drier part of the study area (including the Brisbane River and tributaries upstream of Wivenhoe Dam, Munna Creek and Wide Bay Creek). However, the adjusted Rand Index (0.382) indicated relatively little similarity between these classifications. The principal flow class changes associated with the Historic flow regime can be summarised as follows:

- loss of two RFCs (RFCs 3 and 6, although none of the IQQM nodes in class 6 had a corresponding gauge in the Historic flow classification)
- re-distribution of RFC 1 nodes into two HFCs (mainly HFCs 3 and 4)
- creation of a perennial HFC comprised of gauges influenced by flow regime alteration and one unregulated creek (Teewah Creek) with a relatively high groundwater component to discharge.

The main driver of spatial variability in flow regime patterns is discharge magnitude. The dominance of discharge magnitude metrics in driving both Reference and Historic classification schemes made it difficult to align these schemes with the continental classification scheme, since the principal gradient in that scheme is discharge perenniality–intermittency (Kennard et al. 2010a).

Differences in the continental classification scheme and the Reference and Historic classification schemes may be due in part to differences in the number of nodes/gauges included in each scheme. The Reference classification included 88 IQQM nodes and the Historic classification included 59 gauges, exceeding the 42 gauges representing SEQ in the continental classification (Kennard et al. 2010a). Differences in classification schemes may also be due in part to the inclusion of gauges with altered flow regimes in this study. However, the Reference classification showed only slightly better concordance with the continental classification than the Historic classification.

Despite the limited spatial extent of the Reference and Historic classifications, a geographic component is still evident. RFC and HFC 5 include streams and rivers with headwaters close to the coast, whereas systems with headwaters in the lower rainfall areas of the region were classified together. Consequently, most RFCs and HFCs generally included nodes or gauges from a mixture of catchments within the study area. Teewah Creek has a unique flow regime within the study area and was an obvious outlier in Reference and Historic ordinations (Figures 3.3 and 3.8). This discharge regime of this creek is strongly influenced by groundwater (Mary Basin Technical Advisory Panel 2005b) and hence has relatively low discharge variability (Historic CVDaily of 199%) and high constancy (0.511) when compared with other systems in the study area.

All HFCs contained at least one gauge with a flow regime influenced by a dam. HFC 1 included two Brisbane River gauges downstream of Wivenhoe Dam, Burnett Creek downstream of Maroon Dam, and two Logan River gauges downstream of the Burnett Creek–Logan River junction. Teewah Creek at Coops Corner is also in this flow class. The flow regime of Teewah Creek is not influenced by flow regulation, but a small pumping weir is present to supply water for local townships. This reduces discharge through the year, but as this stream is groundwater-fed permanent flows persist (Brizga et al. 2005b). A feature of flow regulation in this flow class is the high value for constancy (CONSTAN) compared with the remaining flow classes (Figure 3.7).

Human influence on the surface water resources of the SEQ study area is extensive. All major river catchments in the region have at least one dam (Table 3.2) and unsupplemented extraction and land use change also appear to have significant impacts upon flow regimes in the study area (e.g. see Running Creek in Figure 3.10).

Flow regime change in the study area is undoubtedly due to the construction and operation of dams, but also due to other factors that have not been determined. For example, while the greatest flow regime changes have been generally downstream of dams, three of the 11 gauges with the greatest flow regime change (as indicated by the Gower metric) are not directly influenced by dams or weirs. These were gauges on Running Creek, Mudgeeraba Creek and the South Pine River.

The geographic extent of flow regime alteration by dams and weirs (and possibly land use) in the study area is considerable. However, the degree of flow alteration from Reference conditions is relatively minor, with a maximum Gower metric of dissimilarity of 0.25, compared to 1 for perfect dissimilarity (total change in every flow metric).

In general the greatest flow regime changes in the study area have occurred in streams/rivers downstream of dams (Nerang River, Reynolds Creek, Yabba Creek, Lockyer Creek, Brisbane River, Burnett Creek), although three of the 11 gauges with the greatest flow regime change are not downstream of dams. These are gauges on Running Creek, Mudgeeraba Creek and the South Pine River. Land use changes in these systems are extensive.

The gauge with the greatest Gower flow regime metric (145010 Running Creek) is not subject to flow regime alteration by dams or weirs. The primary land use change in the Running Creek catchment is agriculture, and urbanisation for Mudgeeraba Creek and South Pine River. Furthermore, the presence of dams does not necessarily imply extensive flow regime change (e.g. Six Mile Creek has a Gower metric of 0.052).

Nevertheless, the extent of flow regime change across the study area is relatively minor, although local significant effects occur downstream of some dams in the study area. This is based on the capacity of the Reference random forests model to allocate stream gauges to the appropriate RFC, and the similarities in IQQM nodes and corresponding stream gauges as determined by the Gower metric which takes account of changes in all flow metrics.

Furthermore, changes in individual metrics from the modelled unregulated Reference state to the Historic regulated state were less than 20% for over half of the flow metrics, while approximately 30% of flow metrics have changed by 10% or less. While the overall extent of flow regime alteration in the study area appears to be relatively low, this is not to say that changes of this relatively low magnitude will not cause ecological changes. It is possible that a 20% change in some metrics could have a significant impact on certain ecological indicators and processes, and this is precisely what the ELOHA method seeks to determine.

It is of special interest to consider which flow metrics have changed most markedly in the study area. LSDur (low spell duration) is the flow metric that has undergone the greatest increase in value relative to the Reference value, as shown by the yellow, orange and red colours in the heat diagrams. Most gauges have experienced an increase in low spell duration. This marked change could represent the effects of dams on downstream flows, or levels of water extraction from impounded and regulated rivers, or effects of increasingly dry conditions over the study period, or all three processes.

Rates of rise and fall have increased substantially when compared to the Reference values, indicating greater flow variability captured in the gauged flow records. In contrast, the moving averages of the annual 3 day to 90 day minima have undergone decreases in value relative to the Reference flow value, as have mean monthly values. Again, these decreases could reflect the effects of dams on downstream flows, or levels of water extraction from regulated rivers, or effects of increasingly dry conditions over the study period, or all three processes.

The results of the analysis presented in the heat diagrams indicate that every dam has altered the flow regime in a different way, thus there is no replication of the full range of changes in flow regime. Instead, a range of different changes has occurred according to the characteristics of the original flow regime, the structure and purpose of the dam, its water release strategies and any downstream water extraction practices. If every individual dam has a different effect on the overall flow regime downstream, we might expect ecological impacts to differ also among regulated sites. However, similarities of ecological response might still become evident if particular flow metrics have a particularly powerful influence on biological systems.

The extent of flow regime change downstream of dams in the study area is highly dependent upon the height of the dam wall, the storage volume and the management strategy for the dam. Therefore, making generalisations about management strategies may be difficult since each dam appears to have generated a unique flow regime downstream. If every dam has different ecological effects, then it follows that ecological restoration by re-regulation of the flow regime (i.e. by providing environmental flows) will be likely to take a different form. However, this form will depend also on the particular ecological changes downstream, and the overall goals associated with an environmental water allocation.

In summary, comparisons of Reference and Historic flow regimes suggest that flow regime changes brought about by dams and other factors within the study area are relatively minor on the Gower scale of dissimilarity across all flow metrics considered during this study. This is not to say that changes of this relatively low magnitude will have no ecological effects. It is possible that a 10–20% change in overall flow regime, or in particular flow metrics could have a significant impact on certain ecological indicators, and this is precisely what the ELOHA method seeks to determine. The implications of ecological changes downstream from dams in SEQ for the overall ecological health of streams and rivers will be addressed in the ecological chapters of this report, and in the final Synthesis section.

3.5 Attachments

Attachment 3.1: IQQM nodes and stream gauges

This attachment provides the IQQM nodes and stream gauges used in the hydrologic classification of streams in SEQ. 'Flow alteration' indicates the presence of a dam or weir upstream of the gauge. 'Upstream dam' and 'Upstream weir' indicate the presence of one or more dams or weirs on tributaries upstream of the gauge. ‡ indicates IQQM node without comparable stream gauge in Historic flow classification (due to absence of a gauge or unsuitable gauge record). 'IaBT' indicates intrabasin water transfer scheme.

IQQM nodes and stream gauges	Latitude (south)	Longitude (east)	Catchment area (km2)	Period of record	Flow alteration
Mary River					
138001a Mary River at Miva	25.9550	152.4947	4755	Jan 1910 - Current	Upstream dams
138002c Wide Bay Creek at Brooyar	26.0069	152.4106	655	Oct 1966 - Current	None
138003d Glastonbury Creek at Glastonbury	26.2178	152.5211	113	Jul 1979 - Current	None
138004b Munna Creek at Marodian	25.9050	152.3481	1193	Oct 1974 - Current	None
138007a Mary River at Fishermans Pocket	26.1711	152.5997	3068	May 1968 - Current	Upstream dams
138009a Tinana Creek at Tagigan Road	26.0789	152.7828	100	Jul 1974 - Current	None
138010a Wide Bay Ck at Kilkivan	26.0811	152.2181	322	Oct 1974 - Current	None
138014a Mary River at Home Park	25.7694	152.5264	6845	June 1982 - Current	Upstream dams
138101b Mary River at Kenilworth‡	26.5967	152.7308	720	Oct 1925 - Sep 1974	None
138102c Amamoor Creek at Zachariah Lane	26.3669	152.6192	133	Nov 1982 - Current	None
138104a Obi Obi Creek at Kidaman	26.6283	152.7681	174	Oct 1920 - Sep 1964	Dam
138105c Yabba Creek at Imbil‡	26.4603	152.6794	623	Oct 1971 - Jun 1982	Upstream dam
138106a Obi Obi Creek at Baroon Pocket‡	26.7053	152.8633	67	Jan 1890 – Jun 1999	None
138107b Six Mile Creek at Cooran	26.3322	152.8125	186	Feb 1981 - Current	Dam
Six Mile Creek at Cooroy‡	Jan 1890 – Jun 1999	None			
138109a Mary River at Dagun Pocket	26.3239	152.7025	2097	Feb 1957 - Jul 2002	Upstream dams
138110a Mary River at Bellbird Creek	26.6289	152.7036	486	Oct 1959 - Current	None
138111a Mary River at Moy Pocket	26.5278	152.7428	820	Oct 1963 - Current	Upstream dams
138113a Kandanga Creek at Hygait	26.3914	152.6436	143	Nov 1971 - Current	None
138119b Yabba Creek at Borumba Dam Release	26.5031	153.5856	498	Apr 1979 - Jul 2002	Dam
138120a Obi Obi Creek at Gardners Falls	26.7606	152.8725	26	Dec 1986 - Current	Weir
138903a Tinana Creek at Bauple East	25.8219	152.7211	783	Jul 1981 - Current	Upstream weirs
Noosa River					
140002a Teewah Creek at Coops Corner	26.0589	153.0412	53	Jan 1972 - Current	None
Kin Kin Creek at Mouth‡	Jan 1890 – Jun 1999	None			
Maroochy River					
141001b South Maroochy River at Kiamba	26.5914	152.9031	33	Nov 1985- Current	Weir
141002a South Maroochy River Kureelpa‡	26.6042	152.8936	20	Jan 1890 – Jun 1999	None
141003c Petrie Creek at Warana Bridge	26.6244	152.9575	38	Oct 1978 - Current	Weirs
141004b South Maroochy River at Yandina	26.5636	152.9381	75	Feb 1982 - Current	Dam
141006a Mooloolah River at Mooloolah	26.7631	152.9811	39	Dec 1971 - Current	None
141008a Eudlo Creek at Kiels Mountain	26.6636	153.0175	62	Jan 1982 - Current	None
141009a North Maroochy River at Eumundi	26.4961	152.9606	38	Feb 1982 - Current	None
Rocky Creek at Cooloolabin Dam‡	Jan 1890 – Jun 1999	None			
Mooloolah River at Addlington Creek‡	Jan 1890 – Jun 1999	None			
Pine-Caboolture River					
142001a Caboolture River at Upper Caboolture	27.0981	152.8911	94	Oct 1965 - Current	None
North Pine River at Dam Outflow‡	27.2634	152.9379	Jan 1889 – Jun 2000	None	
142101a North Pine River at Youngs Crossing‡	27.2686	152.9542	403	Jan 1889 – Jun 2000	None
142103a North Pine River at Laceys Crossing‡	27.1917	152.7969	118	Jan 1889 – Jun 2000	None
142202a South Pine River at Drapers Crossing	27.3486	152.9228	156	Oct 1965 - Current	None

IQQM nodes and stream gauges	Latitude (south)	Longitude (east)	Catchment area (km2)	Period of record	Flow alteration
Brisbane River					
143001c Brisbane River at Savages Crossing	27.4400	152.6686	10 180	Oct 1958 - Current	Dam
143018a Brisbane River at Avoca Vale‡	26.7525	152.2350	1498	Jan 1889 – Jun 2000	None
143007a Brisbane River at Linville	26.8047	152.2725	2009	Oct 1964 - Current	None
143009a Brisbane River at Gregors Crossing	26.9897	152.4039	3866	Feb 1962 - Current	None
143010b Emu Creek at Boat Mountain	26.9786	152.2850	915	Nov 1976 - Current	None
143015b Cooyar Creek at Dam Site	26.7411	152.1367	963	Dec 1990 - Current	None
143028a Ithaca Creek at Jason Street	27.4508	152.9925	10	Sep 1972 - Current	None
143032a Moggill Creek at Upper Brookfield	27.4900	152.8908	23	Jul 1976 - Current	None
143033a Oxley Creek at New Beith	27.7311	152.9467	60	Dec 1976 - Current	None
143035a Brisbane River at Wivenhoe Dam Tailwater	27.4011	152.6067	7023	May 1986 - Current	Wivenhoe Dam
143112a Reynolds Creek at Moogerah Dam Tailwater	28.0267	152.5514	227	31/10-1980-21/06/2002	Moogerah Dam
143107a Bremer River at Walloon	27.6033	152.6928	620	Oct 1961 - Current	None
143108a Warrill Creek at Amberley	27.6661	152.6989	914	Oct 1961 - Current	Weirs/IaBT
143110a Bremer River at Adams Bridge	27.8331	152.5086	125	Sep 1968 - Current	None
143210b Lockyer Creek at Rifle Range Road	27.4553	152.5164	2490	Feb 1988 - Current	Weirs
143303a Stanley River at Peachester	26.8392	152.8403	104	Jul 1927 - Current	None
Stanley River at Woodford Weir Inflows‡	Jan 1889 – Jun 2000	None			
143306a Reedy Creek at Upstream Byron Creek	27.1356	152.6394	56	Jun 1975 - Current	None
143307a Byron Creek at Causeway	27.1308	152.6500	79	Jun 1975 - Current	None
143921a Cressbrook Creek at Rosentretters Crossing	27.1381	152.3294	447	Aug 1986 – Current	Dams
Cressbrook Creek at Cressbrook Dam Outflows‡	27.2630	152.2097	Jan 1889 – Jun 2000	None	
Cressbrook Creek Outflows to Brisbane River‡	27.0786	152.4448	Jan 1889 – Jun 2000	None	
Logan-Albert River					
145003b Logan River at Forest Home	28.2022	152.7697	175	Oct 1953 - Current	None
145008a Logan River at Round Mountain	28.0725	152.9252	1262	Jul 1957 - Current	Upstream dam
145010a Running Ck at Deickmann Bridge	28.2478	152.8908	128	Nov 1965 - Current	None
145011a Teviot Brook at Croftby	28.1494	152.5686	83	Feb 1966 - Current	None
145012a Teviot Brook at The Overflow	27.9308	152.8578	503	Mar 1966 - Current	None
Teviot Brook at Logan River Junction‡	27.8433	152.9462	Jan 1890 – Jun 2003	None	
145014a Logan River at Yarrahappini	27.8328	152.9858	2416	Apr 1969 - Current	Weirs
145018a Burnett Creek upstream of Maroon Dam	28.2219	152.6075	82	May 1970 - Current	None
145020a Logan River at Rathdowney	28.2183	152.8664	533	Dec 1973 - Current	Upstream tributary
145099a Burnett Creek at Maroon Dam Tailwater	28.1828	152.6586	106	11/06/1974-1/07/2002	Maroon Dam
145101d Albert River at Lumeah No. 2	28.0614	153.0408	169	Oct 1953 - Current	None
145102b Albert River at Bromfleet	27.9114	153.1056	544	Oct 1927 - Current	None
145103 Cainbable Creek at dam site	28.0936	153.0764	42	Jun 1962 - Current	None
145107a Canungra Creek at Main Road Bridge	27.9989	153.1567	101	Jan 1973 - Current	None
Albert River at Glendower‡	Jan 1890 – Jun 2003	None			
Logan River at Bromelton Rocks‡	Jan 1890 – Jun 2003	None			
Logan River at Teviot Brook Junction‡	27.8433	152.9462	Jan 1890 – Jun 2003	None	
South Coast					
146002b Nerang River at Glenhurst	28.0000	153.3103	240	Oct 1967 - Current	Hinze Dam
146004a Little Nerang Creek at Neranwood‡	28.1258	153.2922	40	Jan 1890- Jun 2000	None
146009a Little Nerang Creek at 4.0km‡	28.0756	153.3053	53	Jan 1890- Jun 2000	None
146010a Coomera River at Maybury	28.0258	153.1928	88	Oct 1962 - Current	None
146011a Nerang River at Whipbird‡	28.0917	153.2594	122	Dec 1965 - Jul 1986	None
146012a Currumbin Creek at Nicolls Bridge	28.1797	153.4219	30	Feb 1970 - Current	None
146014a Back Creek at Beechmont	28.1256	153.1878	7	Jun 1971 - Current	None
146020a Mudgeeraba Creek at Springbrook Road	28.0853	153.3500	36	Dec 1989 - Current	None
146095a Tallebudgera Creek at Tallebudgera Creek Rd	28.1519	153.4011	56	Jun 1970 - Current	Weir

4. Field study design

This chapter presents an outline of the methods used to design the fieldwork component of the project, including selection of study sites. Selection of study sites is based on the results of the flow classification (Chapter 3) and requirements for surveying each of the focal biotic components (riparian vegetation, aquatic vegetation, fish) in relation to the project aims (Chapter 1).

4.1 Principles for field research program

The types and degree of ecological impact on rivers brought about by flow regime alteration can be examined in at least five different ways:

- before–after comparisons (BACI designs)
- referential approach (comparison of impacted sites with Reference sites)
- comparison of Observed vs Expected ecological attributes (as in RIVPACS, and Kennard et al. 2006a,b for fish)
- experiments to tease out the mechanisms underlying patterns identified by other methods
- multiple lines and levels of evidence (Downes et al. 2002).

This study applied the referential approach to establish potential gradients of response to flow regime alteration within hydrologic classes.

4.1.1 Referential approach for south-east Queensland streams

In the first stage of designing the field research program, a series of principles and criteria for the selection of field sampling sites was developed.

Site selection was based primarily upon the classification of stream flow regimes within the study area (Chapter 3) which identified six RFCs representing pre-development flow conditions, and five HFCs representing the actual flow regime recorded at stream gauges through time, which may be influenced by changes to land use through time, water resource development and unsupplemented extraction.

4.1.2 Selection of study reaches and sites

The effects of flow conditions preceding each biotic sampling event were of primary interest in this study and it was therefore essential that discharge data were available for the duration of the field program. Thus all study sites had to be located near a currently operating stream gauge.

The section of stream or river upstream and downstream of an individual stream gauge for which the discharge recorded at the gauge could be considered representative is defined as a *study reach*. Study reaches were defined by local topography and the presence of major inflows upstream or downstream of individual stream gauges. Sixty-nine gauges used in the Historic flow classification (Attachment 3.11) were currently operating at the time of the study.

Following delineation of study reaches, a subset of reaches was selected for which suitable study sites were expected to be found. This was based on ease of access, the presence of suitable riparian vegetation (see below), the location of tributaries near gauges, and workplace health and safety issues. Study reaches in the main channel of the Brisbane River were not considered since the Brisbane River was the focus of a recent environmental flow study (Arthington et al. 2000), the riparian vegetation of the main channel is highly modified (Arthington et al. 2000), and the river downstream of Wivenhoe Dam is difficult to survey due to the high volume of water released constantly from the dam.

However, study reaches were selected downstream of the other major dams in the study area where discharge data from stream gauges were available and access was available. Site selection was also limited by depth requirements for sampling aquatic vegetation and fishes (maximum suitable depth 1–1.5 m) and current and historical land use (in particular grazing, vegetation clearance and fire regimes) for riparian vegetation surveys.

4.1.3 Site location and extent on the ground

The site selection principles and criteria provided approximate locations (reaches) for field sampling within the proximity of stream gauges. The selection of specific field sampling locations or sites within reaches was made using aerial photos, satellite imagery, discussion with landholders, previous field experience and field visits.

Field sites were defined as a stream length of 100 m (maximum) as this length allowed multiple vegetation transects to be conducted within a site and usually incorporated multiple in-stream habitats (riffles, runs and pools) whilst minimising variation due to changes in stream morphology, geology and neighbouring land use. Sites were located as close as possible to the nearest stream gauge (upstream or downstream) to minimise differences between flow at the gauge and the actual field site as described above.

Two field sites were surveyed within each reach. Sites within a study reach were a minimum of 2 km apart to ensure site independence as far as is possible whilst maintaining a reasonable proximity to the nearest stream gauge. A further important consideration, particularly for the riparian vegetation element, was current and historical land uses. Sites were selected initially wherever possible that were not currently grazed, had not been cleared in the last 20–30 years and were not subject to regular burning.

These criteria were stipulated in order to reduce other influences on flow–ecology relationships, particularly for the riparian vegetation component which is likely to be strongly influenced by such factors (but also for in-stream biota and processes that are influenced, in turn, by riparian vegetation structure and processes (Pusey and Arthington 2003). Anthropogenic impacts upon riparian vegetation at potential sites were inferred from observation and by discussion with landowners. Grazed sites were unavoidable in some reaches (particularly given other restrictions on site selection) due to the widespread extent of grazing in some catchments and lack of livestock exclusion from the riparian zone (e.g. Burnett Creek and Teviot Brook).

4.1.4 Selection of Reference sites for sites downstream of dams

Following classification, regulated reaches (stream reaches close to a stream gauge or IQQM node located downstream of a dam or weir) within SEQ were identified (Figure 4.1) and their RFC and HFC membership determined (Table 4.1).

Regulated reaches were selected where hydrological analyses (Chapter 3) indicated flow regimes strongly deviated from natural (regulated by dams rather than weirs) and for which appropriate non-regulated References could be found. Limitations on data availability (both gauged and modelled) precluded the inclusion of some regulated reaches in these analyses. Once the RFC and HFC membership of each regulated reach had been established, Reference reaches (stream sections proximal to a gauge or IQQM node not subject to significant flow regime alteration) were selected in several categories:

- unregulated References of the pre-development condition (from IQQM data – geographically close)
- unregulated References of pre-development condition (from IQQM data – not geographically close, e.g. a different catchment)
- unregulated References of the developed condition (from stream gauge data –geographically close)
- unregulated References of developed condition (from stream gauge data – not geographically close, e.g. a different catchment)
- a replicate regulated reach (from pre-development and developed condition).

Additional criteria were:

- The Reference reaches occurred within the same flow class within the classification as the regulated reach. However, where an appropriate Reference condition could not be established for a particular flow class due to limited data availability (modelled and gauged) or availability of potential stream reaches within each flow group, Reference reaches were selected from the next closest flow class determined from the classification.
- If the closest Reference reach was upstream of a dam (i.e. Burnett Creek and the South Maroochy River) and therefore an upstream–downstream comparison, we selected an additional proximal Reference reach that was independent of the regulated system.
- Where a reach was used for a number of comparisons (e.g. Amamoor Creek and Burnett Creek upstream of Maroon Dam), we selected an additional replicate Reference reach (e.g. Glastonbury Creek and Teviot Brook) to provide greater confidence in the Reference condition.
- An additional reach was selected near the location of the partially completed Wyaralong Dam on Teviot Brook and appropriate References were also selected 'before' baseline data in the event of the construction of these dams. Wyaralong Dam has since been completed.

In addition to those reaches identified using the above criteria, we also selected reaches from RFCs and HFCs that were not represented or poorly represented by the above site selection criteria (Table 4.2). Final sites selected for the study are also presented in Table 4.2.

Figure 4.1: Locations of proposed field sites (reaches proximal to gauges and/or IQQM nodes)

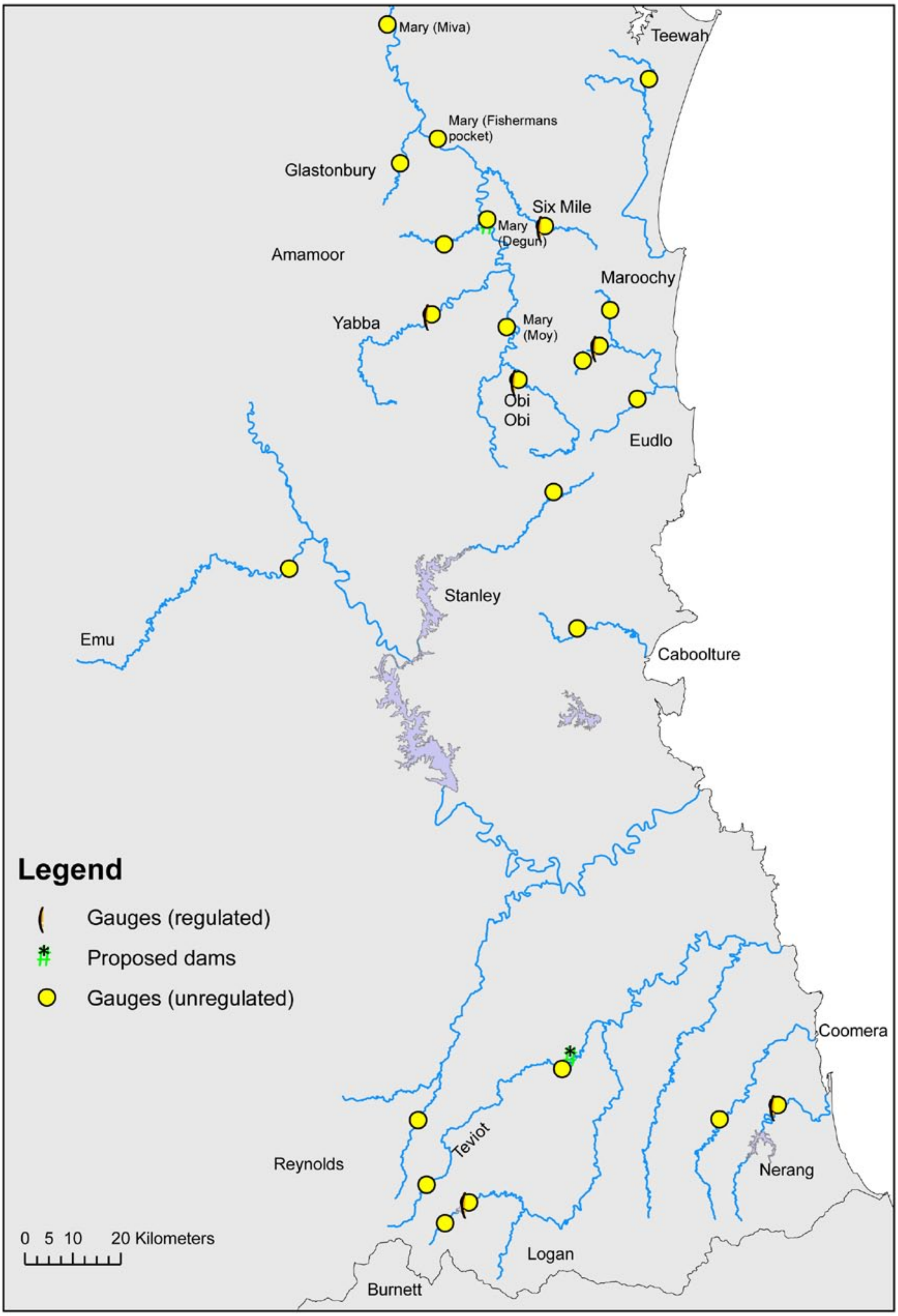


Table 4.1: Pre-development and Historic reference reaches for flow regulated reaches in the study area, as determined from the site selection principles and criteria

Each regulated reach has a geographically close and geographically distant reference site.

Dam	RFC	HFC	Pre-development reference (geographically close)	Pre-development reference (geographically distant)	Historic reference (geographically close)	Historic reference (geographically distant)
Yabba Creek	2	2	Glastonbury Creek	Logan River at Rathdowney	Amamoor Creek (HFC4)[1]	Teviot Brook at The Overflow
Six Mile Creek	1	4	Amamoor Creek	Coomera River	Amamoor Creek	Coomera River
Obi Obi Creek	1	3	Amamoor Creek	Coomera River	Mary River at Moy Pocket	Burnett Creek upstream of Maroon Dam
Burnett Creek	2	1	Logan River at Rathdowney	Glastonbury Creek	Teviot Brook at Croftby	Glastonbury Creek
Reynolds Creek	2	1	Logan River at Rathdowney	Glastonbury Creek	Teviot Brook at Croftby	Glastonbury Creek
Nerang River	5	3	Currumbin Creek	Stanley River	Burnett Creek upstream of Maroon Dam	Mary River at Moy Pocket

[1] An appropriate Historic reference (geographically close) could not be found in HFC 2 for Yabba Creek.

Table 4.2: Summary of study sites

Sites 29–42 were added in 2009 to provide coverage of RFCs and HFCs poorly represented through other criteria

Site number and name	Latitude	Longitude	RFC	HFC
1. Stanley River at Cove Road	-26.9197	152.7725	5	5
2. Burnett Creek downstream of gauge 145018a	-28.2163	152.6138	No class	3
3. Burnett Creek upstream of gauge 145018a	-28.2274	152.6040	No class	3
4. Nerang River at Grand Manor Golf Course	-28.0223	153.3026	5	3
5. Coomera River at Coomera Scouts Hall	-28.0468	153.1899	1	4
6. Nerang River at Weber Court near Chantrill Avenue	-28.0060	153.3139	5	3
7. Teviot Brook near Brennan Road[1]	-28.1621	152.5583	No class	3
8. Teviot Brook at Croftby	-28.1562	152.5718	No class	3
9. Amamoor Creek at Harrys Creek Road[2]	-26.3461	152.6560	1	4
10. Yabba Creek at Stirling Crossing	-26.4901	152.6275	2	2
11. Yabba Creek at No. 8 Crossing	-26.4983	152.5916	2	2
12. Obi Obi Creek downstream of number 2 crossing	-26.6340	152.7837	1	3
13. Obi Obi Creek upstream of number 2 crossing	-26.6397	152.7901	1	3
14. Mary River downstream of Walker Road	-26.5117	152.7461	1	3
15. Six Mile Creek at Old Noosa Road	-26.3297	152.8092	1	4
16. Six Mile Creek at Grahams Road	-26.3420	152.8641	1	4
17. Glastonbury Creek at Greendale Road Crossing	-26.1835	152.5276	2	3
18. Eudlo Creek at gauge site	-26.6625	153.0181	5	5
19. Eudlo Creek upstream of Bruce Highway	-26.6860	152.9964	5	5
20. Reynolds Creek at Yarramalong camp ground	-28.0116	152.5565	2	1
21. Reynolds Creek at downstream of Purdons Bridge	-28.0007	152.5699	2	1
22. Amamoor Creek at Zachariah Lane	-26.3669	152.6223	1	4
23. Glastonbury Creek at 2 km from Mary River confluence	-26.1544	152.5528	2	3
24. Mary River at Moy Pocket north of quarry	-26.5257	152.7395	1	3
25. Coomera River at Tucker Lane	-28.0561	153.1786	1	4
26. Stanley River at gauge site	-26.8392	152.8403	5	5
27. Burnett Creek at 2 km downstream of Maroon Dam	-28.1759	152.6722	2	1
28. Burnett Creek at Splityard Creek Road	-28.1659	152.6814	2	1
29. Currumbin Creek at Currumbin Valley Primary School	-28.2075	153.3953	5	5
30. Currumbin Creek at Fordyce Court	-28.1908	153.4167	5	5
31. Wide Bay Creek downstream of gauge 138002c	-26.0019	152.4286	4	2
32. Wide Bay Creek upstream of gauge 138002c	-26.0047	152.4072	4	2
33. Munna Creek at gauge 138004b	-25.9042	152.3489	4	2
34. Munna Creek downstream of gauge 138004b	-25.9014	152.3500	4	2
35. North Maroochy River at Eumundi	-26.4697	152.9544	5	5
36. North Maroochy River at North Arm-Yandina Creek Road	-26.5231	152.9600	5	5
37. Mary River at Bauple-Woolooga Road	-25.8861	152.4864	3	3
38. Mary River at Orphants Road	-25.9533	152.4956	3	3
39. Tinana Creek at gauge site	-25.8200	152.7222	2	3
40. Tinana Creek at upstream of gauge	-25.8356	152.7229	2	3
41. Logan River at Running Creek Road	-28.2128	152.8739	2	1
42. Logan River at upstream Tilleys Bridge	28.2233	152.8583	2	1
43. Teviot Brook at Wyaralong a	27.8969	152.9014	2	2
44. Teviot Brook at Wyaralong b	-27.9061	152.8583	2	2

5. Land use

5.1 Introduction

Whilst stream flows are recognised as one of the principal orchestrators of stream ecologies (Poff et al. 1997; Bunn and Arthington 2002), many other catchment characteristics not directly related to stream flows are also important drivers of stream ecological processes. Understanding the influences of flow on ecological responses, given the inherent underlying variability in environmental and anthropogenic activities across the study region, presents significant challenges to the ELOHA trial in SEQ.

There is considerable natural environmental variation across the study region with a complex geology (Bridges et al. 1990; Ellis 1968; Murphy et al. 1976; Whitaker and Green 1980) and associated soils. Distinct topographic regions are identifiable within the region with coastal plains, river floodplains and estuaries in the east, and foothills and mountains with plateaux over 300 m above sea level to the west, north and south of the study region. The climate is subtropical and dominated by summer rainfall with warm summers and mild winters, but sits adjacent to the temperate/subtropical transitional zone. The area also exhibits a strong rainfall gradient with a decrease in rainfall in a westerly (inland) direction across the study area (Bridges 1990).

In addition to the natural environmental variation present within the study region, land uses and land management are also relatively diverse. More than two thirds of the native vegetation of the SEQ region has been cleared since human settlement began (Caterall and Kingston 1993).

Land uses includes urban and industrial areas, forestry in native and plantation forests, national parks, and dry land and irrigated production of sugar, dairy, beef, grain, fruit and vegetables. These are not evenly distributed across the catchments, with more intensive land uses such as horticulture and urbanisation tending to occur in river valleys and on floodplains. Furthermore, localised land management activities such as riparian clearing, riparian grazing, weed control and vegetation replanting and burning may also be undertaken within the stream riparian zones themselves, and can have a direct impact on the riparian vegetation communities and stream ecosystems more generally.

Anthropogenic activities such as agriculture, urbanisation and water management are likely to co-vary with natural environmental variation found within the study region, as many natural factors often determine the suitability of sites for such anthropogenic activities (Allan 2004). A key question for the ELOHA trial, and in fact for the ELOHA methodology as a whole, is therefore whether the influences of flow can be extricated from the influences of landscape-scale environmental variability and anthropogenic disturbances in order to develop the generalised flow – ecology response models for the distinctive river flow classes fundamental to the ELOHA method.

This chapter is divided into two principal sections. The first section describes the methods used for the collation of landscape environmental variables and land use and management data. The second section investigates how landscape and land use variables that are potentially relevant to ecological responses co-vary with each other, with hydrological alteration and across the RFCs (based on modelled IQQM pre-development data) and HFCs (based on gauge data) identified in the hydrological analysis (Chapter 3).

5.2 Environmental variable collation methods

5.2.1 Selection criteria

Landscape environmental variables for use in this study were selected based on:

- Their relevance (based on professional judgement and published research) to one or more of the habitat and/or ecological responses measured here
- Their lack of strong influence from direct short-term effects of flow regime and alterations to flow regimes, although many physical variables may control aspects of stream hydrology (acknowledging that aspects such as channel morphology and substrate attributes may be influenced by flow and hence indirectly influence our measured ecological responses)
- Their improbability of being strongly affected by human activity (at least in the short-term)
- The availability of data with an appropriate, consistent resolution and quality across the entire sampling region
- Independence/lack of correlation between variables (acknowledging that where variables are highly correlated it may be impossible to extricate their individual effects and only one need be selected for inclusion in further analysis).

5.2.2 Data collation

All study sites were located within a Geographic Information System (GIS) (ARCGIS 9.2 Environmental Systems Research Institute ESRI). Physical variables based on upstream catchment and local valley characteristics were acquired or derived from a number of readily available digital datasets. In particular, reach-scale variables were sourced from a recently derived national dataset (Stein et al. 2009). Reach extents were as defined by Stein et al. (2009) where reaches are stream sections (less than 10 km in length) between tributary confluences.

Other datasets included the digitised geology maps at a scale of 1:100 000 and the SEQ Region Geoscience dataset (version 2) (DNRM 2002b). This latter data package was compiled from regional geological mapping at a nominal scale of 1:100 000 conducted by the Geological Survey of Queensland with additional data from university theses and geological exploration company mapping.

5.2.3 Landscape-scale environmental variables

Variables defined as landscape-scale variables and likely to be used in the analysis of ecological responses to flow include climate, catchment, reach and site topography and geological characteristics (Table 5.1).

Climate variables

Reach mean annual temperature, reach hottest month mean temperature and reach coldest month mean temperature variables were acquired from Stein et al. (2009) for individual reaches on which sites were situated. Mean annual rainfall was supplied by the Bureau of Meteorology (2009). Rainfall values are based on a rainfall grid generated using the ANU 3-D Spline. The nearest grid point to each individual site was used in the analysis.

Table 5.1: Landscape-scale environmental variables

Variables	Units	Acronym	Source
Topography and morphology			
Latitude	Degrees	Declat	1
Longitude	Degrees	Declong	1
Site elevation	m.a.s.l	Elevat	1
Distance to source	km	DtoS	1
Distance to mouth	km	DtoM	1
Channel aspect – north	Degrees	C_Asp_N	1
Channel aspect – east	Degrees	C_Asp_E	1
Bank aspect – north	Degrees	B_Asp_N	1
Bank aspect – east	degrees	B_Asp_E	1
Stream gradient	$m.m^{-1}$	S_Grad	2
Stream bank gradient	$m.m^{-1}$	B_Slope	1
Catchment area	km^2	CAT_AREA	1
Elongation ratio	No units	CAT_ELON	1
Relief ratio	No units	CAT_RELI	1
Reach valley confinement	%	V_Conf	2
Climate variables			
Annual mean rainfall	mm	A_Rainfall	3
Annual mean temperature	°C	A_Temp	2
Hottest month mean temperature	°C	HMA_Temp	2
Coldest month mean temperature	°C	CMA_Temp	2
Substrate characteristics			
% mafic	%	Mafic	
% felsic	%	Felsic	4
% sedimentary (% siliclastic and undifferentiated)	%	Sed-Silic	4
% sedimentary (% carbonates)	%	Sed-Carb	4
% mixed sedimentary and igneous	%	Mixed	4
% unconsolidated rocks (alluvium, colluvium etc)	%	Unc_Catch	4
Unconsolidated material for reach	%	Unc_Reach	2

Source data:

1. *Measured in this study*
2. *Stein et al. (2009)*
3. *Bureau of Meteorology (2009)*
4. *Queensland DNRM (2002)*

Topography and morphology

Catchment boundaries for each site were delineated in a GIS using detailed stream networks based on 1:25 000 and 1:100 000 scale maps and a 30 m DEM (digital elevation model) (NASA DTED2 2007). Catchment area was computed using geometry functions of the GIS software. Drainage basin shape was represented by the elongation ratio (Re; the diameter of a circle with the same area as that of the basin divided by the length of the basin) which is considered to have a reasonable correlation with stream hydrology (Morisawa 1958).

Reach morphology variables, stream gradient (elevation difference for each reach divided by its length) and valley confinement (percentage of stream grid cells and their immediate neighbours that are not defined as valley bottoms) were acquired from Stein et al. (2009) and are based on the Multi-resolution Valley Bottom Flatness Index of Gallant and Dowling (2003).

Site topography variables, distance to source, distance to mouth and elevation were determined for each site using GIS with detailed stream networks based on 1:25 000 and 1:100 000 scale drainage maps and the 30 m digital elevation map (DEM). Cross sectional surveys were undertaken at each site to determine channel morphology and used in the calculation of various stream hydraulic parameters.

Surveys were conducted with a dumpy and staff. One cross section was surveyed per site, but additional surveys were conducted in heterogeneous channels. Cross sections were located in riffles and were marked with a 10 cm bolt and quick dry cement at a point above bankfull discharge (Harrelson et al. 1994).

The cross section information was used to determine an average bank slope for the channel side surveyed for riparian vegetation. Aspect was measured with a compass in the field or from the 30 m DEM.

Geology

Substrate geological characteristics were derived for the field sites from the SEQ Region Geoscience dataset (Queensland DNRM 2002) and digital 1:100 000 scale geology maps for the region. Geological groupings were based on broad composition characteristics and followed the classes of Stein et al. (2009) (Table 5.1).

5.2.4 Land use and land management variables

Landscape-scale land use and disturbance for field site catchments were assessed using the Queensland Land Use Mapping Program (QLUMP) dataset (Witte et al. 2006) generated from baseline surveys conducted in 1999. Draft updates available from 2006 surveys for the Maroochy and Logan–Albert were also incorporated (Stanley and Bremer River catchment information were not available at the time of analysis).

The primary land use classes were based on the Australian Land use and Management Classification version 6 (BRS 2002) as these represent broad land use categories differentiating conservation and relatively natural land uses from intensive land uses (Table 5.2). The percentage of each primary land use class within each catchment was calculated as lumped metrics which are non-spatially explicit and treat the whole catchment area with a similar weighting.

Numerous studies, however, suggest that the impacts of land use and land management on streams and their biota are scale dependent (e.g. Allan et al. 1997; Sponseller et al. 2001). As land use close to streams may have a disproportionate affect on stream health and condition relative to land uses distal to the stream (Tran et al. 2010), a distance weighted metric for each primary land use class was also calculated.

In a recent study of relationships between land use metrics and various ecological indicators conducted in SEQ, Peterson et al. (2011) found that distance weighted metrics generally outperformed lumped metrics but no single metric was best overall for the indicators studied. An inverse-distance weighting $(d+1)^{-1}$ metric following Peterson et al. (2011) was therefore chosen.

Land use based on the QLUMP data provides information at a relatively coarse scale (smallest mapped feature is 1 hectare and minimum width for linear features is 50 m). Land management practices at a local scale, however, can have a strong impact upon riparian communities and stream ecosystems through extremely localised activities (e.g. selective weed control, riparian replanting, localised riparian grazing and burning) that are unlikely to be reflected in the broader-scale land use datasets available.

Landholder surveys were undertaken in order to establish the extent of local management activities specific to our riparian surveyed zones (Ethics approval GU Ref No: ENV/36/08/HREC, Attachment 5.1). The information returned was condensed into a number of simple metrics. For sites for which surveys were not returned, the metrics were generated through personal observations and knowledge of the area.

Table 5.2: Primary categories used to describe catchment land uses based on The Australian Land use and Management Classification (version 6)

Primary land use class	Acronym	Subcategories
Production from relatively natural environments	PNE	Grazing natural vegetation, production forestry
Production from dryland agriculture and plantations	PDA	Cropping, grazing modified pastures, horticulture, plantation forestry
Production from irrigated agriculture and plantations	PIA	Irrigated cropping, irrigated modified pastures, irrigated horticulture, irrigated plantation forestry
Conservation and natural environments	CAN	Conservation areas (national parks, nature reserves), other protected resources
Intensive uses	IU	Residential, industrial, transport and utilities, intensive horticulture, intensive animal production

5.2.5 Catchment and land use variation across study sites

This section explores how landscape and land use variables that are potentially relevant to all types of ecological response co-vary with each other, across the RFCs and HFCs identified in the hydrological analysis (Chapter 3) and with hydrological alteration. The focus of this analysis is on landscape variables that may influence the condition of all riparian and aquatic vegetation and fish ecological response metrics. Site variables of specific relevance to individual study components, and usually measured on each sampling occasion, are described within the individual ecological chapters.

5.2.6 Statistical analyses

Relationships amongst landscape and land use were investigated using Pearson's product moment correlation coefficients and scatter plots. Potential correlations between landscape variables and land use and flow regime alteration were examined using the Gower dissimilarity metric based on the differences between the modelled natural (Reference) and gauged (Historic) flow conditions.

PCA was used to investigate patterns in landscape variables across the sites. Prior to the PCA proportional variables (% of different catchment geologies and valley confinement) were arcsine transformed after examination of the raw data. Catchment area (CAT_AREA) variables were log10 transformed. As aspect is a circular variable in that large values are close to small values (e.g. 350° is close to 10°), we used cosine and sine transformations of the channel and bank aspects to give two continuous functions indicating degree shifts from north and east. Other variables remained untransformed.

Variables were scaled to unit variance and zero mean. The significance of the principal components was evaluated using the broken-stick rule (Jackson 1993). Variables that had loadings ≥ 0.5 on the significant principal components identified from the broken-stick rule were examined further. The non-parametric Kruskal–Wallis test was used to test for significant differences in variables identified from the Redundancy Analysis (RDA) between the flow classes.

Where differences between flow groups were identified, a multiple comparison test was performed to determine which flow classes were significantly different. The significance of all tests was assessed using Bonferroni-adjusted significance to account for the number of comparisons.

All analyses were conducted using SYSTAT (version 11.00.01) and R version 2.10.1 (R Development Core Team 2010) with the R packages vegan, (Oksanen et al. 2010), pgirmess (Giraudoux 2010) and gplots (Warnes 2009).

5.3 Results

5.3.1 Landscape variables

Raw site data for landscape variables are presented in Attachment A. A heat map shows the magnitude of Pearson's correlation coefficients amongst landscape variables (Figure 5.1). 'These reveal that, in general, most landscape variables were only weakly correlated with each other.

Most correlation patterns were, predictably, observed within landscape variable types, e.g. climatic (A_temp, CMA_temp and HMA_temp), topographic (DECLONG, Elevat, Cat_Area and Cat_Reli) and geological (Sed_silic) variables.=

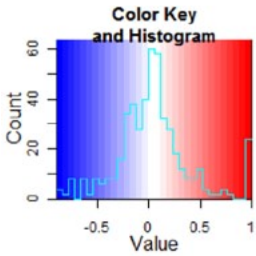
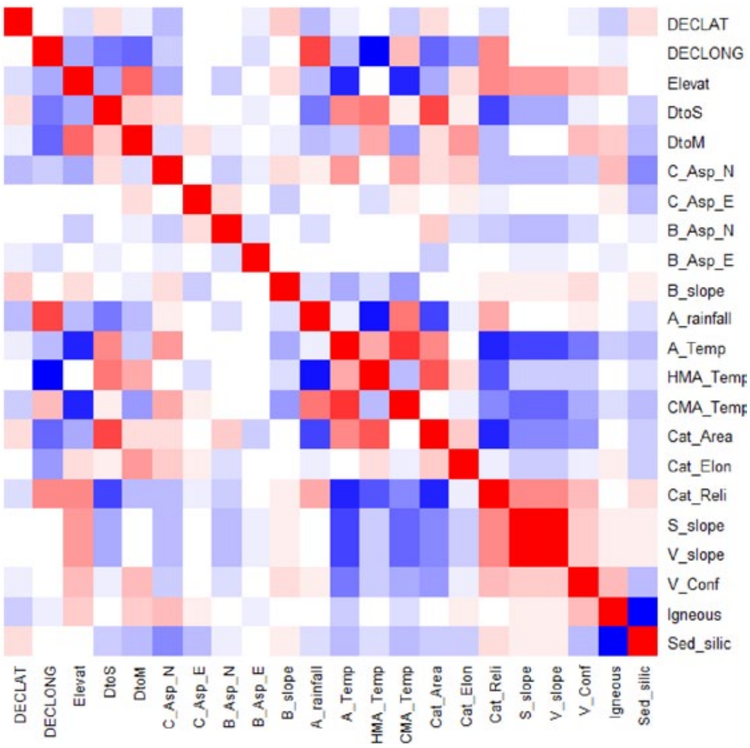


Figure 5.1: Heat map showing the magnitude of Pearson's correlation coefficients between landscape variables

Red and pink indicate positive correlations amongst variables whilst dark blue and light blue indicate negative correlations amongst variables. See Table 5.1 for variable acronyms.



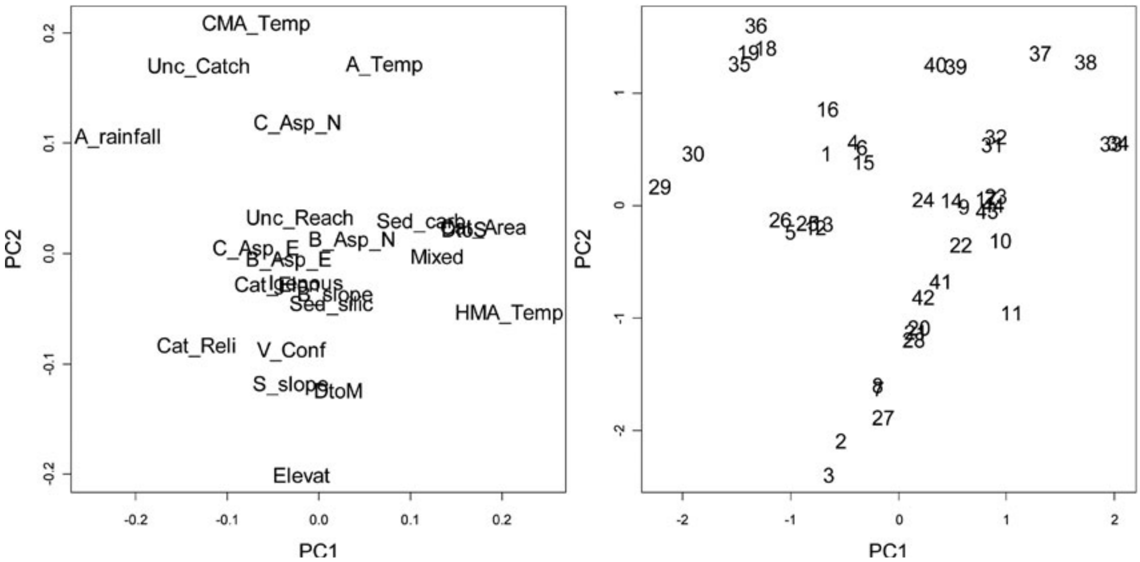
5.3.2 Patterns in landscape variables across sites

PCA of the landscape variables explained 51% of the variance within the first three principal components (Table 5.3, Figure 5.2). PC1 explained just over 20.17% of the variation and was positively associated with one climatic variable (HMA_Temp), two topographic variables (CAT_AREA, DtoS) and two geological variables (Sed_Carb and Mixed). This component was also negatively associated with one climatic variable (A_Rainfall) and one topographic variable (CAT_RELI).

PC2 explained 20.23% of the variance and was positively associated with two climatic variables (A_Temp and CMA_Temp) and one geological variable (Unc_Catch). It was negatively associated with three topographic variables (CAT_RELI, S_Slope and Elevat). PC3 explained 10.6% of the variance and was positively associated with one geological variable (Igneous) and negatively associated with geological variable Sed_Silic. Additional principal components explained less than 10% of the total variance.

Figure 5.2: Varimax rotated PCA score plots of landscape variables and site scores

Landscape variable acronyms are defined in Table 5.1. Principal components 1 and 2 explained 20.17% and 20.23% of the variance, respectively.



The PCA for the catchment characteristics captures the broad geographic and climatic trends found across the study region. Hottest month mean temperature (HMA_Temp) and catchment area (CAT_AREA) had the highest positive loadings on component 1 and were associated with many of the most northerly sites in the study region (e.g. Mary River at Miva, Yabba Creek, Wide Bay Creek and Munna Creek). Catchment relief ratio (Cat_Reli) and mean annual rainfall (A_Rainfall) were negatively loaded on this component and were generally associated with coastal streams (e.g. Currumbin Creek, Eudlo Creek and North Maroochy River).

Mean annual temperature (A_Temp) and coldest month mean temperature (CMA_Temp) had the highest positive loadings on component two and were generally associated with more northerly sites (North Maroochy River, Eudlo Creek, Tinnana Creek and Mary River at Miva). Elevation (Elevat) was strongly negatively loaded on this component and was associated with sites in the south-west of the study region (e.g. Burnett Creek).

Table 5.3: Variable loadings from PCA of landscape variables

See Table 5.1 for variable acronyms.

Variable	Component and variation explained		
	PC1 (20.2%)	PC2 (20.2%)	PC3 (10.6%)
Elevat	-0.112	-0.897	0.050
DtoS	0.745	0.200	0.121
DtoM	0.268	-0.460	0.252
C_Asp_N	0.054	0.484	0.414
C_Asp_E	-0.086	0.037	0.192
B_Asp_N	0.198	0.077	-0.063
B_Asp_E	-0.082	0.001	-0.131
B_Slope	0.013	-0.237	-0.029
A_Rainfall	-0.840	0.355	0.166
A_Temp	0.440	0.848	0.030
HMA_Temp	0.897	-0.096	-0.016
CMA_Temp	-0.138	0.921	0.139
CAT_AREA	0.850	0.224	0.020
CAT_ELON	0.076	-0.064	0.144
CAT_RELI	-0.693	-0.501	-0.001
S_Slope	-0.284	-0.605	0.051
V_Conf	-0.175	-0.466	0.437
Igneous	0.029	-0.206	0.900
Sed_Silic	-0.118	-0.161	-0.929
Sed_Carb	0.521	0.180	0.154
Mixed	0.642	0.095	0.311
Unc_Catch	-0.429	0.733	-0.199
Unc_Reach	0.143	0.247	-0.187

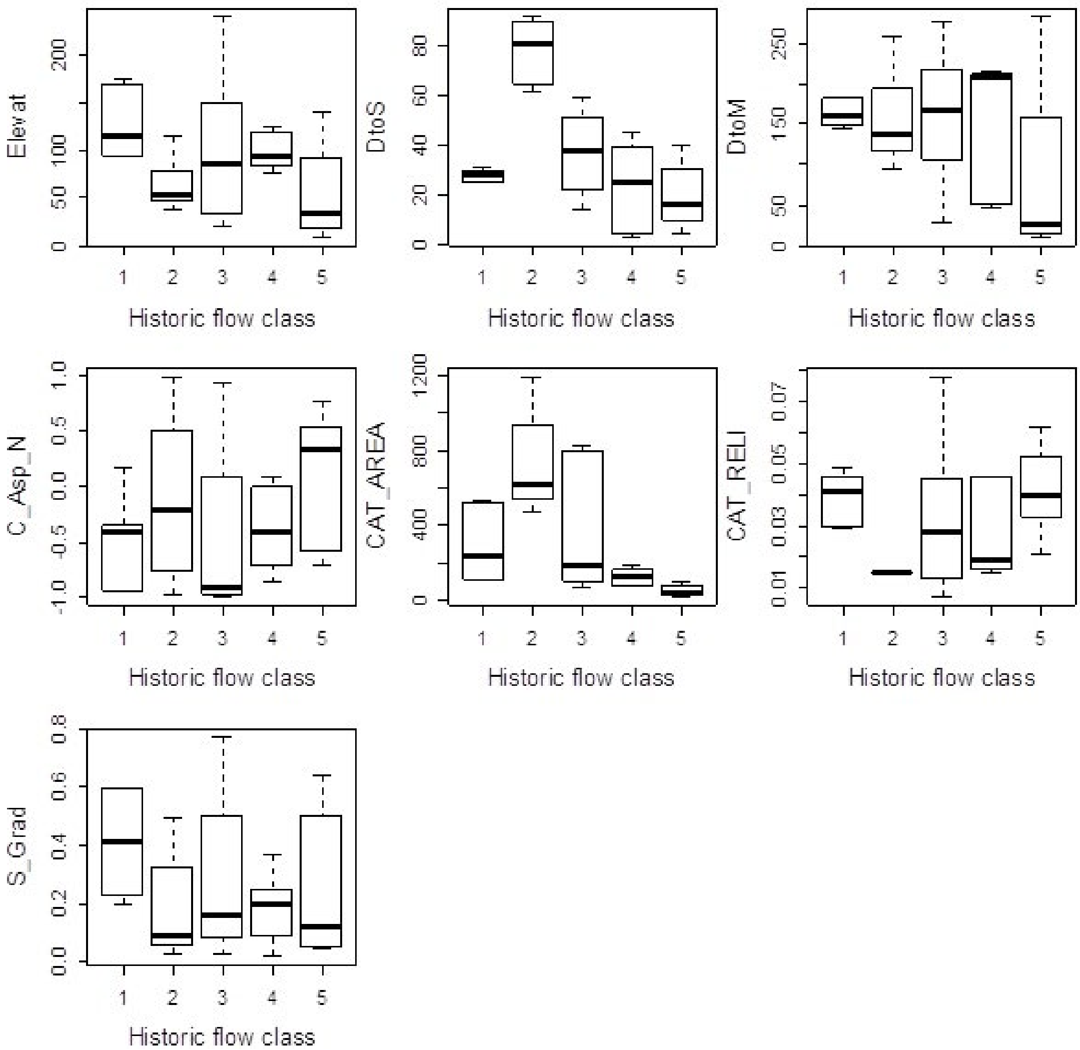
5.3.3 Patterns in landscape variables across flow classes

Landscape variables were compared across the IQQM and gauge flow classes using the non-parametric Kruskal–Wallis test (Bonferroni-adjusted significance = 0.005). Significant differences were identified between HFCs for variables: DtoS, Cat_Area, Elevat, A_Rainfall and Unc_Catch. For the Reference classification significant differences were found for variables: Elevat, C_Asp_N, Cat_Area, A_Rainfall, A_Temp, HMA_Temp, CMA_Temp and Sed_Carb.

Amongst the topographic variables, significant differences were detected between gauge classes 2 and, 3 and 5 for distance to source (DtoS) (Figure 5.3). Catchment area was found to be significantly smaller in HFC 5 compared with HFCs 1 and 2 (Figure 5.3). Catchment areas were significantly greater in RFC 4 compared with RFC 5 (Figure 5.4). No significant differences in elevation were found between the RFCs or HFCs. Significant differences in channel aspect (northerly) were found between RFCs 2 and 4 (Figure 5.4).

Figure 5.3: Box plots of selected topographic variables for HFCs

Significant differences using the Kruskal–Wallis test (Bonferroni-adjusted significance = 0.005) were identified between HFCs for variables Elevat, DtoS and Catch_Area. Pairwise differences amongst classes are indicated in the relevant plots.



Climate variables were also found to be significantly different between flow classes. Annual rainfall (A_rainfall) was significantly higher in HFC 5 compared with HFC 2. Differences were also detected between annual rainfall between RFC 5 and, RFCs 2 and 4. Hottest month mean temperatures (HMA_Temp) were significantly lower in RFC 5 compared with RFCs 2 and 4. Coldest month mean temperatures (CMA_Temp) were significantly lower in RFC 2 compared with RFC 5.

Differences between flow classes were only detected for two geological variables. HFC 4 has a significantly lower proportion of unconsolidated materials (Unc_Catch) in its catchment compared with HFC 5. Although differences in IQQM classes were detected using the Kruskal–Wallis test for Sed_Carb (sedimentary rocks (carbonates)), no pairwise differences between individual RFCs were detected for the proportion of Sed_Carb in the catchments.

Figure 5.4: Box plots of selected topographic variables for RFCs

Significant differences using the Kruskal–Wallis test (Bonferroni-adjusted significance = 0.005) were identified between RFCs for variables Elevat, C_Asp_N and Cat_Area. Pairwise differences amongst classes are indicated in the relevant plots.

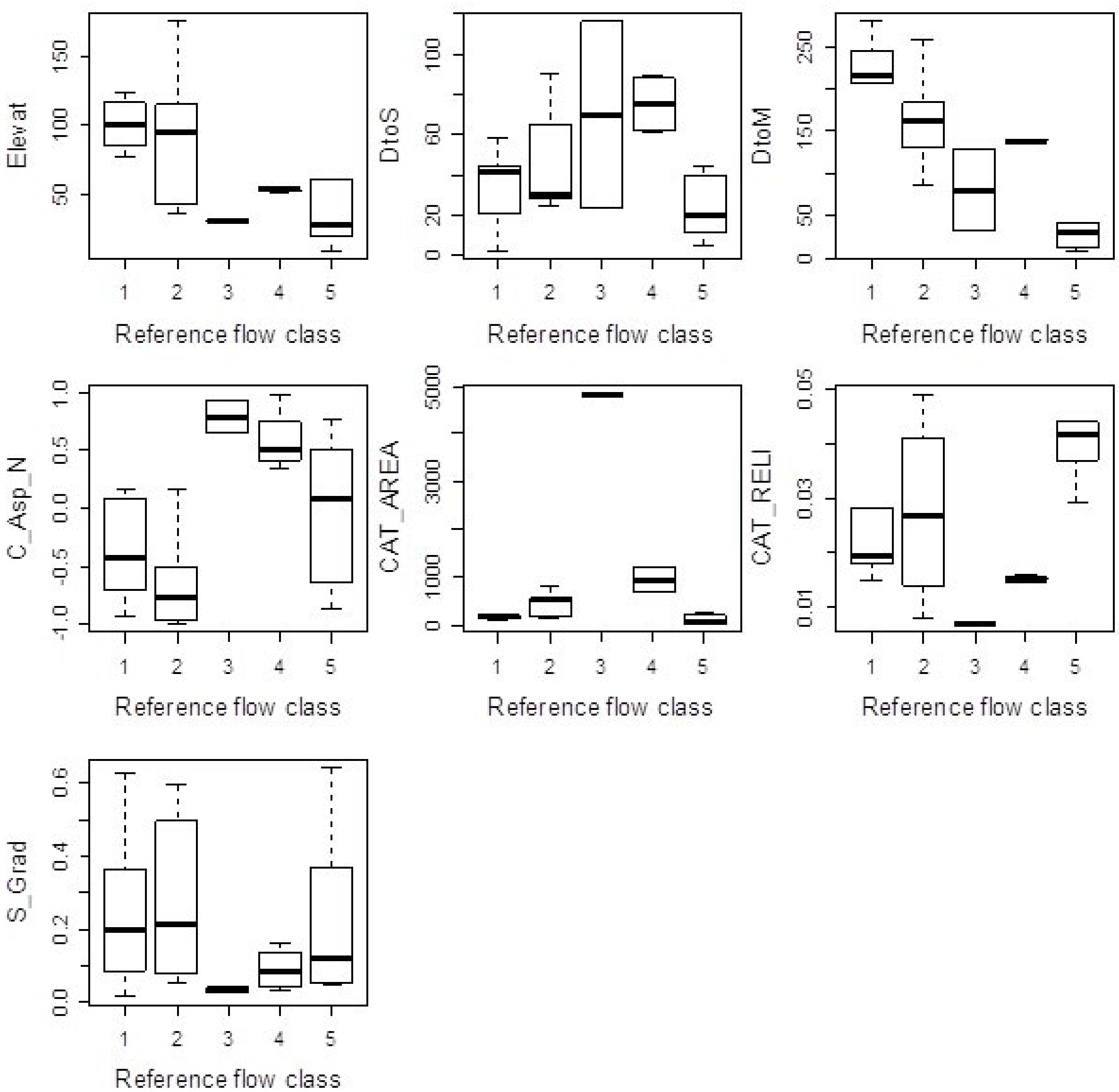


Figure 5.5: Box and whisker plots of climatic variables for RFCs and HFCs

Significant differences using the Kruskal–Wallis test (Bonferroni-adjusted significance = 0.005) were identified between HFCs for variable A_rainfall and between RFCs for variables A_rainfall, A_Temp, HMA_Temp and CMA_Temp. Pairwise differences amongst classes are indicated in the relevant plots.

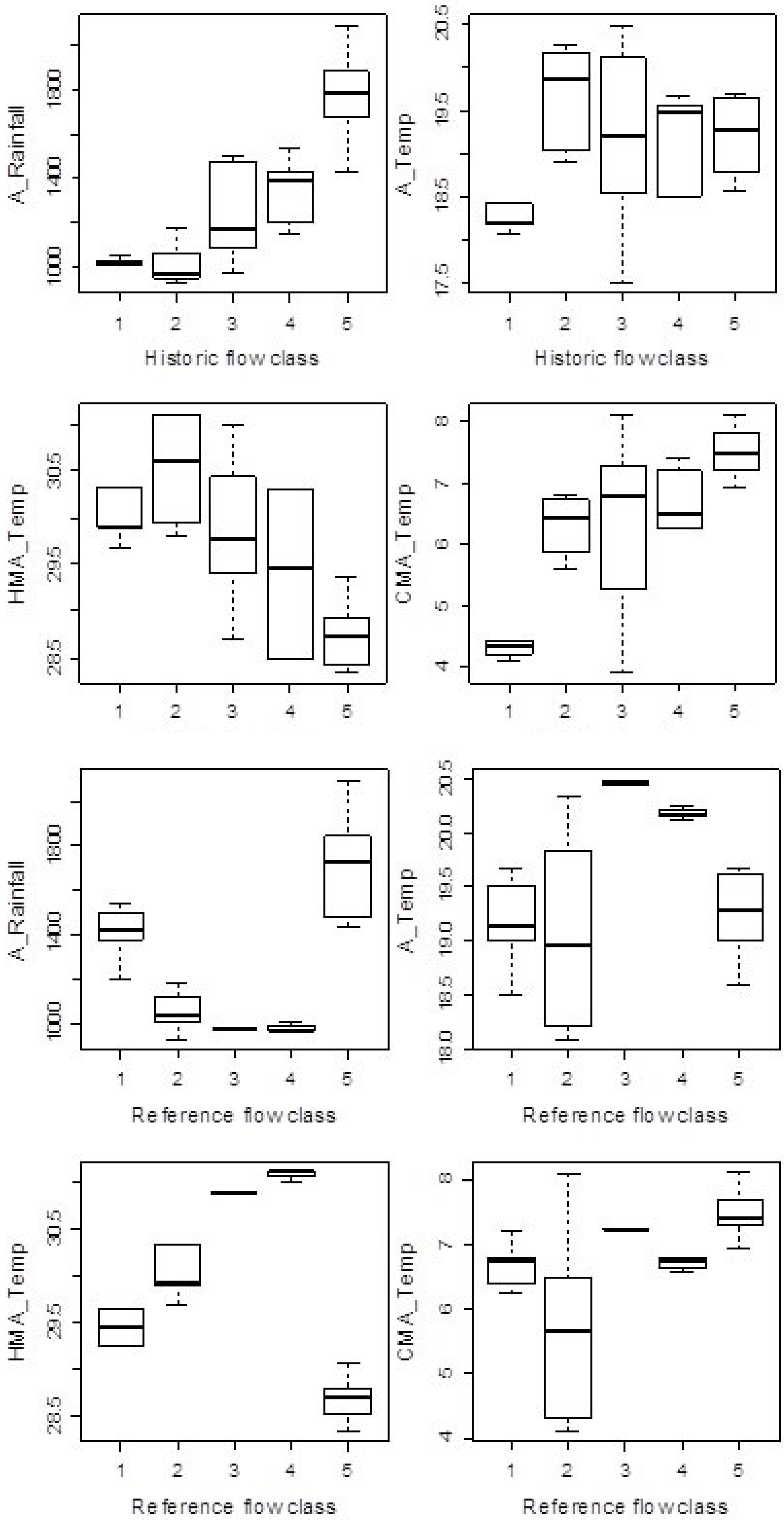
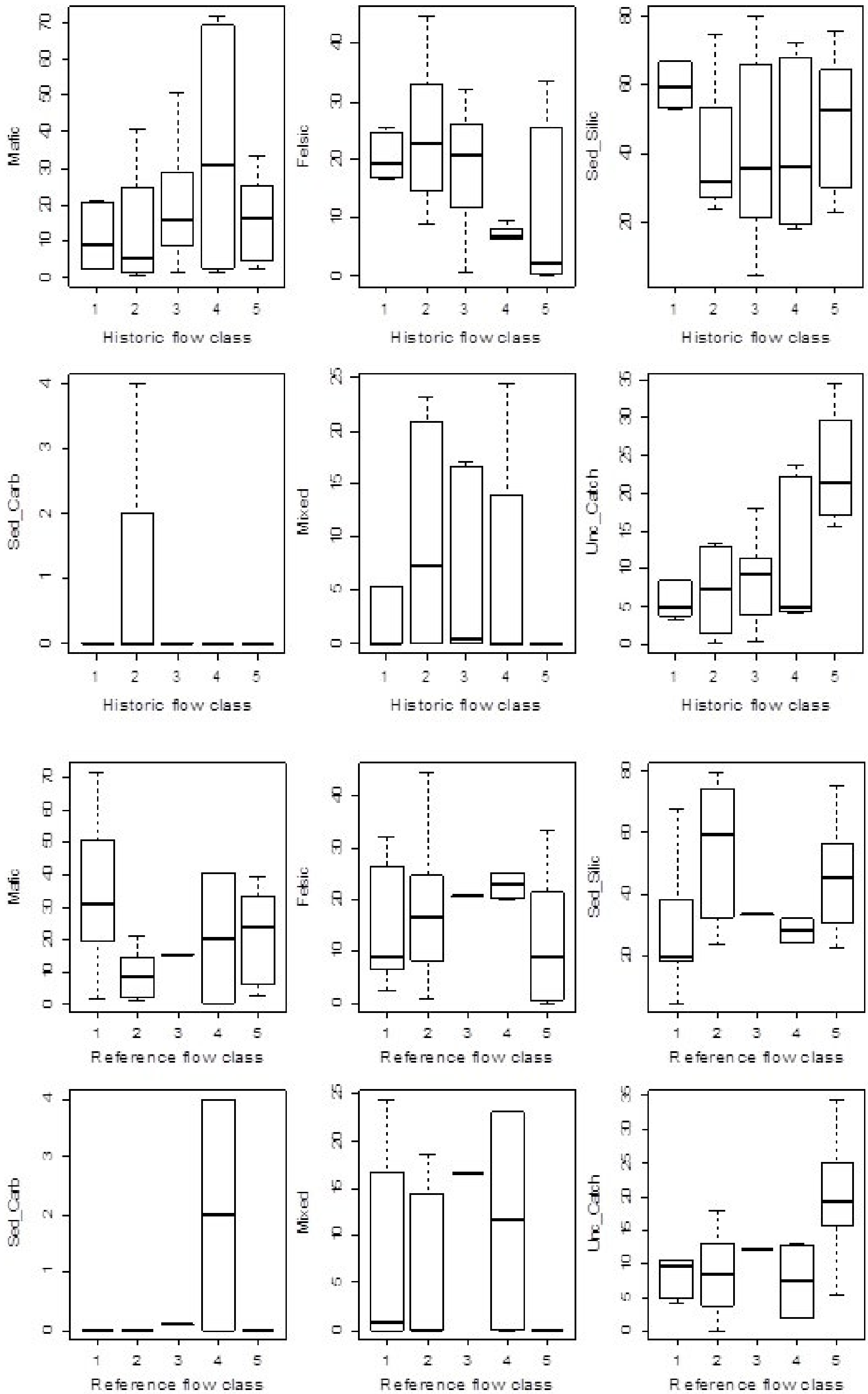


Figure 5.6: Box and whisker plots of selected geological variables for RFCs and HFCs

Significant differences using the Kruskal–Wallis test (Bonferroni-adjusted significance = 0.005) were identified between HFCs for variable Unc_Catch and between RFCs for variable Sed_ Carb. Pairwise differences amongst classes are indicated in the relevant plots.



5.3.4 Land use metrics

Land uses within the study site catchments were predominantly composed of 'Production from relatively natural environments' which accounted for 58% of the total land uses within the 44 site catchments, with 'Conservation and natural environments' accounting for 26% of the catchment land uses. The remaining land uses (Intensive uses, Production from dryland agriculture and plantations and Production from irrigated agriculture and plantations) (excluding water bodies) were 10%, 5% and 1% of the site catchment areas, respectively.

Land use metrics were generally only weakly correlated with each other (Table 5.4) with the strongest negative correlations observed between classes 'Conservation and natural environments' and 'Production from relatively natural environments metrics'. IDW and lumped metrics were, however, strongly correlated with each other within the land use classes (Table 5.4). Examination of the IDW compared with the lumped metric for the 44 site catchments revealed some predictable patterns.

Using a weighted metric decreased the relative importance of the land use class 'Conservation and natural environments' for 36 out of the 44 site catchments. Using the weighted metric also increased the relative importance of the class 'Production from irrigated agriculture and plantations' for the majority of those catchments (25 out of the 34 sites) with this land use type compared with the lumped metric.

Of the remaining classes both the 'intensive use' class and 'Production from relatively natural environments' land use class increased in importance for just over half of the site catchments (26 out of 44 sites and 23 out of 39 sites respectively) relative to the lumped metrics whilst the class 'Production from dryland agriculture and plantations' decreased in importance for more than half the site catchments for which this land use was recorded (17 out of the 32 site catchments for which this land use was recorded).

Table 5.4: Pearson's Correlation Coefficients for lumped (L) and Inverse Distance Weighted (IDW) for primary land use class

Key: PDA = Production from dryland agriculture and plantations, IU = Intensive Uses, CAN = Conservation and natural environments, PIA = Production from irrigated agriculture and plantations, PNE = Production from relatively natural environments.

Metric		PDA		IU		CAN		PIA		PNE	
		L	IDW	L	IDW	L	IDW	L	IDW	L	IDW
PNE	IDW	-0.424	-0.418	-0.545	-0.607	-0.636	-0.444	0.094	0.105	0.916	1
	L	-0.289	-0.241	-0.475	-0.504	-0.761	-0.665	0.181	0.081	1	
PIA	IDW	-0.096	-0.092	-0.109	-0.126	-0.083	-0.017	0.715	1		
	L	0.173	0.192	0.088	-0.028	-0.335	-0.409	1			
CAN	IDW	-0.221	-0.230	-0.168	-0.034	0.910	1				
	L	-0.067	-0.089	-0.002	0.067	1					
IU	IDW	-0.031	-0.066	0.906	1						
	L	0.040	0.018	1							
PDG	IDW	0.944	1								
	L	1									

5.3.5 Patterns in land use metrics across flow groups

As lumped and IDW metrics were highly correlated, analysis across flow groups is only presented for IDW metrics in this section. Results for the lumped metrics were the same. Land uses were relatively uniform across both the gauge and flow groups with only land use class ‘ Production from relatively natural environments’ showing significant differences between HFCs 1 and 2.

Figure 5.7: Box and whisker plots of land use variables for HFCs

Significant differences using the Kruskal–Wallis test (Bonferroni-adjusted significance = 0.005) were identified between HFCs for PNE. Pairwise differences amongst classes are indicated in the relevant plot.

Key: PDA = Production from dryland agriculture and plantations, IU = Intensive Uses, CAN = Conservation and natural environments, PIA = Production from irrigated agriculture and plantations, PNE = Production from relatively natural environments.

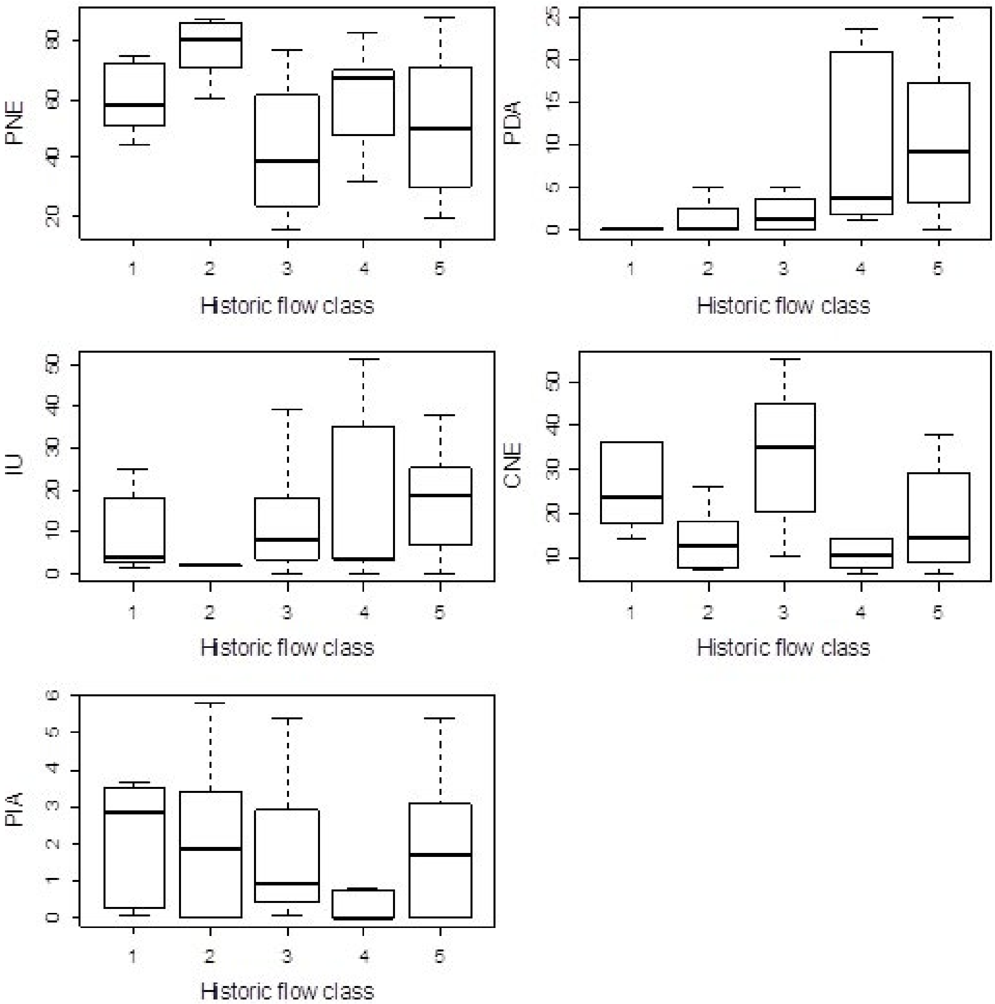
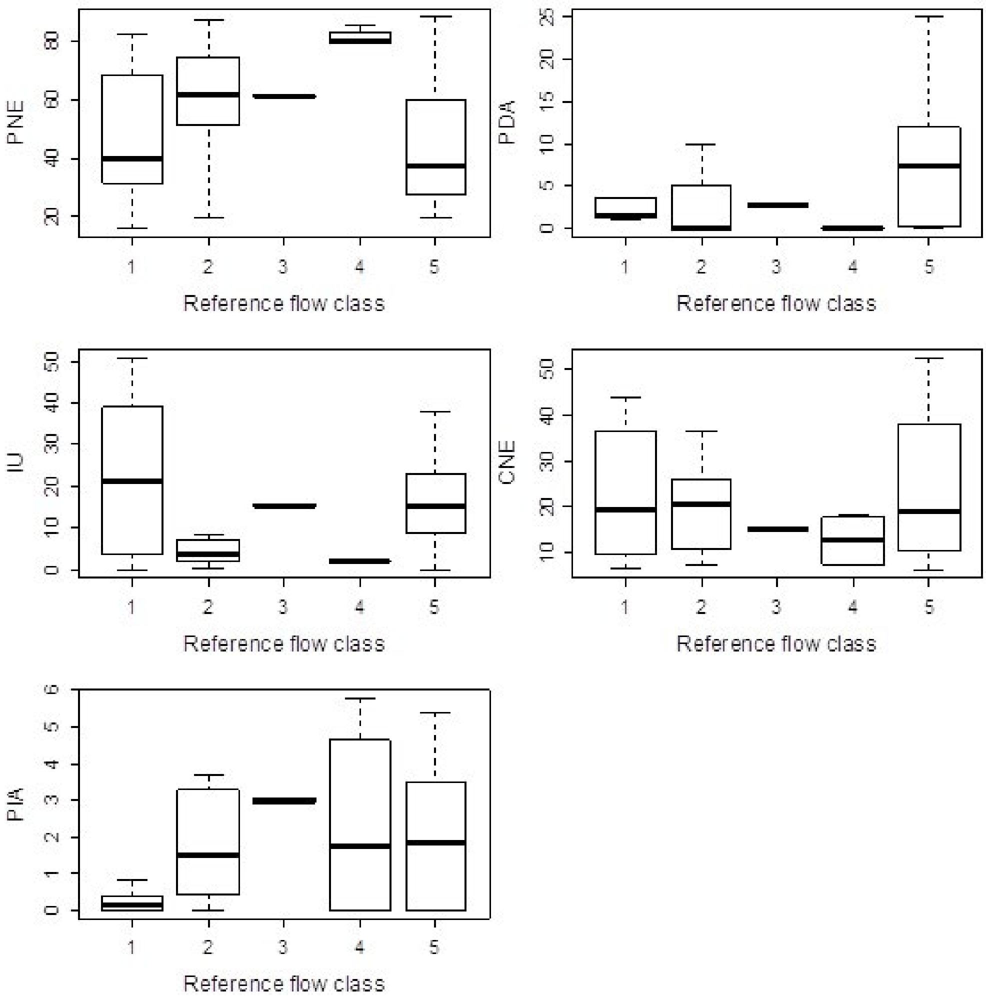


Figure 5.8: Box and whisker plots of land use variables for RFCs

No significant differences were detected between RFCs for any of the land use classes. Key: PDA = Production from dryland agriculture and plantations, IU = Intensive Uses, CAN = Conservation and natural environments, PIA = Production from irrigated agriculture and plantations, PNE = Production from relatively natural environments.



5.3.6 Relationships between land use and flow regime alteration

Relationships between land use in the catchment and degree of flow regime alteration were examined using scatter plots and Pearson's Correlation Coefficients. Measures of flow alteration were based upon the Gower dissimilarity metric calculated using Reference (pre-development IQQM) and Historic (gauge) data.

Relationships were examined based on both the overall flow dissimilarity across all the flow metrics and dissimilarity in individual metrics. Individual flow metrics selected for analysis were those identified by the *clustvarsel* routine as being important in informing the structure of the two flow classifications.

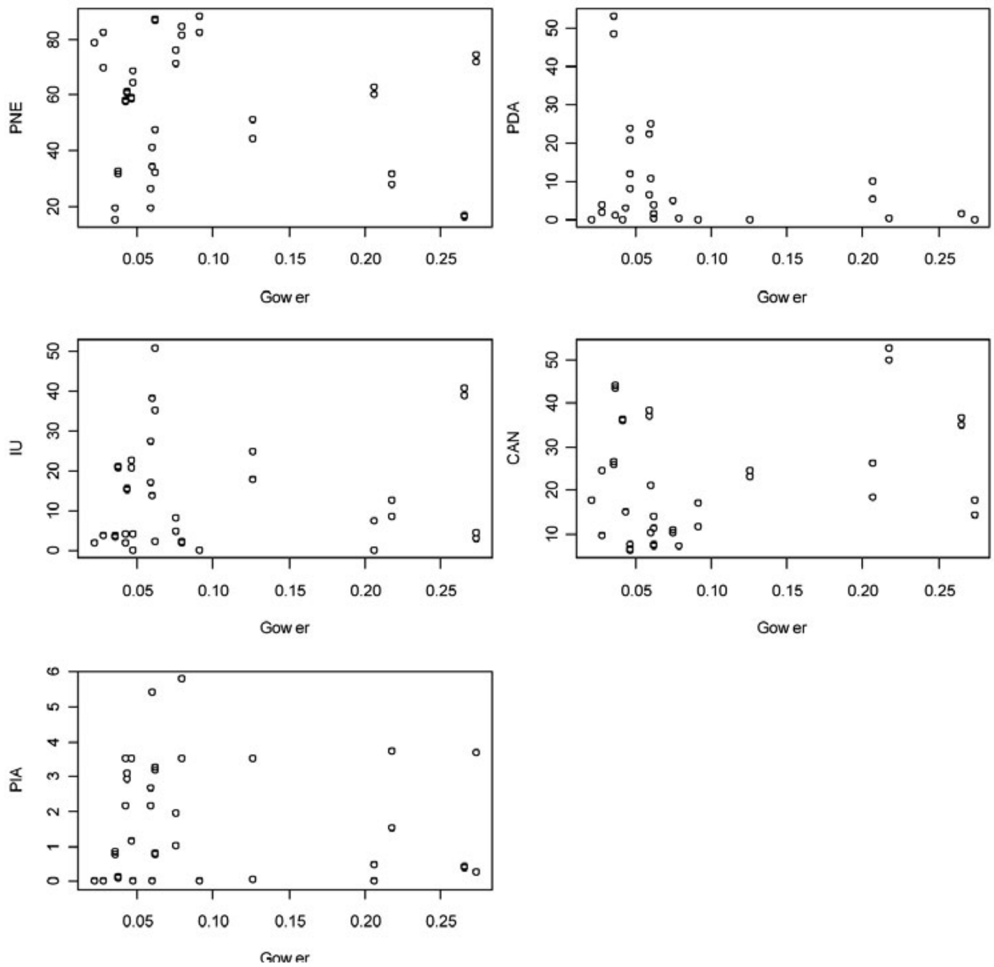
Table 5.5: Pearson's correlation coefficients between land use classes and Gower dissimilarity metric calculated between Reference and Historic flow metrics

Sites 2, 3 (Burnett Creek) and 7, 8 (Teviot Brook) are excluded as Reference flow data were not available for these sites.

Land use classes	Lumped	IDW
Production from relatively natural environments	-0.164	-0.181
Production from dryland agriculture and plantations	-0.247	-0.265
Intensive uses	0.087	0.137
Conservation and natural environments	0.263	0.297
Production from irrigated agriculture and plantations	-0.274	-0.012

Figure 5.9: Relationships between land use classes and degree of flow regime alteration

Flow regime alteration is based on the Gower dissimilarity metric calculated between Reference (pre-development IQQM) and Historic (gauge) data (Chapter 3). Sites 2, 3, 7 and 8 are excluded as IQQM data were not available for these sites.



Correlations between the two disturbance types (land use and overall flow regime alteration) were generally very weak with highest correlations observed between the Gower metric and the proportion of land in the class ‘ Conservation and natural environments’ (0.263 and 0.297 for IWD and lumped metrics respectively) (Table 5.5 and Figure 5.9).

An examination of correlations between alterations in individual flow metrics and land use metrics (Table 5.6) revealed only weak correlations between types of flow alteration and proportion of different land uses within the catchments. The highest correlation coefficients were negative with an inverse correlation between the change in ARI (1 year and 10 years) and the proportion of intensive land use in the catchments.

5.4 Discussion

Stream flows are recognised as one of the principal drivers of stream ecology (Poff et al. 1997; Bunn and Arthington 2002). However, many other catchment characteristics (catchment size, shape, geology and topography) not necessarily related directly to stream flows are also important drivers of stream ecological processes. The possible influence of these variables must be teased out from the direct influences of variations in the flow characteristics of study sites in order to develop robust flow–ecology relationships for natural and regulated streams.

The broad patterns of variation we have recorded in many of these variables are to be expected, and the details are interesting and important to the ELOHA study. In summary, the PCA for the catchment characteristics has captured the broad geographic and climatic trends found across the study region.

Hottest month mean temperature (HMA_Temp) and catchment area (Cat_AREA) had the highest positive loadings on component 1 and were associated with many of the most northerly sites in the study region (e.g. Mary River at Miva, Yabba Creek, Wide Bay Creek and Munna Creek). Catchment relief ratio (Cat_Reli) and mean annual rainfall (A_Rainfall) were negatively loaded on this component and were generally associated with coastal streams (e.g. Currumbin Creek, Eudlo Creek and North Maroochy River).

Mean annual temperature (A_Temp) and coldest month mean temperature (CMA_Temp) had the highest positive loadings on component 2 and were generally associated with more northerly sites (North Maroochy River, Eudlo Creek, Tinana Creek and Mary River at Miva). Elevation (Elevat) was strongly negatively loaded on this component and was associated with sites in the south west of the study region (e.g. Burnett Creek).

The influence of these patterns of variation in catchment and climatic variables on ecological patterns and processes must be teased out during the analysis of biological data (Chapters 6–8).

Table 5.6: Pearson's correlation coefficients between land use classes (IDW and Lumped) and Gower dissimilarity calculated between Reference and Historic flow metrics for those flow metrics identified as discriminating between flow classes within each classification (see text for details)

Sites with correlation coefficients >0.4 or <-0.4 are highlighted in bold. Sites 2, 3, 7 and 8 are excluded as Reference flow data were not available for these sites. Key: PDA = production from dryland agriculture and plantations; IU = intensive uses; CAN = conservation and natural environments; PIA = production from irrigated agriculture and plantations; PNE = production from relatively natural environments.

Flow metrics	IDW Land use					Lumped land use				
	PNE	PDA	IU	CAN	PIA	PNE	PDA	IU	CAN	PIA
ARI_1yr	0.436	-0.222	-0.459	-0.129	0.023	0.434	-0.214	-0.609	0.032	-0.168
ARI_10yr	0.381	-0.073	-0.464	-0.130	0.149	0.405	-0.114	-0.652	0.048	0.000
MedAnnMax	0.035	-0.301	0.001	0.174	-0.166	0.031	-0.320	-0.166	0.282	-0.409
MA1dayMin	0.051	-0.018	-0.087	-0.014	0.226	0.010	-0.019	-0.019	0.005	0.111
MA90dayMin	-0.099	0.090	0.093	0.044	-0.003	-0.153	0.098	0.107	0.059	0.179
MeanZeroDay	-0.157	0.129	0.124	0.057	-0.176	-0.149	0.096	0.078	0.098	-0.313
LSDur	-0.310	0.127	0.185	0.315	-0.151	-0.440	0.154	0.237	0.318	-0.124
HSDur	-0.219	0.210	0.104	0.162	0.055	-0.318	0.204	0.235	0.116	0.232
CONSTAN	-0.196	0.176	0.458	-0.200	-0.414	-0.205	0.192	0.525	-0.223	-0.157
MDF_Mar	0.169	-0.101	-0.364	0.130	0.081	0.206	-0.137	-0.542	0.225	-0.107
MDF_Sep	0.108	0.010	-0.121	-0.047	0.139	0.031	0.089	0.009	-0.112	0.316
MA1dayMax	-0.052	-0.061	0.001	0.163	-0.024	-0.105	-0.087	-0.120	0.283	-0.015

Given that catchment size, shape and topography are important influences upon stream number, size, water yield and hydrograph shape (Gordon et al. 2005), it is unsurprising that differences in morphology and catchment variables should be detected amongst flow classes. In fact, it is surprising that differences were not detected amongst a greater number of variables.

Many natural variables such as those describing catchment topography, geology, soils and climate are likely to be drivers of the hydrology regimes (Poff et al. 1997; Kennard et al. 2010a) and, as such, one would expect these to vary in a predictable manner with the natural flow regimes. However, the IQQM models used to generate the natural flow regimes (and subsequent flow classes defined in this study) do not take into account factors such as land use and vegetation changes (Bronstert et al. 2002), and have limited capacity to model groundwater interactions which can strongly influence stream hydrologies.

A key question for this trial of the ELOHA framework is whether hydrological Reference sites (based on classification of IQQM flow data) are appropriate References with respect to other key landscape and land use characteristics. Fortunately for this study region, few differences in land uses were found across the hydrological groups and only weak correlations between flow alteration and different land uses.

These results suggest that the development of generalised flow–ecology relationships across gradients of flow regime alteration is unlikely to be compromised by other co-varying disturbances within the catchments selected for study. The possible effects of these disturbances must still be considered, but may not directly influence any differential ecological responses among the flow regime classes.

5.5 Attachments

Attachment 5.1: Cover letter and survey sent to landholders

Australian Rivers Institute
Nathan Campus, Griffith University
170 Kessels Road, Nathan
Brisbane, Q 4111

28th January 2010

Dear Landholder

Thank you for participating in our research project by providing our research team with access to the Eudlo River via your property. Your involvement to-date has greatly assisted our project.

For your information, please find attached the riparian vegetation species we have identified from our surveys on your property.

As we briefly discussed with you on the phone, please find attached a questionnaire which seeks your knowledge on land use and management history of field sites we have surveyed on your property. We may have already discussed some of these issues with you during our field visits, however, if you could briefly complete the survey attached we would be very grateful. We may also contact you upon completion of the survey for further clarification of answers provided. Below you will find information on our project and contact details if you have any further questions or queries.

Project Title	Hydro-ecological relationships and thresholds to inform environmental flow management and river restoration
Research team	Professor Angela Arthington Dr Cassie James Dr Stephen Mackay Ms Anna Barnes Mr David Sternberg
Contact details	Australian Rivers Institute, Griffith University, Nathan QLD 4111 Tel. 07 3735 7402, Fax 07 3735 7615

Why is this research being conducted?

This project is investigating relationships between river flows and the ecology of rivers in south east Queensland. Researchers from the Australian Rivers Institute at Griffith University are undertaking field studies on fish and vegetation in natural rivers and rivers that have been regulated by dams to try and understand how they respond to changes in stream flows.

A critical aspect of this research is concerned with understanding how other influences such as historical vegetation clearance, grazing, weed control and burning affect river ecology. With the information provided from this survey, researchers will attempt to tease out the effects due to changes in stream flows from other influences.

What we are asking you to do?

You are asked to complete a short questionnaire with questions relating to historical vegetation clearance, grazing, burning and any weed control that you have undertaken at a selected site (or sites) neighbouring a river on your property. Along with the questionnaire, a map is provided indicating the location of the site(s) of interest.

The basis by which participants are selected

As part of this project over 50 sites are being surveyed throughout south east Queensland during 2008 and 2009. You were chosen for participation in the survey as a landholder with land neighbouring a river and/or because you may have relevant knowledge of historical land management in the area.

The expected benefits of the research

The information collected from this research will be used to assist us in understanding how rivers in the south east Queensland region have changed in response to regulation by dams. The information will be used to inform the management of rivers and provide practical advice to water management agencies and individuals so as to achieve healthier rivers.

Your confidentiality

The information collected during this questionnaire will be used in broader analysis across the whole of the south east Queensland region. Individual participants will not be identifiable in any publication or reporting.

Your participation is voluntary

Your participation in this short questionnaire is completely voluntary and you are free to withdraw from the study at any time.

Questions / further information

If you have any questions or queries regarding this project please contact:

- Ms Anna Barnes Tel. (07) 3735 3993, email: anna.barnes@griffith.edu.au
- Dr Cassie James email: c.james@griffith.edu.au.

The ethical conduct of this research

Griffith University conducts research in accordance with the National Statement on Ethical Conduct in Research Involving Humans. If you have any concerns or complaints about the ethical conduct of this research project please contact the Manager, Research Ethics on (07) 3735 5585 or research-ethics@griffith.edu.au.

Feedback to you

Results from this study will be available through the project final report and scientific publications. In addition, participants in the questionnaire will be provided with a summary of the key study findings along with specific ecological information/data relating to sites on their properties collected during field visits (such as fish and vegetation species lists).

Privacy Statement

The conduct of this research involves the collection, access and/or use of your identified personal information. The information collected is confidential and will not be disclosed to third parties without your consent, except to meet government, legal or other regulatory authority requirements. A de-identified copy of this data may be used for other research purposes. However, your anonymity will at all times be safeguarded. For further information consult the University's Privacy Plan at www.gu.edu.au/ua/aa/vc/pp or telephone (07) 3735 5585.

Consent

If you are willing and able to participate in this questionnaire, please detach and retain this information sheet and, complete and return the attached questionnaire in the stamped addressed envelope provided or by return email. The return of a completed, or part-completed, questionnaire will be accepted as an indication that you have consented to participate in this research.

Once again, thank you for supporting our project. We look forward to receiving your reply and please feel free to contact either Anna or Cassie if you have any further questions or comments.

Kind regards,

Anna Barnes
Research Assistant, Australian Rivers Institute
Griffith University, Nathan Campus, Brisbane
Office: (07) 3735 399

Please return survey in envelope provided:
Australian Rivers Institute
c/o Griffith University

Attention: Dr Steve Mackay, N55 Mail Box 72

Landholder Survey – National Water Commission Project

Property details	
Name of land occupant or holder:	Length of time property owned:
Dominant type of farming or land use (please circle one or more as appropriate): • Beef • Dairy • Horticulture • Cropping • Horse adjistment Other (please indicate):	

Section 1 – Current and historical land management
Please see attached map for area of interest. We are interested in the area(s) of your property indicated on the map that front a river or stream. Please circle the appropriate answer or provide further details where indicated.
Native vegetation clearance
1. Has the area indicated on the map previously been cleared of native vegetation? • YES • NO • UNSURE
2. If the area was cleared, to what extent was it cleared? • Totally clear-felled (removal of all trees and shrubs) • Partially cleared (with major canopy trees remaining)
3.. If the area was cleared, what was the approximate year in which it was cleared?
4. Do you control vegetation regrowth in the river frontage area indicated on the map (e.g. thinning of woody regrowth by removing some young vegetation) • YES • NO
5. How do you control vegetation regrowth in the river frontage? • Fire • Grazing • Mechanical or physical removal • Chemical control • Other (please provide details):
Planting or revegetating
6. Do you undertake any planting or revegetation in the river frontage area indicated on the map? • YES • NO
7. On what date(s) (month/year) was revegetating last undertaken?
8. Which species or types of species have you planted?
9. What method(s) have you used for revegetating? • Seedling planting • Direct seeding

Grazing
10. Do stock have access (including occasional access) to the river frontage area indicated on the map? • YES • NO
11. If the river frontage area is grazed, what type(s) of stock have access to this area? (please circle more than one if appropriate) • Cattle • Sheep • Horses • Other (please specify):
12. If the river frontage area is grazed could you indicate how often this area is grazed? • Continuously (This area is stocked all year) • Rotational (Livestock are periodically removed to allow pastures to regrow) • Opportunistic (Stock are occasionally introduced to control weeds for example)
13. If the river frontage area is grazed could you indicate a stocking rate (as an approximate number of animals per hectare)?
14. Has the river frontage area indicated on the map been grazed historically? • YES • NO • UNSURE
15. If you know that the river frontage area indicated was grazed historically but is no longer grazed, could you indicate what year grazing ceased?
Weed control
16. Have you undertaken any weed control in the river frontage area indicated on the map? • YES • NO
17. What weeds have you attempted to control?
18. How have you attempted to control these weeds? (please circle more than one option if appropriate) • Grazing • Biological control • Fire • Mechanical or physical removal • Chemical control • Other (please give details):
19. How many times a year do you undertake weed control?
20. On approximately what date (month/year) did you last undertake weed control in the area indicated?
Burning
21. Have you undertaken any burning in the river frontage area indicated on the map? • YES • NO
22. If the area has been burnt could you indicate when (month/year) this area was last burned?
Earthworks
23. Have you undertaken any major earth-works (for example river bank reinforcements) on the river frontage area indicated on the map? • YES • NO
24. Could you briefly describe the earth works you have undertaken?

Section 2 – General river condition on your property
The following questions relate to **ALL** areas of your property that front a river or stream. Please circle those answers that apply and provide further details where you are able to. If you wish, you can illustrate the enclosed map to indicate locations and provide comments relating to the questions below.
25. Is there any evidence of river bank erosion in areas of your property that front rivers? • YES • NO If Yes could you provide more details (for example, its location, the time period over which you have observed erosion occurring):
26. Is there evidence of sand or sediment accumulation in the river bed? • YES • NO If YES could you provide more details (for example, its location, over what time period you have observed sediment accumulation occurring):
27. Do you think that the river channel on your property has changed shape (for example, become deeper, shallower, wider or narrower)? • YES • NO If YES could you provide more details of how it's changed shape:
28. Have you observed any changes in river water quality (for example, changes in water clarity and algal blooms)? • YES • NO If YES could you provide more details:
29. Is there evidence of excessive growth of weeds in the river channel? • YES • NO If YES could you provide more details (for example, the species, when you first observed the weed growth, their locations):
30. Is there evidence of excessive growth of weeds on the river edges or margins? • YES • NO If YES could you provide more details (for example, which species, their locations):
31. Have you noticed any evidence of changes in exotic or unwanted species (animals and plants)? • YES • NO If YES could you provide more details (the species, their locations, increases or decreases in abundance):
32. Have you noticed any evidence of changes in native species (animals and plants)? • YES • NO If YES could you provide more details (the species, their locations, increases or decreases in abundance):
Is there anything else that you would like to add?

Thank you for your participation

6. Riparian vegetation

6.1 Introduction

The ELOHA framework is a new approach for assessing and managing the flow regime necessary to maintain a healthy ecosystem through the development of regional environmental flow standards (Poff et al. 2010). By flow regime we mean the pattern of flow, including seasonality, predictability and the frequency, timing and duration of specific flow events. Its utility is in being able to define flow requirements across large regional areas where neither time and/or resources are available to define environmental flow requirements on an individual river basis.

Existing information (hydrological and ecological) for rivers within the region is used to develop flow alteration – ecology response relationships for rivers with different flow regime 'types' – this information can then be used to define an acceptable range or threshold and inform environmental flow recommendations for the region as a whole.

The ELOHA method assumes that flow is a key determinate of the ecological community and therefore ecological characteristics will be more similar within hydrological classes than between classes (Arthington et al. 2006). Evidence for the influence of flow and various facets of the flow regime (timing, magnitude, duration, rate of change, predictability and variability) on riparian vegetation patterns and processes of riparian systems are prevalent within international literature.

Flow has been shown to influence many key processes that occur during a plant life cycle from seed dispersal (Merritt and Wohl 2002, Chambert and James 2008), to germination and seedling growth (Scott et al. 1997) and adult plant growth and productivity (Burke et al. 1999, Anderson and Mitsch 2008). The timing of key life history events such as flowering, seed release and germination of some riparian species may coincide with specific hydrological events (Blom et al. 1990, Pettit and Froend 2001a). Complex relationships with flow may also include the mediation by flow of activities of a third party such as predators, both granivores and herbivores (Andersen and Cooper 2000, Elderd and Doak 2006), and effects on competitive interactions between plant species themselves (Busch and Smith 1995).

Although flow is an important determinant of the health of rivers, it cannot be considered in isolation. Riparian vegetation patterns are likely to reflect a complex mix of underlying local and landscape drivers and their interactions. Published analyses of riparian vegetation distribution patterns highlight the importance of broad-scale predictors such as climate, geology and soils in providing the overriding controls on the distribution of riparian vegetation at a broader landscape scale (e.g. Tabacchi et al. 1996*, Dixon et al. 2002, Sarr and Hibbs 2007).*

At the local scale riparian vegetation distribution patterns are typically zonated laterally along transverse gradients with distance and height from the stream edge. These lateral vegetation distribution patterns reflecti the relative tolerances of species to physical disturbances such as shear stresses associated with stream hydraulic conditions and chemical stresses (anoxia and chemical toxicities) associated with water logged soils adjacent to streams, and the ability of species to acquire or intercept resources such as moisture (Lite and Stromberg 2005*), light (Hall and Harcombe 1998, Battaglia and Sharitz 2006) and nutrients (Kotowski et al. 2006) at different positions along the lateral gradient.*

This chapter presents a test of the ELOHA methodology applied to the riparian vegetation of the SEQ region. In the first section of this chapter we present a number of hypotheses generated from the literature review 'Riparian vegetation–Flow Relationships and Responses to Flow Regime Alteration: a Review of Evidence from SEQ Streams' (Appendix 1). In subsequent data analysis and results sections these hypothesis are addressed. In the final section of this chapter we discuss the implications of the results, both in terms of the ELOHA trial and the management of riparian vegetation of the SEQ (SEQ) region.

6.1.1 Hypotheses and objectives

This report presents an application of the ELOHA framework to riparian vegetation in SEQ. Specifically the aims of this report are to:

1. identify how existing flow regime alterations in the study area have impacted on habitat structure/heterogeneity, and the structure, dynamics and productivity of riparian vegetation assemblages
2. identify thresholds (if present) or relationships of habitat and ecological response to flow regime alteration
3. identify a limiting suite of flow variables that together governs the condition or 'health' of each river system (or river zone, or set of rivers in a bioregion) and threshold levels of ecological response to flow regime alteration for the whole suite of flow variables
4. assess the relative influence of flow regime alteration versus other pressures on habitat condition and ecological condition.

To assess the utility of the ELOHA framework for riparian vegetation a number of hypotheses are tested. The central objective of the ELOHA framework is to quantify flow alteration – ecological response relationships for different types of river system classified according to their natural hydrological characteristics (magnitude, timing, frequency, duration and variability).

A review of riparian vegetation for SEQ (Appendix 1), however, revealed a lack of knowledge of explicit links between riparian vegetation and river flows for the region. This necessitates that the first step in this study was the exploration of linkages between the structure and composition of riparian assemblages in the SEQ (SEQ) region and stream flows (Hypothesis 1).

Hypothesis 1: *The structure and composition of riparian assemblages in the SEQ region will be influenced by stream flow.*

Stream flows are a major control on the distribution, abundance and diversity of plants on stream and river banks (Merritt et al. 2010). Here it is suggested that riparian plant distributions, abundances, diversity and variability of streams in SEQ will be largely governed by stream flow regimes.

Whilst there is significant evidence to suggest links between stream flows and riparian vegetation for other regions of Australia and internationally, this link has not been made for riparian vegetation of SEQ. Under this hypothesis the relative contributions of flow as a physical disturbance and as a resource provider in structuring riparian vegetation will be investigated. We propose a number of sub-hypotheses:

- flood and high flow disturbance are a major control on the composition and structure of riparian vegetation
- *baseflow and low flows are a major control on the composition and structure of riparian vegetation*
- variability in stream flows will drive variability in riparian vegetation.

Hypothesis 2: *The structure and composition of riparian assemblages in the SEQ region will be influenced by interactions between flow variables, and other natural factors and anthropogenic disturbances.*

This hypothesis recognizes that riparian vegetation is likely to be influenced by interactions between flow variables, and other natural factors and anthropogenic disturbances. It is predicted that assemblages at a site scale (overall abundances, species richness, diversity and community composition) will be strongly controlled by landscape variables and that the influence of specific flow variables on local riparian plant distributions will be conditional on regional settings.

Under this hypothesis focus will be on landscape variables that are likely to influence the importance of the physical effects of flood disturbances (i.e. topography, stream gradients) and moisture availability (climate) in driving local scale riparian vegetation patterns. It is also suggested that the influence of flow will be greatest for NS vegetation. As distance and elevation increase away from the stream edge, other disturbance and environmental influences upon riparian vegetation patterns become increasingly important.

Hypothesis 3: *Riparian assemblage structure in streams of SEQ will differ across the Reference and Historic hydrological flow classes*.

This hypothesis addresses the basic premise of the ELOHA framework that different hydrological classes will have different riparian assemblages because stream flows are the major control on the distribution, abundance and diversity of plants on stream and river banks. Tests of this hypothesis will be based on the Historic (i.e. the classification based on stream gauge data) and Reference flow regime classification (i.e. the classification based on the modelled natural flow regime) as riparian vegetation community composition and structure are expected to respond to long-term flow regime characteristics.

Following on from this, it is suggested that if flow is a major driver of riparian vegetation patterns, regulated/supplemented sites should be DISSIMILAR to unregulated/unsupplemented sites for a given RFC. Conversely, for a given HFC regulated/supplemented sites should be SIMILAR to unregulated/unsupplemented sites if flow is important.

Furthermore, if the influence of flow is greatest for near-stream vegetation as suggested under hypothesis 2, it is proposed that the near-stream tree and shrub assemblages should be better predictors of flow class relative to bankfull vegetation.

Hypothesis 4: *Changes in stream flow regimes will alter the distribution, abundance and diversity of plants on stream and river banks.*

This hypothesis focuses on responses of riparian vegetation to flow alteration. A review of the evidence for the impacts of flow regime alteration on riparian vegetation suggested a number of potential effects. The manner in which vegetation is likely to respond depends on the nature of the flow regime alteration. In this report we focus on the following predictions:

- changes to near-stream vegetation density where flow regimes are altered
- reduction in the regeneration of native species in the bank full channel where high flows and flood disturbance are reduced
- reduction in the proportion of species characteristic of early successional stages where flood disturbance is reduced
- reductions in species diversity where flow variability is reduced
- increased proportion of exotic species with flow regime alteration.

The ELOHA framework assumes that increasing degree of flow regime change (from Reference condition) is associated with increasing degree of ecological change (Poff et al. 2010). The capacity of this project to test this hypothesis is limited by the degree of flow alteration across the study area. Flow regimes downstream of dams in SEQ have undergone varying degrees of change from baseline (Reference) condition (Chapter 3), hence there is scope to examine changes in vegetation assemblages over a relatively subtle gradient of flow regime change.

6.2 Methods

6.2.1 Study area, flow classification and site selection

The study was conducted in the coastal river catchments of SEQ. There is considerable natural environmental variation across the study region with a complex geology (Bridges et al. 1990; Ellis 1968; Murphy et al. 1976; Whitaker and Green 1980) and associated soils. Distinct topographic regions are identifiable within the region with coastal plains, river floodplains and estuaries in the east, and foothills and mountains with plateaux over 300 m above sea level to the west, north and south of the study region.

The climate is subtropical and dominated by summer rainfall with warm summers and mild winters, but sits adjacent to the temperate/subtropical transitional zone. The area also exhibits a fairly strong rainfall gradient with a decrease in rainfall in a westerly (inland) direction across the catchments (Bridges 1990).

Selection of sites for riparian vegetation surveys was based on classification of modelled natural and historic (actual) flow regimes in SEQ. Both classifications were derived using model-based clustering (Fraley and Raftery 2002) and 35 flow metrics (Chapter 3). The natural flow regime for individual sites was modelled using an IQQM.

The natural classification is hereafter termed the Reference classification and the flow classes derived are termed the 'RFCs'. The Historic flow regime is the flow regime recorded by a stream gauge. Thus the Historic flow regime includes flow regime changes through time. Flow metrics calculated from stream gauge data are termed 'Historic flow metrics' and the classes derived from these metrics are termed the ''HFCs'. Six RFCs and five HFCs were identified.

Sites for riparian vegetation surveys were selected from RFCs and HFCs (Table 6.1). Final site selection was based on proximity to a stream flow gauge, distance between sites within the same reach, the location of major tributaries and current and historical land use (in particular grazing and vegetation clearance).

Wherever possible the sites selected were not currently grazed, had not been cleared in the last 20–30 years and were not subject to regular burning. These criteria were stipulated in order to reduce land use influences on flow–ecology relationships, particularly influences on riparian vegetation which is likely to be strongly influenced by such factors.

Table 6.1: Details of sites surveyed for riparian vegetation

Site number and name	RFC	HFC
1. Stanley River at Cove Road	5	5
2. Burnett Creek downstream of gauge 145018a	No class	3
3. Burnett Creek upstream of gauge 145018a	No class	3
4. Nerang River at Grand Manor Golf Course	5	3
5. Coomera River at Coomera Scouts Hall	1	4
6. Nerang River at Weber Court	5	3
7. Teviot Brook near Brennan Road	No class	3
8. Teviot Brook at Croftby	No class	3
9. Amamoor Creek at Harrys Creek Road	1	4
10. Yabba Creek at Stirling Crossing	2	2
11. Yabba Creek at No. 8 Crossing	2	2
12. Obi Obi Creek downstream of number 2 crossing	1	3
13. Obi Obi Creek upstream of number 2 crossing	1	3
14. Mary River downstream of Walker Road Bridge	1	3
15. Six Mile Creek at Old Noosa Road	1	4
16. Six Mile Creek at Grahams Road	1	4
17. Glastonbury Creek at Greendale Road Crossing	2	3
18. Eudlo Creek at gauge site	5	5
19. Eudlo Creek upstream of Bruce Highway	5	5
20. Reynolds Creek at Yarramalong camp ground	2	1
21. Reynolds Creek at downstream of Purdons Bridge	2	1
22. Amamoor Creek at Zachariah Lane	1	4
23. Glastonbury Creek at 2 km from Mary River confluence	2	3
24. Mary River at Moy Pocket (north of quarry)	1	3
25. Coomera River at Tucker Lane	1	4
26. Stanley River at gauge 143303a	5	5
27. Burnett Creek at 2 km downstream of Maroon Dam	2	1
28. Burnett Creek at Splityard Creek Road	2	1
29. Currumbin Creek at Currumbin Valley Primary School	5	5
30. Currumbin Creek at Fordyce Court	5	5
31. Wide Bay Creek downstream of gauge 138002c	4	2
32. Wide Bay Creek upstream of gauge 138002c	4	2
33. Munna Creek at gauge 138004b	4	2
34. Munna Creek downstream of gauge 138004b	4	2
35. North Maroochy River at Eumundi	5	5
36. North Maroochy River at North Arm–Yandina Creek Road	5	5
37. Mary River at Bauple–Woolooga Road	3	3
38. Mary River at Orphants Road	3	3
39. Tinana Creek at gauge 138903A	2	3
39. Tinana Creek up stream of gauge 138903A	2	3
41. Logan River at Running Creek Road	2	1
42. Logan River at upstream Tilleys Bridge	2	1
43. Teviot Brook below Wyaralong Dam	2	2
44. Teviot Brook at Conlons	2	2

Forty-four study sites (Table 6.1) were selected for sampling of riparian vegetation assemblages along 22 river sections that reflected the major flow regime gradients in SEQ (Chapter 3). We defined our field sites as a stream length of 100 m as this length allowed multiple transects to be conducted within a site and usually incorporated multiple in-stream habitats (riffles, runs and pools) whilst minimising variation due to changes in stream morphology, geology and adjacent land use.

For the first phase of fieldwork (August to October 2008) 28 sites were sampled. Surveys of the remaining 16 sites were completed in 2009 and 2010. Time constraints precluded completion of the entire survey within a single field season and splitting the sampling over two years rather than sampling across seasons within a year was deemed a more appropriate strategy for the riparian vegetation to avoid confounding potential seasonal effects.

6.2.2 Environmental parameters

Flow and other environmental variables for inclusion in these analyses were selected on the basis of their perceived relevance to riparian vegetation.

Flow regime metrics

We selected metrics that represent gradients of water availability and fluvial disturbance to the riparian habitat. A number of studies have illustrated a link between the depth to groundwater and riparian vegetation (reviewed in Merritt et al. 2010).

For this study we assumed that surface flows would be analogous with the local riparian water table and used hydrological metrics describing average flow and low flow conditions to provide proxies for riparian water table dynamics (Table 6.2). Variables describing mean wet season (November – April) and dry season (May – October) flows and their variation were also calculated.

Flood characteristics (frequency, timing and/or duration) are often defined with respect to individual stream landforms. Whilst this makes sense for locations where fluvial landforms are clearly demarked and are analogous in terms of their fluvial and geomorphic conditions, across broader geographic extents and across streams such characteristics may be more variable.

Furthermore, such landforms are often assumed to be invariant with respect to their hydrological (and edaphic) characteristics, an assumption that clearly doesn't hold true for the steeply sloping stream banks characteristic of many of SEQ's streams.

To assess impacts of flood disturbance we therefore analysed vegetation patterns using flooding and high flow statistics that described the frequency, duration and intensity of bankfull and high spell flow conditions rather than individual landforms.

Table 6.2: Flow regime metrics

Category	Parameter	Acronym
Measures of average and low flow conditions	Median daily flow	MEDDaily
	Median annual flow	MEDAnnual
	Mean wet season flow (Nov to Apr)	MDFWet
	Mean dry season flow (May to Oct)	MDFDry
	Mean dailybaseflow	MDBF
	Baseflow index	BFI
	Low spell discharge (75th percentile)	LSDis
	Mean duration of low flow spells (75th percentile)	LSDur
	Number of low flow spells (75th percentile)	LSNum
Measures of flood disturbance	High spell discharge (25th percentile)	HSDis
	Mean duration of high spells (25th percentile)	HSDur
	Mean number of high spells (25th percentile)	HSNum
	Bank full discharge	BFDis
	Mean duration of bankfull flow	BFDur
	Mean number of bankfull flow events	BFNum
	Shear stress at bankfull flow	BFShear
Measures of flow variability	CV of mean daily flow	CVDaily
	CV of annual flow	CVAnnual
	CV of wet season flows (Nov to Apr)	CVWet
	CV of dry season flows (May to Oct)	CVDry
Flow classes	Reference flow class	RFC
	Historic flow class	HFC

Bankfull (BF) discharge estimates were calculated from cross sectional surveys and stream water slope using a staff and dumpy level, and stage height and discharge measures collected at the time of surveys. Cross sections were located in riffles (Harrelson et al. 1994) and were marked with a 10 cm bolt and quick dry cement at a point above bankfull discharge (Harrelson 1994).

A one-dimensional hydraulic model (WINXSPRO) was used to calculate bankfull discharge and shear stress at bankfull flows for each cross section. These values were validated where possible against stage-height relationships for gauges where gauges were in close proximity to the survey site (e.g. sites 18, 22 and 33). Literature estimates suggest bankfull discharge is likely to be equivalent to a 1.5 flow event (Dury et al. 1963, Leopold et al. 1964).

For rivers and streams in SEQ however, bank full discharge is more akin to a 1 in 20 year flood (De Rose et al. 2002) whilst other studies have assumed a bankfull discharge reoccurrence interval of one in ten years (e.g. Fentie et al. 2006). The selection of appropriate roughness coefficients (Mannings's n) for each cross section is likely to be a significant source of error in the determination of bankfull discharge (Harman et al. 2008).

Whilst ideally estimates for Manning's n at each cross section should be determined through calibration based on observed water height at a known discharge, calibration information was not available during high or bank full flows. Estimates of Manning's n made at low flow are not likely to be applicable to high flow conditions as Manning's n varies considerably with factors such as water depth, channel cross sectional area, hydraulic radius and vegetation density. Following Harman et al. (2008) we used the Dingman and Sharma (1997) empirical equation to estimate n:

$$n = 0.217A^{-0.173}R^{0.267}S^{0.156}$$

Shear stress (the frictional force exerted per unit area) at bankfull discharge was determined using the following equation:

$$\pi = pgRS$$

where p is the density of water (1000 kgm^{-3}), g is the acceleration due to gravity (9.8ms^{-2}), R is the hydraulic radius and S is the slope of the energy line, approximated by the water slope.

The temporal scale over which to assess changes in flow metrics requires particular consideration. The relevance of a particular temporal scale is dependent upon the vegetation attribute being measured. Physiological and biochemical characteristics, for example, are more likely to reflect short-term changes in flow regimes, whereas measures of population structure and species composition may reflect changes in flow regimes over longer periods (Merritt et al. 2010).

The metrics presented in this study are at the community and population level and hence measures of longer temporal scales are more relevant to this study. To investigate the influence of flow history we therefore used metrics calculated over a 20 year period immediately prior to sampling (with the exceptions of sites on Nerang, Reynolds and Burnett Creek downstream of Maroon Dam where only shorter flow records were available).

Topography, climate, substrate

All sampled sites were located within a Geographic Information System (GIS) (ARCGIS 9.2 Environmental Systems Research Institute ESRI). Physical variables based on upstream catchment and local valley characteristics were acquired or derived from a number of readily available digital datasets. In particular, we sourced reach scale variables from a recently derived national dataset (Stein et al. 2009).

Reach extents were as defined by Stein et al. (2009) where reaches are stream sections (less than 10 km in length) between tributary confluences. Other datasets included the digitised geology maps at a scale of 100 000 and the SEQ Region Geoscience dataset (version 2) (DNRM 2002b). This latter data package was compiled from regional geological mapping at a nominal scale of 100 000 conducted by the Geological Survey of Queensland with additional data from university theses and exploration company mapping.

Reach mean annual temperature, reach hottest month mean temperature and reach coldest month mean temperature variables were acquired from Stein et al. (2009) for individual reaches on which sites were situated. Mean annual rainfall was supplied by the Bureau for Meteorology (2009). Rainfall values are based on a rainfall grid generated using ANU 3-D Spline. The nearest grid point to each individual site is used in the analysis.

Catchment boundaries for each site were delineated in a GIS using detailed stream networks based on 25 000 and 100 000 scale maps and a 30 m DEM (digital elevation model) (NASA DTED2 2007). Catchment area was computed using geometry functions of the GIS software.

Table 6.3: Environmental parameters

See text for explanation of variables.

Category	Parameter	Unit	Acronym
Local topography	Latitude	Degrees	DECLAT
	Longitude	Degrees	DECLONG
	Site elevation	M a.s.l	ELEV
	Bank aspect – north	Degrees	B_ASP_N
	Bank aspect – east	Degrees	B_ASP_E
	Stream bank gradient	$m.m^{-1}$	B_SLOPE
	Ratio of bankfull width:bankfull depth	No unit	BFWIDTH_DEP
Catchment topography	Catchment area	ha	CAT_AREA
	Distance to source	km	DISTS
	Catchment elongation	No units	CAT_ELONG
	Catchment relief	No units	CAT_RELI
	Valley confinement	%	V_CONF
Climate	Annual mean rainfall	mm	A_RAIN
	Annual mean temperature	ºC	A_TEMP
	Hottest month mean temperature	ºC	HMA_TEMP
	Coldest month mean temperature	ºC	CMA_TEMP
	Valley slope	$m.m^{-1}$	V_SLOPE
Substrate	Igneous – Mafic	%	MAFIC
	Igneous – Felsic	%	FELSIC
	Mixed sedimentary and igneous (Felsic)	%	MIXEDF
	Mixed sedimentary and igneous (Mafic)	%	MIXEDM
	Sedimentary (siliclastic and undifferentiated)	%	SED-SILIC
	Unconsolidated rocks (alluvium, colluvium etc)	%	UNC_CATCH
	Unconsolidated material for reach	%	UNC_REACH
	Sand	%	SAND
	Clays and silts	%	CLAY
Land use	Production from relatively natural environments	%	PNE
	Production from dryland agriculture and plantations	%	PDA
	Production from irrigated agriculture and plantations	%	PIA
	Conservation and natural environments	%	CAN
	Intensive uses	%	IU

Drainage basin shape is represented here by the elongation ratio (Re; the diameter of a circle with the same area as that of the basin divided by the length of the basin) which is considered to have a reasonable correlation with stream hydrology (Morisawa 1958).

Reach morphology variables, stream gradient (elevation difference for reach divided by its length) and valley confinement (percentage of stream grid cells and their immediate neighbours that are not defined as valley bottoms) were acquired from Stein et al. (2009) and are based on the Multi-resolution Valley Bottom Flatness Index of Gallant and Dowling (2003).

Site topography variables, distance to source and elevation were determined for each site using GIS with detailed stream networks based on 1:25 000 and 1:100 000 scale drainage maps and the 30 m DEM. Aspect was measured with a compass in the field or from the 30 m DEM.

Substrate geological characteristics were derived for the field sites from the SEQ Region Geoscience dataset (Queensland DNRM 2002) and digital 1:100 000 scale geology maps for the region. Geological groupings were based on broad composition. Percentages of silts and sands were determined from surface soil samples collected at the time of riparian surveys using a soil hydrometer method.

Surface soil samples were collected from each corner and the centre of a one meter quadrat and pooled into a single sample for analyses. A minimum of three quadrats were sampled per site (at the stream edge, midway along the transect and at the bankfull elevation). Additional soil samples were taken where cross sections intersected other distinct landforms (e.g. bars, terraces). Results for proportions of clays and sands represent samples means.

Land use

Landscape-scale land use and disturbance data for field site catchments were obtained from the QLUMP dataset (Witte et al. 2006) generated from baseline surveys conducted in 1999. Draft updates available from 2006 surveys for the Maroochy and Logan–Albert (Stanley and Bremer river catchment information were not available at the time of analysis for this report) were also incorporated.

We chose to use the primary land use classes based on the Australian Land use and Management Classification version 6 (BRS 2002) as these represent broad land use categories differentiating conservation and relatively natural land uses from intensive land uses.

As land use close to streams may have a disproportionate affect on stream health and condition relative to land uses distal to the stream (Tran et al. 2010), we calculated a distance weighted metric for each primary land use class. In a recent study of relationships between land use metrics and various ecological indicators conducted in SEQ, Petersen et al. (2010) found that distance weighted metrics generally outperformed the lump metrics but no single metric was best overall for the indicators studied. We therefore chose an inverse–distance weighting $(d+1)^{-1}$ metric following Petersen et al. (2010), which is relatively straightforward to calculate.

6.2.3 Riparian vegetation survey methods

Riparian vegetation surveys followed a relatively standard survey methodology. Within each 100 meter field site we randomly located three transects along cross sections running perpendicular to the river. Additional transects were conducted at sites 27, 31 and 44 where extremely low species densities were encountered and less than 100 individuals were counted in the initial 3 transects conducted.

Transects extended from the water's edge to bankfull or 50 m distance from the water's edge where no perceivable changes in vegetation beyond 50 m were observed. A standard sampling area was not utilised because of the variation in vegetation densities, channel forms and adjacent land uses across the study region.

Site sampling areas varied between 260 and 1013 m^2 with the sampling area for the majority of sites being greater than 400 m^2 as recommended by Walker and Hopkins (1984). All transects were conducted on the same side of the river to ensure that any land use impacts were similar amongst transects within each site.

Along each transect we located all trees, shrubs, ferns, reeds, rushes and sedges within a 5 meter wide band. We restricted our surveys to recruits ≥ 50 cm in height. For each element we recorded the species, its location along the transect (as a distance from the water's edge), canopy height, trunk diameter at breast height (DBH), the presence and composition of vines (both exotic and native) and a health measure ranging from 0 (dead) to 4 (healthy with >75% canopy cover and little or no evidence of disease or insect damage).

Reed, rush (including mat-rush *Lomandra* spp.) and sedge densities were estimated based on clump number and size to provide an estimate of their cover at each site. Specimens for all species were collated and where species appeared to differ significantly in morphology additional specimens were taken. Species identifications were verified by the Queensland Herbarium.

6.2.4 Riparian vegetation metrics

Riparian datasets were generated on a site basis by combining the results for all the transects from each site. For analysis of assemblage data, trees and shrub abundance data were standardised by unit area (ha). Tree and shrub assemblage data was analysed together and separately for initial analyses. The total tree and shrub assemblage within stream bankfull is termed the 'Bankfull vegetation' in subsequent analyses.

Vegetation of near-stream communities may be influenced to a greater degree by hydrology compared with vegetation further away from the stream edge, hence we also subset the data into near-stream vegetation only (termed 'Near-stream vegetation' in subsequent analyses). Near-stream was defined as vegetation within 5m of the waterline as this zone was found to contain chiefly only those tree and shrub species generally considered riparian.

In addition to the assemblage data, various riparian vegetation metrics were generated for each site based on the bankfull and near-stream vegetation datasets (Table 6.4). Functional groups based on growth forms and successional status was generated from a comprehensive literature survey. Growth forms were determined as Shrubs (S) or Tree (T) based on available literature information on size at maturity and form.

Following Kariuki and Kooyman (2005) over storey trees with a diameter at breast height ≤10 cm were all categorised as regenerating. Successional stages followed the terminology of Kanowski et al. (2010) as Early (E), Intermediate (M) or late (L) and combinations of these stages where the species occurred in more than one successional stage (i.e. EM, ML or EML). Early successional stage species are usually light-demanding, intermediate stage species gap-demanding and later successional species shade-tolerant.

Successional stages usually followed those of the database recently compiled by Kanowski (2010), and for additional species absent from this database an extensive search of the available literature was made. In addition to the tree and shrub metrics, the total density of reeds, rushes (including mat-rushes, Lomandra spp.) and sedges was calculated per unit area (ha) for each site.

Table 6.4: Riparian vegetation metrics

Details for classification of growth form, successional stage and regeneration are provided in the text.

Type	Metric acronym	Definition
Total	RICH	Tree and shrub species richness
	D_SPECIES	Density of trees and shrubs (number of species per m^2)
	D_ALL	Density of all trees and shrubs per ha
	D_REGEN	Regeneration density per ha
	BA_ALL	Total basal area per ha
Origin	D_EXOTIC	Exotic density per ha
	D_NATIVE	Native density per ha
	D_REGEN_ NATIVE	Native regeneration density per ha
	D_REGEN_ EXOTIC	Exotic regeneration density per ha
	%_EXOTIC	% exotic taxa
	%_NATIVE	% native taxa
	BA_EXOTIC	Exotic basal area per ha
Growth form	D_SHRUB	Shrub density per ha
	D_TREE	Tree density per ha
	D_LOMAND	Rush, reed and sedge densities per ha
	BA_SHRUB	Shrub basal area per ha
	BA_TREE	Tree basal area per ha
Succession	D_EARLY	Early (E, EM, EML) density per ha
	D_INTER	Intermediate (EM, M, ML, EML) density per ha
	D_LATE	Late (ML, L, EML) density per ha
	BA_EARLY	Early (E, EM, EML) BA per ha
	BA_INTER	Intermediate (EM, M, ML, EML) BA per ha
	BA_LATE	Late (ML, L, EML) BA per ha

6.3 Data analysis

We used a number of different univariate and multivariate approaches to analyse the riparian vegetation component of this project. These are summarised in Table 6.5.

Data analysis for Hypotheses 1 and 2

Multivariate

Hypotheses 1 and 2 were analysed using a combination of unconstrained (nMDS) and constrained (partial Canonical Correspondence Analysis (CCA)) ordination techniques. For these analyses, sites with flow regimes strongly impacted by flow regulation were removed prior to the analysis to avoid confounding relationships with sites that may still be undergoing change in vegetation structure and composition in response to flow regime alteration.

Sites were removed from the analysis that had returned a Gower metric of 0.10 or greater (Figure 3.10). Above this threshold class membership probability for gauges in the RFC were erroneously allocated to the wrong class (Table 3.13), that is flow regime was altered to the degree that the random forests model could not allocate the gauge to the correct RFC.

Riparian vegetation patterns across the study area were first investigated using nMDS as implemented by the *metaMDS* function in the vegan package for R (Oksanen et al. 2010; R Development Core Team 2010). The Bray–Curtis dissimilarity measure was used to calculate the distance matrix (Faith et al. 1987). Fifty random starts were used to find the configuration that minimised stress (i.e. goodness of fit between observed dissimilarities and ordination distances).

The *metaMDS* function rotates the best solution (rotation to principal components) so that maximum variation is displayed on the first ordination axis (Oksanen et al. 2010). The envfit function was used to fit environmental vectors to the ordination space. This function finds the maximum correlation between intrinsic variables and the ordination space. Significance was assessed using a randomisation procedure and 999 permutations.

Individual ordinations were undertaken for tree and shrub assemblage data separately as well as a combined tree and shrub dataset. Initial ordinations were undertaken using raw, log-transformed ($\log^{10}x+1$ for $x>0$), and square root-transformed species density data. Log-transformed data produced the lowest stress value and was subsequently used for all ordinations. Rare species (those occurring in two or less sites) were removed prior to analyses.

Partial CCA was used to determine the variation explained in species assemblage data by flow metrics (Table 6.2), other natural environmental variation (topography, climate and substrate) and land use (Table 6.3). CCA is a constrained ordination technique, where the variation explained is constrained by the environmental variables used. Ordination axes are linear combinations of the explanatory variables (Borcard et al. 1992).

In partial CCA, the effects of a co-variable matrix are first removed and the residual variation explained by a second environmental matrix is then determined (Anderson and Gribble 1998). This method is useful when datasets share an underlying structure and is typically used to determine variance explained by a set of environmental (predictor) variables independent of other confounding factors or underlying structure (Borcard et al. 1992).

Underlying structure shared by the biotic and environmental datasets can result in an overestimation of the interactions between biotic and environmental variables (Borcard et al. 1992). Through a series of partial CCAs the shared structure can be partialled out, leaving ‘pure’ variance (Anderson and Gribble 1998) explained by the environmental data matrix.

Table 6.5: Summary of analytical approaches used to analyse riparian vegetation

Data subsets are defined in the text.

Hypothesis	Data type	Data subsets	Analytical approach
Hypothesis 1: *The structure and composition of riparian assemblages in the SEQ region will be influenced by stream flow.*	Assemblage	Bankfull, trees only, shrubs only, near-stream	non-metric multidimensional scaling (nMDS), partial canonical correspondence analysis (partial CCA)
	Metrics	Bankfull, near-stream	non-metric multidimensional scaling (nMDS), Regression random forests and partial dependence plots, generalized least-squares (GLS) regression
Hypothesis 2: *The structure and composition of riparian assemblages in the SEQ region will be influenced by interactions between flow variables, and other natural factors and anthropogenic disturbances.*	Assemblage	Bankfull, near-stream	partial canonical correspondence analysis (partial CCA)
	Metrics	Bankfull, near-stream	Regression random forests and partial dependence plots
Hypothesis 3: *Riparian assemblage structure in streams of SEQ will differ across the IQQM and gauge hydrological flow classes.*	Assemblage	Bankfull, near-stream	Analysis of Similarity (ANOSIM), permutational Multivariate Analysis of Variance, Classification random forests
	Metrics	Bankfull, near-stream	Kruskal–Wallis tests
Hypothesis 4: *Changes in stream flow regimes will alter the distribution, abundance and diversity of plants on stream and river banks.*	Assemblage	Regulated vs unregulated across all sites (bankfull and near-stream data). Regulated vs unregulated within Reference flow groups (1, 2 and 5) and historical flow groups (1, 2, 3 and 4) for bankfull and near-stream assemblage data.	Analysis of Similarity (ANOSIM)
	Metrics	Kruskal–Wallis tests performed on all metrics, PLS performed on only those metrics for which reasonable predictive models could be constructed.	Kruskal–Wallis tests, Partial least-squares (PLS) regression

Three environmental datasets were used to undertake partial CCA. One matrix contained flow metrics only (HYDRO), since the goal of this analysis was to determine the variation in species composition data explained by hydrologic data independently of other environmental parameters. The other matrices included natural environmental variables apart from flow (ENVIRO) and land use parameters (LAND USE). Correlation matrices of each dataset were first examined and highly correlated variables removed.

Partial CCA was conducted on the tree and shrub assemblage matrix for bankfull and near-stream vegetation datasets separately, using the cca function in the vegan package for R (Oksanen et al. 2010). Density data were transformed ($\log_{10}(x)+1$ for $x>0$) prior to analysis (Jongman et al. 1997). Preliminary analyses indicated that log-transformation improved the variation explained, compared with untransformed data. Rare species (those occurring in two or less sites) were removed prior to analysis. Environmental variables were standardised to unit variance.

First we used CCA with forward selection to determine which variables from each data subset were significant in structuring riparian vegetation assemblages. Following the initial CCA analyses, variance inflation factors (VIFs) were examined to determine whether variables exhibited high collinearity. Variables with high VIFs (>10, Ahmadi-Nedushan et al. 2006) were excluded from further analysis and the CCA repeated with these variables excluded. The significance of each variable was tested with 199 Monte Carlo permutations. Only variables with $P < 0.05$ were used in the subsequent partial CCA analyses.

Following selection of significant variables, partial CCAs were carried out using the procedure described by Borcard et al. (1992) and Anderson and Gribble (1998). The steps in the variance partitioning process are summarised in Table 6.6. Completion of the twelve sets of analyses allowed the total variance explained by each of the three explanatory datasets to be determined. Partial CCA was undertaken using the vegan package for R (Oksanen et al. 2010).

Table 6.6: Summary of Canonical Correspondence Analyses performed (after Anderson and Gribble 1998)

Total explained variation is equal to the variance explained by analysis (1+7+12) or analysis (2+4+12) or analysis (3+5+9). Dataset codes explained in the text.

Analysis	Explanatory dataset	Covariable dataset(s)
1	LAND USE	None
2	HYDRO	None
3	ENVIRO	None
4	LAND USE	HYDRO
5	LAND USE	ENVIRO
6	LAND USE	HYDRO + ENVIRO
7	HYDRO	LAND USE
8	HYDRO	ENVIRO
9	HYDRO	LAND USE + ENVIRO
10	ENVIRO	LAND USE
11	ENVIRO	HYDRO
12	ENVIRO	LAND USE + HYDRO

Univariate

The importance of hydrologic versus other environmental factors in describing patterns in vegetation metrics (Table 6.4) was assessed using regression random forests. This non-parametric method is suited to the analysis of ecological data where complex relationships exist both amongst predictor variables (e.g. colinearity) and between predictor and the response variables (De'arth and Fabricius 2000). A random forest is a decision tree method similar to classification and regression trees (CART), except that a 'forest' of trees is generated compared with a single tree in CART methods. Random forests models cannot be overfit (contain too many predictor variables) if the number of trees created is large (Breiman 2001). The algorithm is as follows:

1. Select 1000 bootstrap subsample of cases representing approximately two thirds of the original dataset.
2. For each bootstrap sample, grow an unpruned regression tree, and at each node select a subset of random predictor variables and choose the best split from amongst those variables.
3. For each iteration predict the samples not included in the bootstrap sample (the OOB sample) and average those response values over all trees.
4. Calculate the importance values for each predictor based on the change in mean squared error (MSE) when that predictor variable is permutated randomly.

Important variables will have a larger effect on the mean squared error relative to irrelevant variables. Functions within the randomForest package for R (Liaw and Weiner 2009) were used to construct random forests. The number of predictors used at each split was determined using the *tuneRF* function. This function constructs random forests and compares the OOB error rate when the number of metrics used at each split is varied. Partial dependence plots were used to examine the effects of changing individual flow predictors whilst holding other predictors at their average.

Relationships between vegetation metrics and selected Historic flow metrics were modelled using generalized least-squares (GLS) regression. GLS regression allows for correlated errors and unequal variances (Bolker 2008; Pinheiro 2011). Each relationship was modelled as a univariate model (i.e. a single flow metric was used as a predictor for each vegetation metric tested).

A GLS regression was fitted for each relationship and if the model was significant ($P<0.05$) the model fit was plotted. Residuals were examined to look for outliers and violations of model concepts, especially heteroscedastic or non-normal errors. The modelled vegetation metrics were based on the random forests regression models. Where hydrology metrics were identified as important variables (the top 5) in metric random forests models, GLS models were fitted using the gls function in the nlme package for R (Pinheiro 2011).

6.3.2 Data analysis for Hypothesis 3

This analysis tested the hypothesis that riparian assemblage structure would vary between HFCs and RFCs. One way multivariate analysis of similarity (ANOSIM) was conducted across RFCs and HFCs using PRIMER (PRIMER 5, Primer-E Ltd. 2001).

To corroborate the results of the ANOSIM we also conducted permutational Multivariate Analysis of Variance (PERMANOVA) using Distance Matrices with the adonis function in the 'Vegan' package to examine tree and shrub assemblages across the flow classes. The Multivariate Analysis of Variance using Distance Matrices method is equivalent to the Permutational MANOVA (PERMANOVA) of M.J. Anderson (Anderson 2001, McArdle and Anderson 2001).

Prior to both the above analyses, species abundance data were $\log10(x)+1$ transformed for $x > 0$ (i.e. zero values were not transformed, see Anderson et al. 2006; Oksanen et al. 2010) and rare species (those occurring in two or less sites) removed. Distances were calculated using the Bray–Curtis distance metric.

As the groups are highly unbalanced it was not possible to analyze pairwise differences between group means for the PERMANOVA. However, we conducted an Analysis of Multivariate Homogeneity of group dispersions (variances) using a test analogous to Levene's test of the equality of variances, followed by pairwise comparisons using Tukey HSD to investigate differences in variance amongst classes.

The indicator values of species for each RFC and HFC were determined as the product of the relative frequency and relative average abundance in the flow classes. Indicator value is maximised (1) when all individuals of a species are found in a single flow class (high fidelity) and when the species occurs in all sites in that class (high constancy). This analysis was conducted using Duferene–Legendre indicator species analysis with the indval function in the 'Labdsv' package of R.

Next, random forests (Breiman 2001) were used to determine how well the Historic and Reference flow classification could be recovered from the composition data (Cutler et al. 2007). Random forests were conducted as described in the previous section but using flow class as a response vector (i.e. classification rather than regression).

For classification, the predicted class of each OOB observation is the class with the highest number of votes. Each tree votes for a class and the class with the most votes is the predicted class for that observation. Variable importance (a measure of misclassification rate when the values for an individual predictor are permuted) was assessed using mean decrease in prediction accuracy (Liaw and Weiner 2009).

Variables in this case are species. Mean decrease in accuracy for individual predictors is assessed using the OOB samples. Values for each predictor variable are randomly permuted for the OOB samples and passed down each tree to get new predictions. The difference in the misclassification rate between the original and modified OOB sample is averaged over all trees and normalised by the standard error (Cutler et al. 2007; Liaw and Weiner 2009). The confusion matrix was used to assess how well vegetation assemblage data could be used to predict HFC and RFC membership of sites.

Differences in riparian vegetation metrics (Table 6.2) across the RFCs and HFCs were examined using Kruskal–Wallis tests. Where significant differences between groups were returned, multiple comparison tests (Tukey's HSD) were conducted (Bonferroni-corrected significance for each test is $\alpha/10 = 0.005$).

6.3.3 Data analysis for Hypothesis 4

Several methods were used to assess the impacts of flow regime alteration on riparian vegetation. Firstly, ANOSIM was used to compare tree and shrub composition between regulated/supplemented and unregulated/unsupplemented sites overall, regardless of flow class. Next, ANOSIMs were used to compare between regulated/supplemented and unregulated/unsupplemented sites in selected RFCs and HFCs (Table 6.7).

It was hypothesised that for a given RFC a regulated/supplemented site should be DISSIMILAR to the unregulated/unsupplemented sites, if flow is a major driver of assemblage structure. Log transformed tree and shrub assemblage data (logx +1 for x>0) were used to calculate Bray–Curtis dissimilarities.

Due to the distribution of regulated/supplemented sites in RFCs this test could only be done for RFCs 1, 2 and 5 (Table 6.7). Following on from the above hypothesis, it was also predicted that if flow was a major driver of tree and shrub assemblages then regulated/supplemented sites should be SIMILAR to unregulated/unsupplemented sites within the HFCs.

Again, due to the distribution of regulated/supplemented sites, this hypothesis could only be tested for HFCs 1, 2, 3 and 4 (Table 6.7).

Table 6.7: Reference sites for Reference flow classes (RFCs) and Historic flow classes (HFCs)

Regulated/supplemented sites	RFC	HFC	Unregulated pre-development reference	Unregulated historic reference
Obi Obi Creek downstream of Baroon Pocket Dam	1	3	Coomera River (5 and 25) Amamoor Creek (9 and 22) Mary River (14 and 24)	Burnett Creek (2 and 3) Teviot Brook (7 and 8) Mary River (14 and 24) Mary River (37 and 38) Glastonbury Creek (17 and 23) Tinana Creek (39 and 40)
Six Mile Creek downstream of Six mile Creek Dam	1	4	Coomera River (5 and 25) Amamoor Creek (9 and 22) Mary River (14 and 24)	Coomera River (5 and 25) Amamoor Creek (9 and 22)
Yabba Creek downstream of Borumba Dam (10 and 11)	2	2	Logan River (41 and 42) Glastonbury Creek (17 and 23) Tinana Creek (39 and 40) Teviot Brook (43 and 44)	Teviot Brook (43 and 44) Wide Bay Creek (31 and 32) Munna Creek (33 and 34)
Burnett Creek downstream of Maroon Dam (27 and 28)	2	1	Logan River (41 and 42) Glastonbury Creek (17 and 23) Tinana Creek (39 and 40) Teviot Brook (43 and 44)	Logan River (41 and 42)
Reynolds Creek downstream of Moogera Dam (20 and 21)	2	1	Logan River (41 and 42) Glastonbury Creek (17 and 23) Tinana Creek (39 and 40) Teviot Brook (43 and 44)	Logan River (41 and 42)
Nerang R downstream of Hinze Dam (4,6)	5	3	Currumbin Creek (29 and 30) Eudlo Creek (sites 18 and 19) North Maroochy (35 and 36)	Burnett Creek (2 and 3) Teviot Brook (7 and 8) Mary River (14 and 24) Mary River (37 and 38) Glastonbury Creek (17 and 23) Tinana Creek (39 and 40)

Finally, PLS projection to latent structures modelling was used to examine relationships between flow regime alteration and assemblage structure, using the methods of Englund et al. (1997a,b) and Zhang et al. (1998). PLS modelling is similar to PCA in that many predictor variables can be summarised into a reduced set of latent components (Eriksson et al. 1995).

An advantage of PLS modelling is that it can handle correlated predictor variables and situations where the number of predictor variables greatly exceeds the number of cases (Eriksson et al. 1995). Individual PLS regression models were developed for selected vegetation metrics (RICH, D_SPECIES, D_NATIVES, D_TREE, D_LOMAND and D_REGEN_NATIVE). Firstly, PLS models were developed for metrics using environmental parameters not directly influenced by flow regulation as predictors.

These models were based on unregulated/unsupplemented sites only. Predictors were selected based on variable importance from the regression random forests models (Section 6.5.2). Environmental data were standardized prior to analysis to have a zero mean and a standard deviation of 1 (Jansson et al. 2000). Models were cross-validated using 'leave one out' cross-validation (Wehrens and Mevik 2009) and the appropriate number of components for individual models was determined from a plot of the root mean squared error of prediction versus the number of components.

The Reference model was then used to predict vegetation metric values for sites influenced by flow regime alteration, where individual PLS models and adequate Reference models could be developed (RICH, D_SPECIES, D_LOMAND, D_NATIVES and D_REGEN_NATIVE). The 'effect' of flow regulation was calculated as [(observed value-predicted value)/predicted value] × 100 (Zhang et al. 1998). The effect of flow regulation was considered to be significant if the mean of the effects did not include zero, corresponding to $P<0.05$ (Zhang et al. 1998). PLS models were fitted using the pls for R (Wehrens and Mevik 2009) and the orthogonal scores algorithm.

Finally, to test the hypothesis that increasing flow regime change causes increased divergence of assemblage structure from Reference condition, residuals from the PLS models (i.e. the effects as described above) were correlated with the Gower metric values obtained for the comparison of Reference and Historic flow regimes for individual stream gauges (Chapter 3). If increasing flow regime change is associated with increasing divergence of biota from Reference condition, then the effect for individual sites should be correlated with the Gower metric.

6.4 Results

6.4.1 Riparian vegetation of south-east Queensland

Over 15 500 individual trees and shrubs were identified and recorded across the 44 survey sites. A total of 191 trees and shrub species and, 43 vine species were indentified (Appendix 6.1).

The most diverse sites on Currumbin Creek, Amamoor Creek, Yabba Creek and Stanley River had 49, 45, 45 and 44 trees and shrub species recorded respectively. The five most abundant native species were *Ficus coronata* (Sandpaper Fig), *Castanospermum australe* (Black Bean), *Cryptocarya triplinervis* (Three Veined Laurel) and *Syzygium floribundum* (Weeping Lilly Pilly).

Exotic taxa comprised 23% of all individuals recorded. The most abundant exotic species were *Celtis sinensis* (Chinese Elm), *Lantana camara* (Lantana), *Leucaena leucocephala* (Leucaena), *Cinnamomum camphora* (Camphor Laurel) and *Ligustrum lucidum* (Broad-leaved Privet).

Densities of trees and shrubs per ha ranged from just under 1000 trees and shrubs per ha (Burnett Creek site 27) to over 21 500 (Teviot Brook). The extremely high tree and shrub density recorded at Teviot Brook was due to a very large number of *Celtis sinensis* recruits, hence this site also had the highest density of tree regeneration (>20 000 per ha).

Proportions of trees belonging to the different successional stages (E, I, L) varied considerably across the sites. Overall, early successional stage species comprised around 24% of all individuals recorded, whilst intermediate and late successional stage species comprised 42% and 33% respectively.

6.4.2 Hypotheses 1 and 2

Ordinations (nMDS) were performed on bankfull and near-stream vegetation and on vegetation data subset into trees and shrubs separately. As ordinations using trees and shrubs were generally poorer with higher stress values, only the combined trees and shrubs analyses are presented here. Ordinations distinguished some broad vegetation types.

Vegetation associated with drier inland sites was typified by a relatively small suite of species including *Grevillia robusta, Casuarina cunninghamiana, Melaleuca viminalis (Callistemon viminalis), Melaleuca bracteata* and the exotic *Celtis sinensis.* Rainforest sites, particularly those of coast creeks and to the north of the study region in the Mary River catchment, were typified by a diverse assemblage of rainforest species including a number of species not generally considered obligate riparian.

An analysis of near-stream vegetation only identified a more limited suite of riparian species associated with rainforest vegetation types (both dry and wet rainforests), including *Syzygium floribundum* and *Ficus coronata*. However, even near-stream communities included some species typical of most rainforest types (e.g. *Guioa semiglauca* and C*leistanthus cunninghamii*) and not restricted to the riparian zone. A large number of predictor variables were significantly correlated with the ordination of tree and shrub assemblage data (Table 6.8 and Figure 6.1). Important flow metrics included a suite of hydrological variables, although notably the hydrology variables correlated with the ordinations differed depending upon whether bankfull vegetation or only near-stream vegetation was analysed.

Hydrology variables characterising high flow and flood conditions tended to correlate more strongly with the ordinations of the bankfull vegetation than near-stream vegetation. However there were some anomalies with this observation as bankfull shear stress (BFShear) and bankfull discharge (BFDis) only correlated significantly with the near-stream vegetation ordination (Figure 6.2). Measures of average flow conditions (median annual flow (MEDAnnual), mean wet season flow (MDFWet) and mean dry season flow (MDFDry) only correlated with the near-stream vegetation ordination.

Climate variables were relatively strongly correlated with both the bankfull and the near-stream vegetation ordinations. However, substrate type was in general poorly correlated, with only the percentage of unconsolidated material in the catchment (UNC_CATCH) strongly correlated with the ordinations. Catchment land use variables – 'production from relatively natural environments' (PNE), 'production from dryland agriculture and plantations' (PDA) and 'intensive uses' (IU) – correlated with both the bankfull and the near stream vegetation ordinations.

Table 6.8: Non-metric MDS ordination of sites based on log(x)+1-transformed tree and shrub assemblage data (densities per ha) (x>0), stress = 0.163, two dimensions

*Environmental variables significantly correlated with the ordination (P=0.05). See Table 6.3 for variable acronyms. Significance codes: *** = <0.001, ** <0.01, * < 0.05.*

Environmental parameter	Bankfull trees and shrubs			Near stream trees and shrubs		
	r2	Pr(>r)		r2	Pr(>r)	
DECLONG	0.204	0.029	*	0.281	0.004	**
ELEV	0.375	0.001	***	0.097	ns	
A_RAIN	0.558	0.001	***	0.516	0.001	***
A_TEMP	0.327	0.002	**	0.376	0.001	***
HMA_TEMP	0.236	0.02	*	0.305	0.004	**
CMA_TEMP	0.646	0.001	***	0.424	0.001	***
CAT_AREA	0.021	ns		0.218	0.01	**
CAT_RELI	0.3477	0.001	***	0.457	0.001	***
V_CONF	0.1966	0.025	*	0.244	0.007	**
V_SLOPE	0.2823	0.011	*	0.204	0.016	*
FELSIC	0.2364	0.018	*	0.032	ns	
SED_SILIC	0.2098	0.025	*	0.100	ns	
UNC_CATCH	0.572	0.001	***	0.472	0.001	***
CLAYS	0.3243	0.004	**	0.205	0.016	*
PNE	0.1945	0.038	*	0.351	0.001	***
PDA	0.3023	0.001	***	0.270	0.004	**
IU	0.3091	0.002	**	0.398	0.002	**
MEDAnnual	0.001	ns		0.189	0.023	*
CVAnnual	0.2041	0.036	*	0.189	0.035	*
MDFWet	0.001	ns		0.199	0.016	*
MDFDry	0.003	ns		0.217	0.010	**
CVWet	0.123	ns		0.267	0.003	**
CVDry	0.5673	0.001	***	0.348	0.002	**
HSDis	0.003	ns		0.196	0.021	*
HSDur	0.3472	0.001	***	0.182	0.032	*
HSNum	0.2746	0.008	**	0.173	0.044	*
LSDur	0.2541	0.01	**	0.098	ns	
BFI	0.2709	0.008	**	0.097	ns	
MDBF	0.001	ns		0.183	0.027	*
BFShear	0.2465	0.011	*	0.320	0.003	**
BFDis	0.094	ns		0.291	0.003	**
BFNum	0.4783	0.001	***	0.218	0.025	*
BFDur	0.3981	0.001	***	0.302	0.002	**

HFCs and RFCs superimposed on the ordinations suggest that vegetation assemblages are relatively indistinct. However, both HFCs and RFCs appeared to be distributed along a climatic gradient with highest rainfall (A_RAIN), highest coldest monthly average temperature (CMA_TEMP) and highest annual mean temperature (A_TEMP) associated with HFCs 4 and 5 and RFCs 1 and 5.

Class 5 for both the Historic and Reference classification was the most distinct. This class was associated with higher rainfall (A_RAIN), higher temperatures (CMA_TEMP) and a higher proportion of intensive land uses within the catchments (IU). This pattern was preserved for both bankfull vegetation and near-stream vegetation.

Ordinations of the riparian metrics found only a very limited suite of variables were significantly correlated with the ordination space. These variables were again associated with climate (CMA_TEMP and A_RAIN for bankfull, and CMA_TEMP and A_TEMP for near-stream). Flow classes superimposed on the ordinations suggested that flow classes were again relatively indistinct (particularly for the RFCs) with respect to the various metrics, although HFCs appeared to be distributed along a climatic gradient.

Figure 6.1: Non-metric MDS ordination of sites based on log(x)+1-transformed ALL tree and shrub assemblage data (densities per ha) (x>0), stress = 0.163, two dimensions.

(a) Position of sites in ordination space. Vectors show taxa significantly correlated with the ordination (P=0.01).

(b Distance to group centroids for sites in each HFC.

(c) Environmental variables significantly correlated with the ordination (P=0.01).

(d) Distance to group centroids for sites in each RFC. For species acronyms see Attachment 6.1.

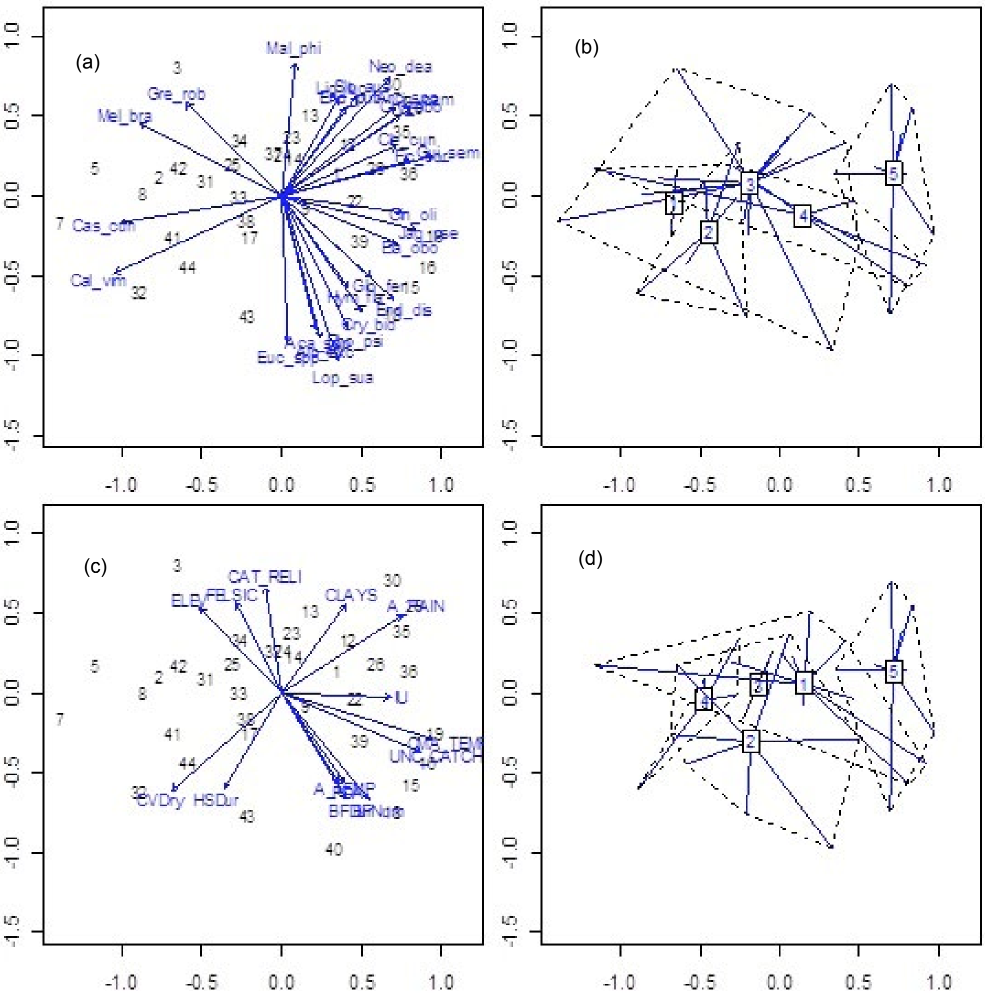


Figure 6.2: Non-metric MDS ordination of sites based on log(x)+1-transformed near-stream tree and shrub assemblage data (densities per ha) (x>0), stress = 0.169, two dimensions.

(a) Position of sites in ordination space. Vectors show taxa significantly correlated with the ordination (P=0.01).

(b) Distance to group centroids for sites in each HFC.

(c) Environmental variables significantly correlated with the ordination (P=0.01).

(d) Distance to group centroids for sites in each RFC. See Tables 6.2 and 6.3 for variable acronyms and Attachment 6.1 for species acronyms.

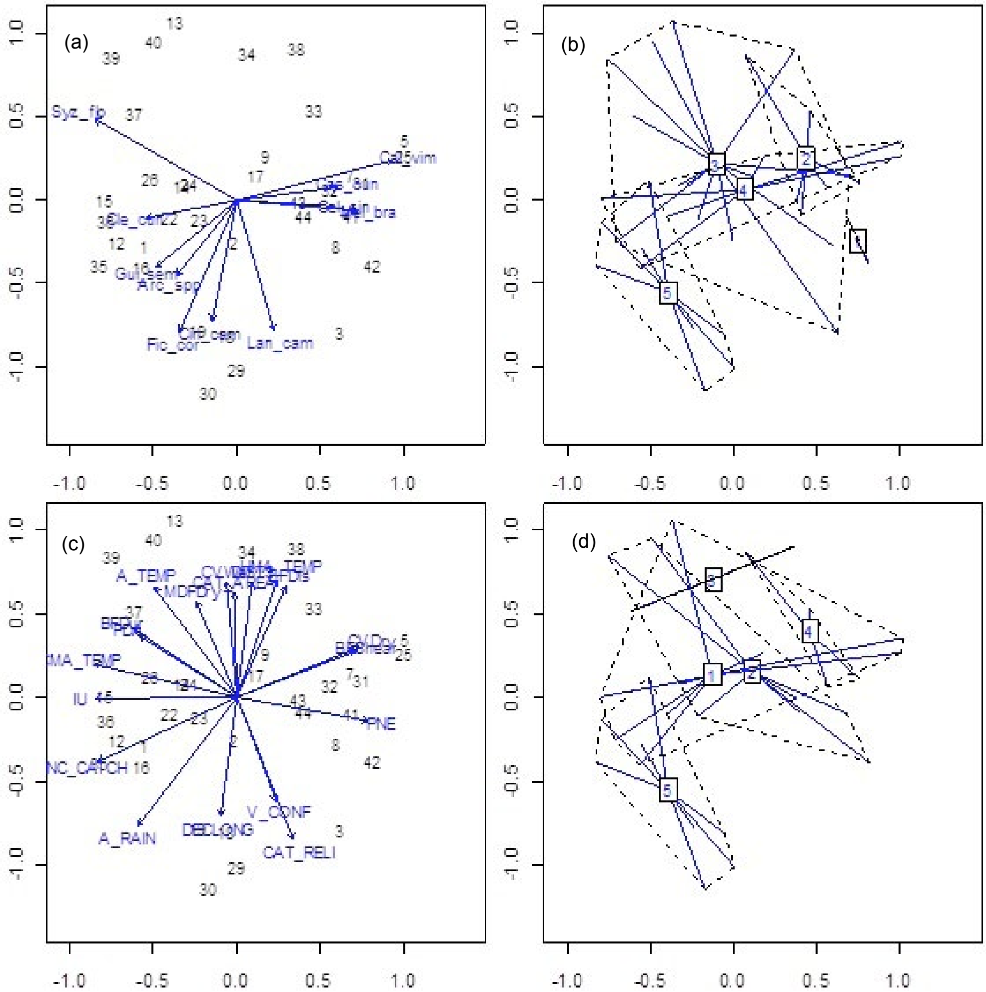


Figure 6.3: Non-metric MDS ordination of sites based on range standardised bankfull riparian vegetation metrics, stress = 0.176, two dimensions

(a) Position of sites in ordination space. Vectors show metrics significantly correlated with the ordination (P=0.01).

(b) Distance to group centroids for sites in each HFC.

(c) Environmental variables significantly correlated with the ordination (P=0.01).

(d) Distance to group centroids for sites in each RFC. See Tables 6.2 and 6.3 for variable acronyms and Table 6.4 for metric acronyms.

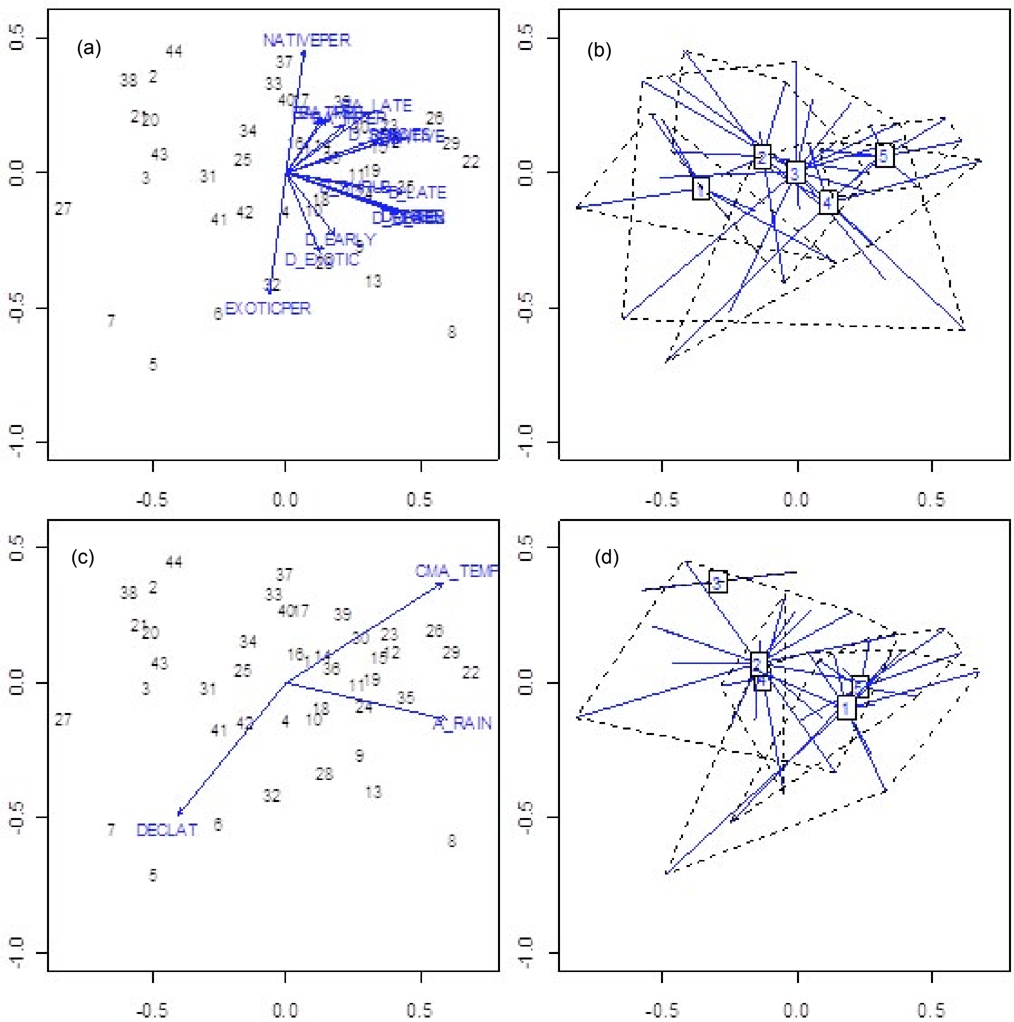


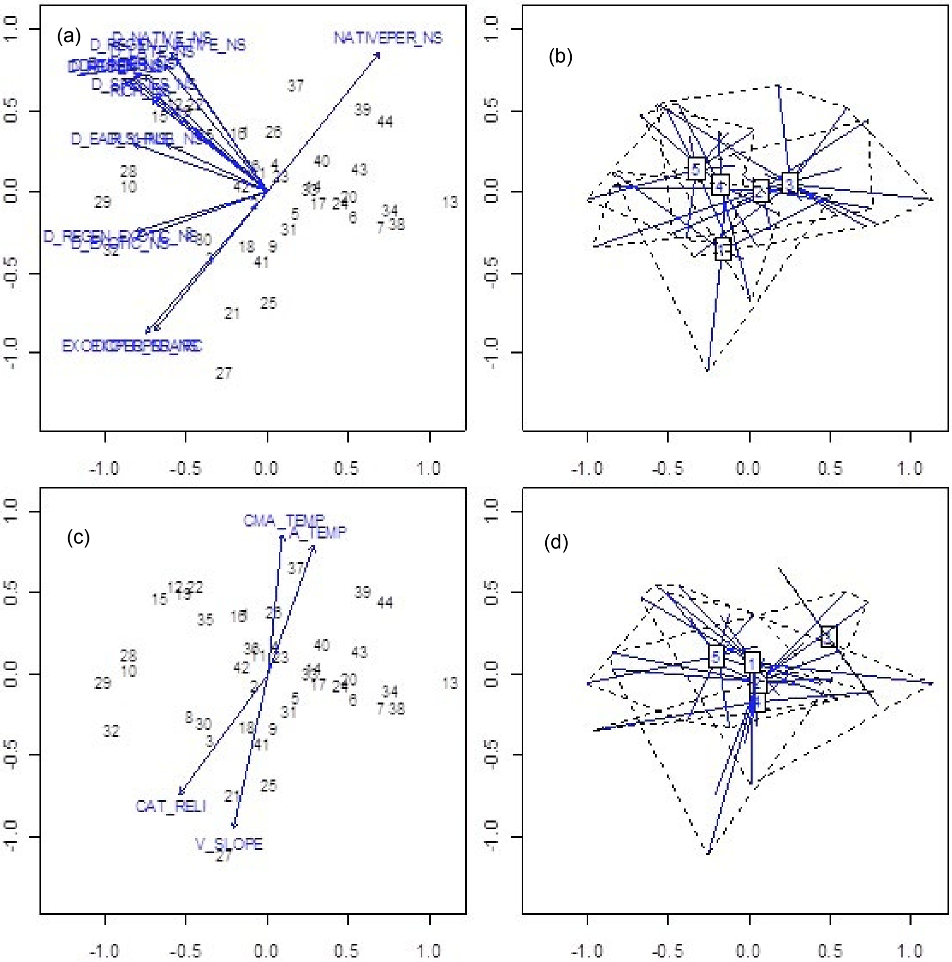
Figure 6.4: Non-metric MDS ordination of sites based on range standardised near-stream riparian vegetation metrics, stress = 0.12, two dimensions

(a) Position of sites in ordination space. Vectors show metrics significantly correlated with the ordination (P=0.01).

(b) Distance to group centroids for sites in each HFC.

(c) Environmental variables significantly correlated with the ordination (P=0.01).

(d) Distance to group centroids for sites in each RFC. See Tables 6.2 and 6.3 for variable acronyms and Table 6.4 for metric acronyms.



Partial CCA analysis suggested flow metrics accounted for approximately 14% of the variance in tree and shrub assemblages independent of influences from other environmental variables (independently explained 16.3%) and land use (independently explained 5.3%) (Figure 6.5; Table 6.9). Partial CCA for the independent influences of hydrology suggest that a relatively similar suite of hydrology variables appeared to be significant in structuring tree and shrub assemblages to the nMDS analyses.

A partial CCA analysis of near-stream vegetation suggested that flow explained less variation in tree and shrub assemblages (9.5 %) independent of influences from other environmental variables and land use compared with the bankfull vegetation (Table 6.9). Other environmental variables explained 19.7% and land use explained 7.0% independently. The significant variables used in the partial CCA analysis of bankfull and near-stream vegetation differed. Initial CCA analysis using hydrology variables only suggested that only CVDry and LSNum were useful predictors for both bankfull and near-stream vegetation assemblages.

Figure 6.5 Canonical Correspondence Analysis biplot showing effects of

(a) Hydrology variables on bankfull riparian tree and shrub assemblages after controlling for the effects of land use and other environmental variables.

(b) Other environmental variables after controlling for the effects of hydrology and land use variables.

(c) Land use after controlling for the effects of hydrology and other environmental variables. See Tables 6.2 and 6.3 for variable acronyms.

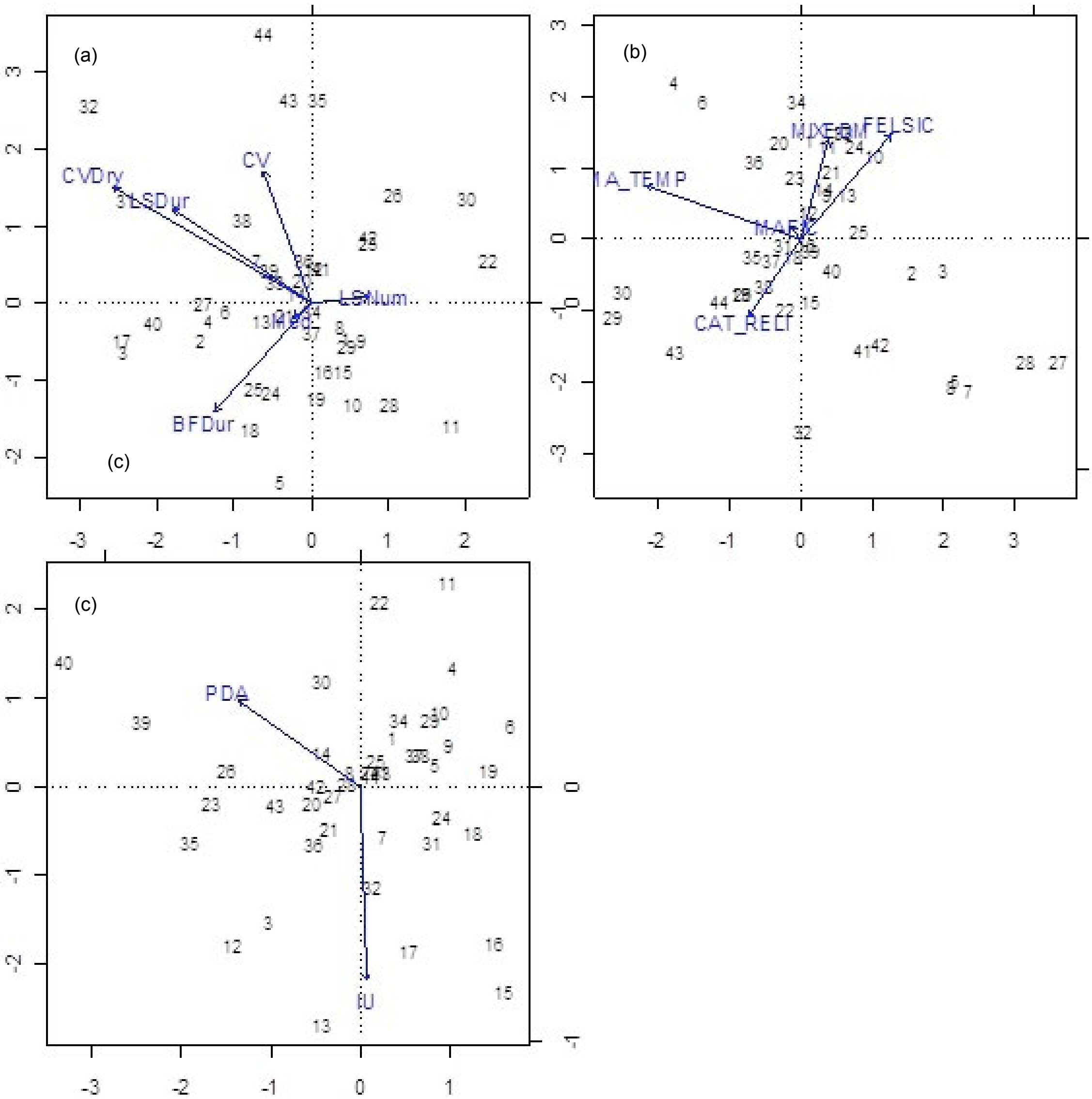


Table 6.9: Results of variance partitioning by partial Canonical Correspondence Analysis

Environmental data range standardised and biotic data log(x)+1-transformed (for x>0) prior to analysis. Variable acronyms given in Tables 6.2 and 6.3.

Component	Bankfull trees and shrubs		Near-stream trees and shrubs	
	Variance explained	Significant variables	Variance explained	Significant variables
Land use	5.28	PDA, IU	6.96	PDA, IU
Hydrologic	14.08	CVDry, BFDur, CV, Med, LSNum, LSDur	9.49	CVDry, CVAnnual, LSNum
Other	16.33	CMA_TEMP, CAT_RELI, FELSIC, MIXEDM, MAFIC	19.71	DECLONG, CMA_TEMP, A_RAIN, FELSIC
Shared hydrology, other environmental variables and land use	4.03		3.39	
Shared hydrology and land use	1.26		0.66	
Shared other environmental variables and land use	1.81		1.81	
Shared hydrology and other environmental variables	7.14		2.04	
Total explained variation	49.9		44.05	
Unexplained variation	50.1		55.95	

Variation in dependent variables explained by the random forests models ranged from 12.4% for the BA_LATE (near stream) model to 63.88% for the BA_EXOTIC (near-stream) model (Table 6.10).

Hydrologic metrics were considered to be relatively important in the regression random forests models of bankfull riparian metrics, as determined by change in mean squared error (Figure 6.6). Hydrology variables describing variation in flows appear to be particularly important. CVDry was in the top five important predictor variables for 6 out of the 11 random forests models (Figure 6.6). Other variables describing variation in flow also appear to be relatively important with CV occurring in the top ten important metrics for five of the random forests models, whilst CVWet and CVAnnual were the most important metrics for BA_EXOTIC and D_REGEN_EXOTIC respectively.

In contrast to the bankfull riparian metric random forest models, variables describing variation in flows (CV, CVAnnual, CVDry and CVWet) appeared to be relatively unimportant in the random forest models of near-stream vegetation metrics. The only exception to this was the near-stream model for BA_EXOTIC where variables describing variation in flow (CVWet, CVAnnual and CV) were the top three most important variables.

Variables describing bankfull flow conditions (particularly BFShear and BFDis) and average flow (particularly Med) were also relatively important in the random forest models of bankfull vegetation metrics. However, variables describing high flow conditions (i.e. HSDur, HSDis and HSNum) were relatively unimportant. Variables describing low flow andbaseflows (e.g. LSDis, LSDur, LSNum and MDBF) were also relatively unimportant and did not occur in the top three variables for any of the random forest models of bankfull riparian vegetation metrics.

Variables describing bankfull flow conditions (particularly BFShear and BFDis) were also relatively important in the random forest models of near-stream vegetation metrics.

Climate variables also appeared to be relatively important in the random forest models of bankfull riparian vegetation metrics, particularly the coldest month mean temperature (CMA_TEMP) which was in the top ten important variables for six of the eleven bankfull riparian metric random forest models.

The random forest's' partial dependence plots of hydrology variables identified as important variables (i.e. the top five variables from the random forest analyses) suggest the presence of potential thresholds in relationships between selected hydrology variables and riparian metrics. Only partial plots for bankfull vegetation are presented here as results for near-stream vegetation were similar. It should be noted when interpreting these plots that the x axis range is standardised to zero mean and unit variance and the y axis range is a relative range not an absolute scale due to averaging so the shape of these plots rather than the value of specific metrics is more meaningful.

After averaging the effects of all other predictor variables, the values of most metrics decreased as the CVDry exceeded -1 (Figure 6.6). This value equates to a CVDry of approximately 0.9. Partial dependence plots of densities of rushes, reeds and sedges (D_LOMAND), densities of native regeneration (D_REGEN_NATIVE) and the basal area of late successional stage species (BA_LATE) all suggest a threshold in average and low flow discharge measures between -1 and 0 on the standardised scale above which metrics increase precipitously (Figure 6.9).

Partial dependence plots between selected riparian metrics and bankfull flow conditions also suggested a number of potential thresholds (Figure 6.10). Densities of intermediate successional stage species (D_INTER), late successional stage species (D_LATE) and regeneration of native species (D_REGEN_NATIVE) decreased precipitously between 0 and 1 on the standardised scale for BFShear (Figure 6.9).

Table 6.10: Summary of regression random forests for selected bankfull and near-stream riparian metrics, and most important variables for each model (as determined by change in mean square error)

Only models explaining greater than 10 % of the variance are presented here. Metric acronyms are given in Table 6.4 and variable acronyms given in Tables 6.2 and 6.3.

	Metric	pseudoR^2	Most important variable (% change in mean square error)	Most important flow variable (% change in mean square error)
Bankfull	RICH	46.38	PDA (11.73)	CV (9.25)
	D_SPECIES	60.77	PDA (10.78)	BFDis (10.26)
	D_NATIVE	25.07	CVDry (9.45)	CVDry (9.45)
	BA_EXOTIC	15.17	CVWet (7.71)	CVWet (7.71)
	D_INTER	13.01	A_RAIN (7.37)	BFShear (5.45)
	D_LATE	25.28	BFShear (6.76)	BFShear (6.76)
	BA_INTER	15.48	MEDAnnual (7.22)	MEDAnnual (7.22)
	BA_LATE	37.66	Med (10.09)	Med (10.09)
	D_SHRUB	49.17	CAT_AREA (9.60)	BFDis (7.45)
	D_LOMAND	16.33	ELEV (10.13)	HSDis (5.36)
	D_REGEN_NATIVE	52.06	Med (10.79)	Med (10.79)
	D_REGEN_EXOTIC	23.46	CVAnnual (9.91)	CVAnnual (9.91)
Near-stream	RICH	13.1	BFDis (8.84)	BFDis (8.84)
	D_SPECIES	14.05	CAT_AREA (8.10)	BFDis (7.96)
	D_SHRUB	26.37	BFDis (26.37)	BFDis (26.37)
	EXOTICPER	23.45	DECLAT (8.72)	CVWet (6.49)
	BA_EXOTIC	63.88	CVWet (7.00)	CVWet (7.00)
	BA_EARLY	24.32	Med (6.6)	Med (6.6)
	BA_LATE	12.37	CMA_TEMP (9.31)	BFNum (5.22)

Figure 6.6: Variable importance (expressed as a change in MSE when that predictor variable is permutated randomly) for selected bankfull riparian vegetation metric random forests models

For clarity only the ten most important variables are shown. See Table 6.4 for metric acronyms and Tables 6.2 and 6.3 for variable acronyms.

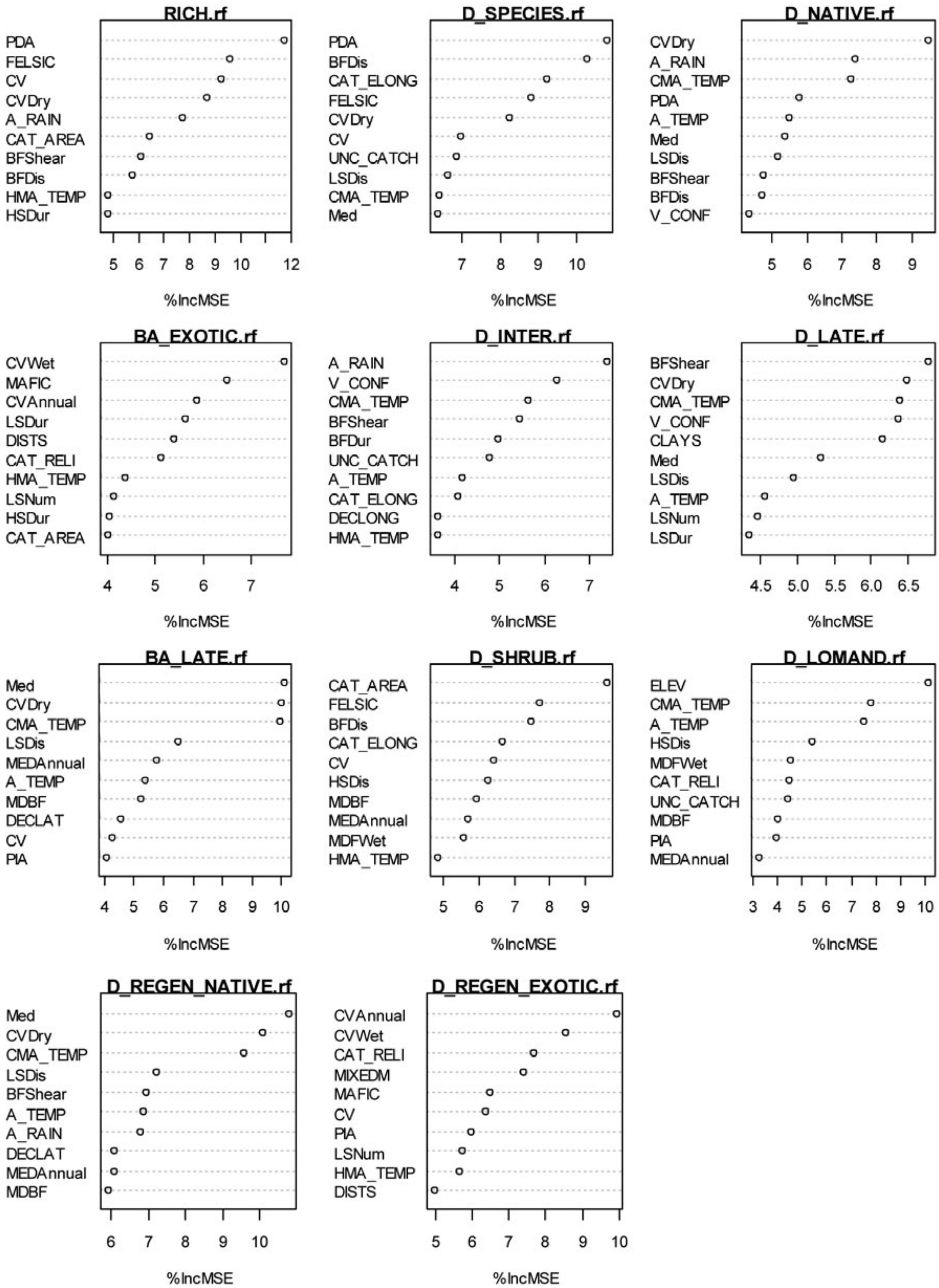


Figure 6.7: Variable importance (expressed as a change in MSE when that predictor variable is permutated randomly) for selected near-stream riparian vegetation metric random forest models

For clarity only the ten most important variables are shown. See Table 6.4 for metric acronyms and Tables 6.2 and 6.3 for variable acronyms.

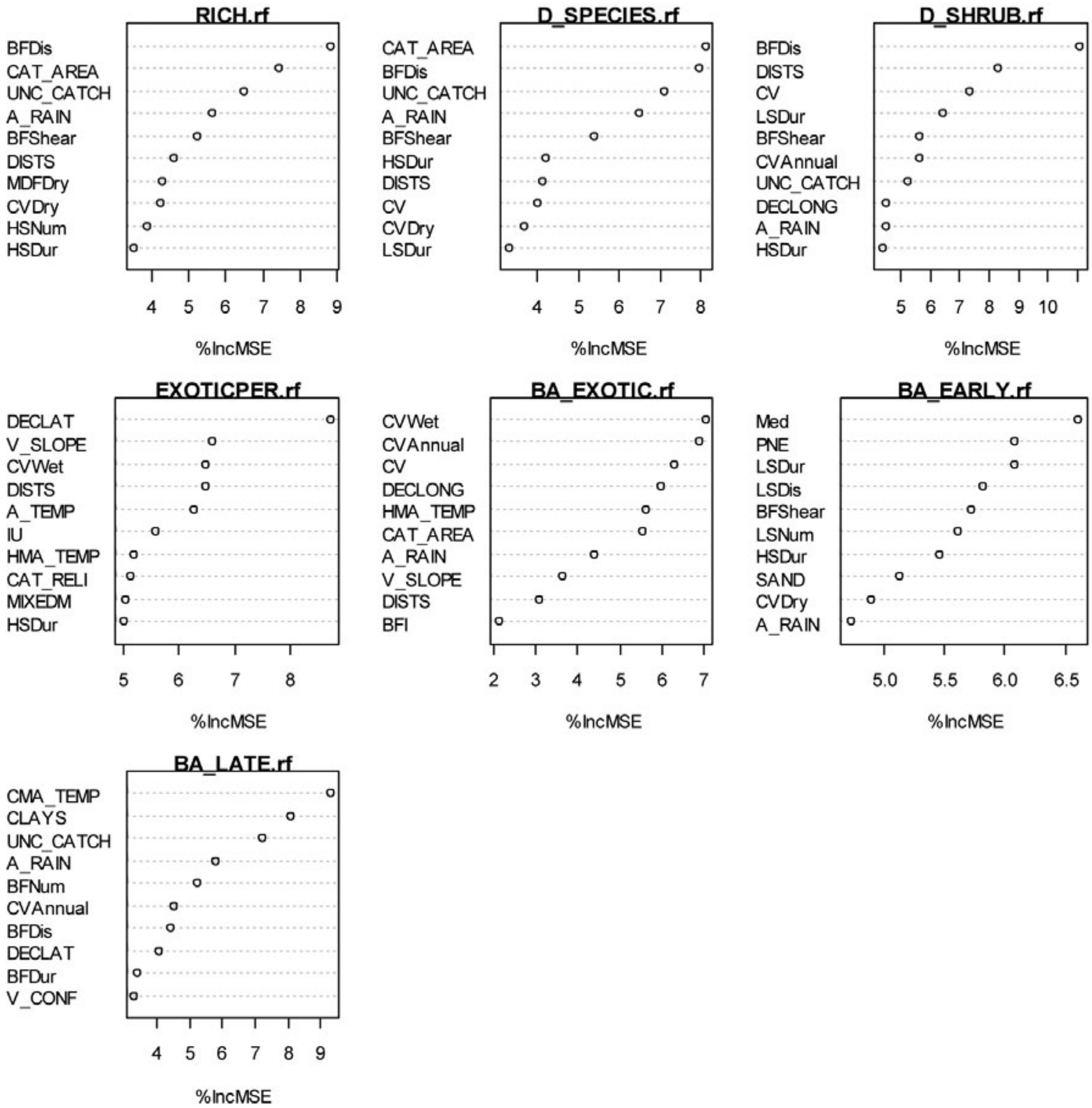


Figure 6.8: Partial dependence plots from random forests predictions of selected bankfull riparian metrics on predictors describing variations in flow (CVDry, CV, CVAnnual and CVWet)

Predictions are only shown for hydrological predictors occurring in the five most important variables for those models explaining greater than 10% of the variation. Partial dependence is the predicted value of the response based on the value of one predictor after averaging out the effects of all other predictors in the model. Variable acronyms are given in Table 6.2.

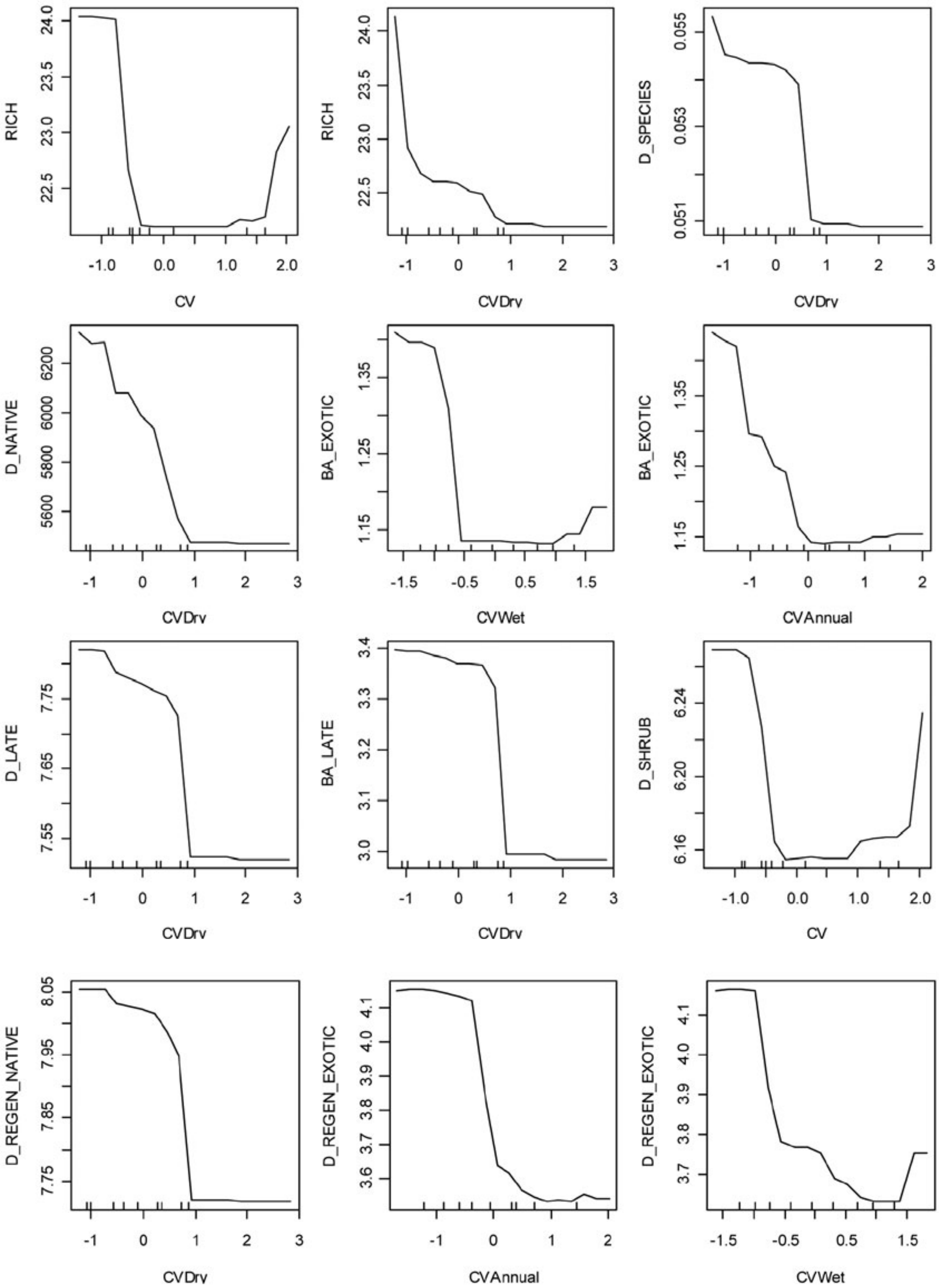


Figure 6.9: Partial dependence plots from random forests predictions of selected bankfull riparian metrics on predictors describing average and low flow conditions (LSDur, Med, LSDis and MDFWet)

Predictions are only shown for hydrological predictors occurring in the five most important variables for those models explaining greater than 10% of the variation. Partial dependence is the predicted value of the response based on the value of one predictor after averaging out the effects of all other predictors in the model. Variable acronyms are given in Table 6.2.

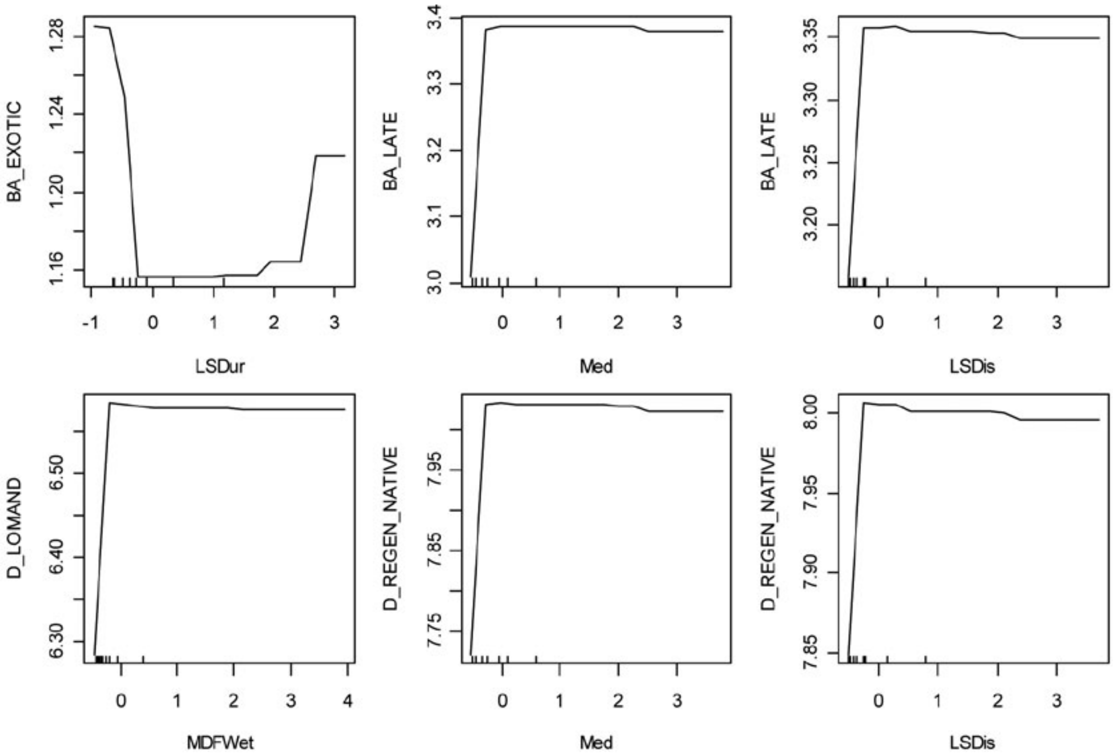


Figure 6.10: Partial dependence plots from random forests predictions of selected bankfull riparian metrics on predictors describing high and flood conditions (BFDis, BFShear, BFDur and HSDis)

Predictions are only shown for hydrological predictors occurring in the five most important variables for those models explaining greater than 10% of the variation. Partial dependence is the predicted value of the response based on the value of one predictor after averaging out the effects of all other predictors in the model. Variable acronyms are given in Table 6.2.

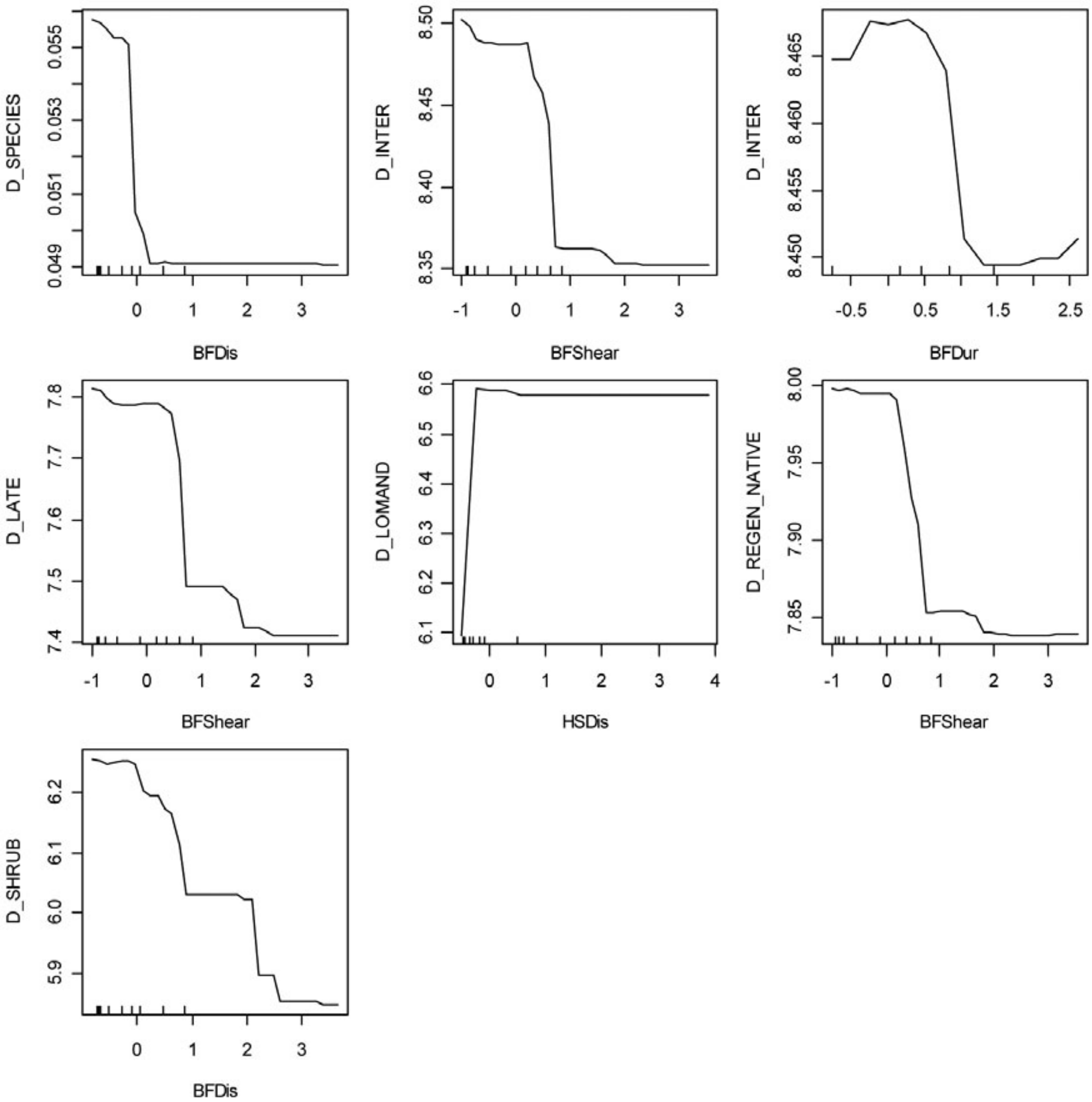


Table 6.11: Summary of GLS regression models describing relationships between selected bankfull (BF) and near-stream (NS) riparian metrics and flow metrics

D *is the value of the log-likelihood ratio test comparing model fits (i.e. the null model to the alternate model). Significance for* D *is determined by comparison with X^2 with df=1 (3.841). Significant alternative models are in bold text.*

Metric (transformation)	Dataset	Flow metric	Intercept	Slope	logLike	logLike (null model)	D
RICH	BF	CVDry 1	49.663 (7.516)	-20.347 (5.079)	-116.455	-127.416	**21.922**
	NS	BFDis 2	9.813 (1.046)	-4.400 x 10^{-5} (1.32 x 10^{-5})	-102.215	-97.740	**-8.951**
D_SPECIES (log10+1)	BF	CVDry 1	0.125 (0.023)	-0.054 (0.016)	66.660	65.284	2.752
	BF	BFDis 2	0.070 (0.007)	3.4 x10^{-7} (1.08 x 10^{-7})	54.923	65.436	**-21.025**
	NS	BFDis 2	0.121 (0.013)	-5.3 x 10^{-7} (1.63 x 10^{-7})	39.076	48.224	**-18.295**
	NS	BFShear 3	0.125 (0.016)	-3.963 x 10^{-4} (1.579 x 10^{-4})	41.994	47.294	**-10.599**
D_NATIVE (log10+1)	BF	CVDry 1	10.470 (0.405)	-1.564 (0.350)	-30.864	-41.881	**22.034**
D_INTER (log10+1)	BF	BFShear 3	8.735 (0.197)	-3.318 x 10^{-3} (2.124 x 10^{-3})	-40.811	-36.686	**-8.250**
D_LATE (log10+1)	BF	CVDry 1	11.237 (0.611)	-2.581736 0.5898811	-45.733	-62.318	**33.170**
BA_LATE	BF	CVDry 1	171.449 (35.180)	-88.095 (24.296)	-166.908	-178.174	**22.531**
D_REGEN_NATIVE (log10+1)	BF	CVDry 1	10.510 (0.486)	-1.865 (0.428)	-37.033	-49.773	**25.48**

1. Outlier sites 43 and 44 removed 2. Outlier sites 37and 38 removed 3. Outlier sites 5 and 41 removed

Table 6.12: Summary of quadratic models describing relationships between selected bankfull (BF) and near-stream (NS) riparian metrics and flow metrics

D is the value of the log-likelihood ratio test comparing model fits (i.e. the null model to the alternate model). Significance for D is determined by comparison with X^2 with df=1 (3.841). Significant alternative models are in bold text.

Metric	Dataset	Flow metric	Intercept	X	X^2	logLike	logLike (null model)	D
RICH	BF	CV	118.226	-24.092	1.346	-121.510	-135.599	**28.178**
	NS	BFShear 3	8.172	0.062	-2.888 x 10-4	-95.790	-98.644	**5.708**

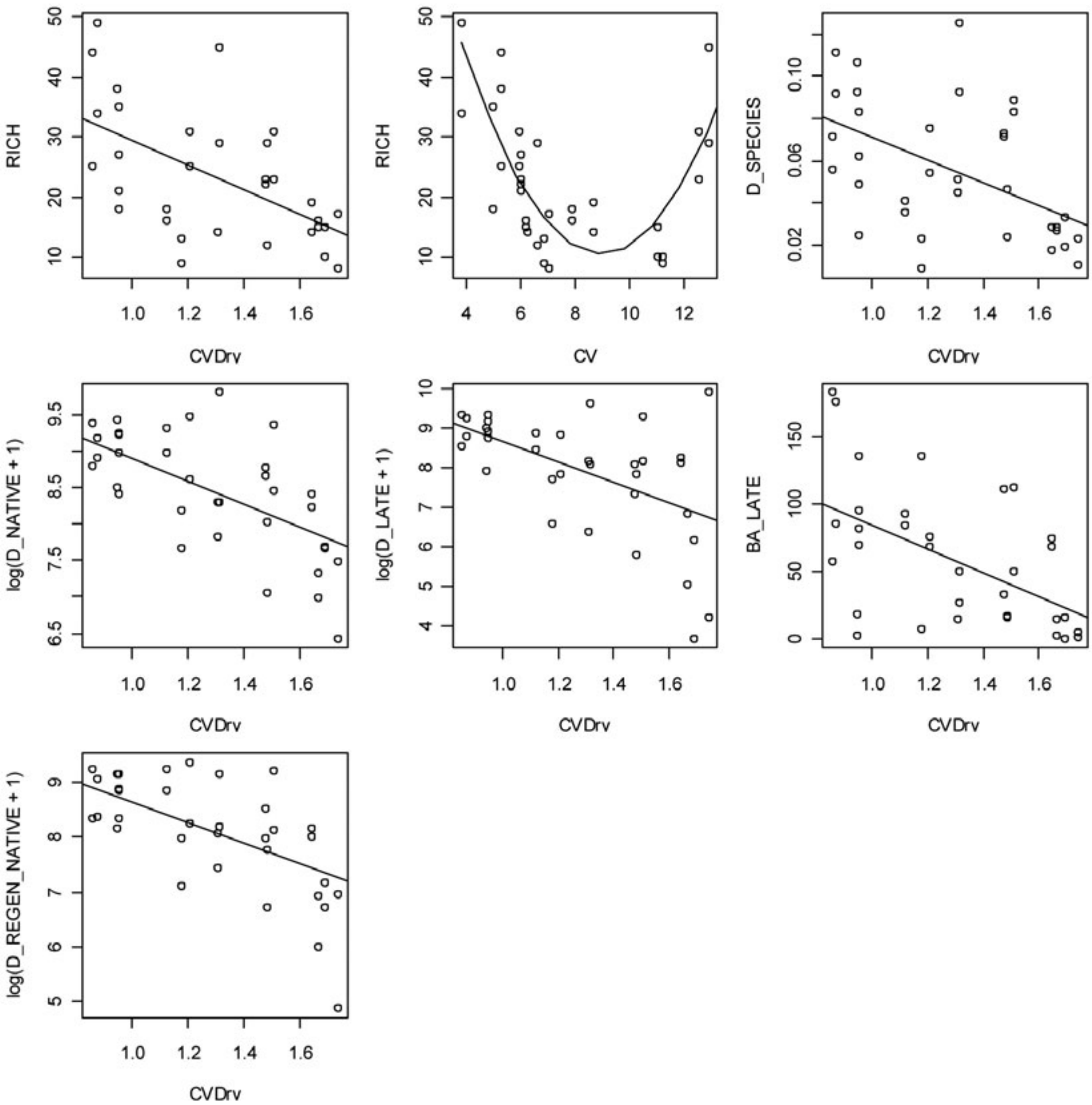
3. Outlier sites 5 and 41 removed

Analysis of relationships between selected flow metrics and riparian vegetation metrics suggested a number of significant relationships (Tables 6.11 and 6.12). Thirteen relationships were identified as significant. The CV of dry season flows (May to October) was a significant predictor in six of these models.

Relationships were negative for these metrics indicating that as CVDry increased values for riparian metrics RICH, D_SPECIES, D_NATIVE, D_LATE, BA_LATE and D_REGEN_NATIVE decreased linearly (Figure 6.11). A significant quadratic relationship was found between species richness (RICH) and the coefficient of variation in annual flows (CV), with lowest values of species richness found at intermediate values of CV.

Figure 6.11 Scatter graphs showing significant linear or quadratic relationships between variables describing variation in flow (CVDry and CV) and selected riparian vegetation metrics

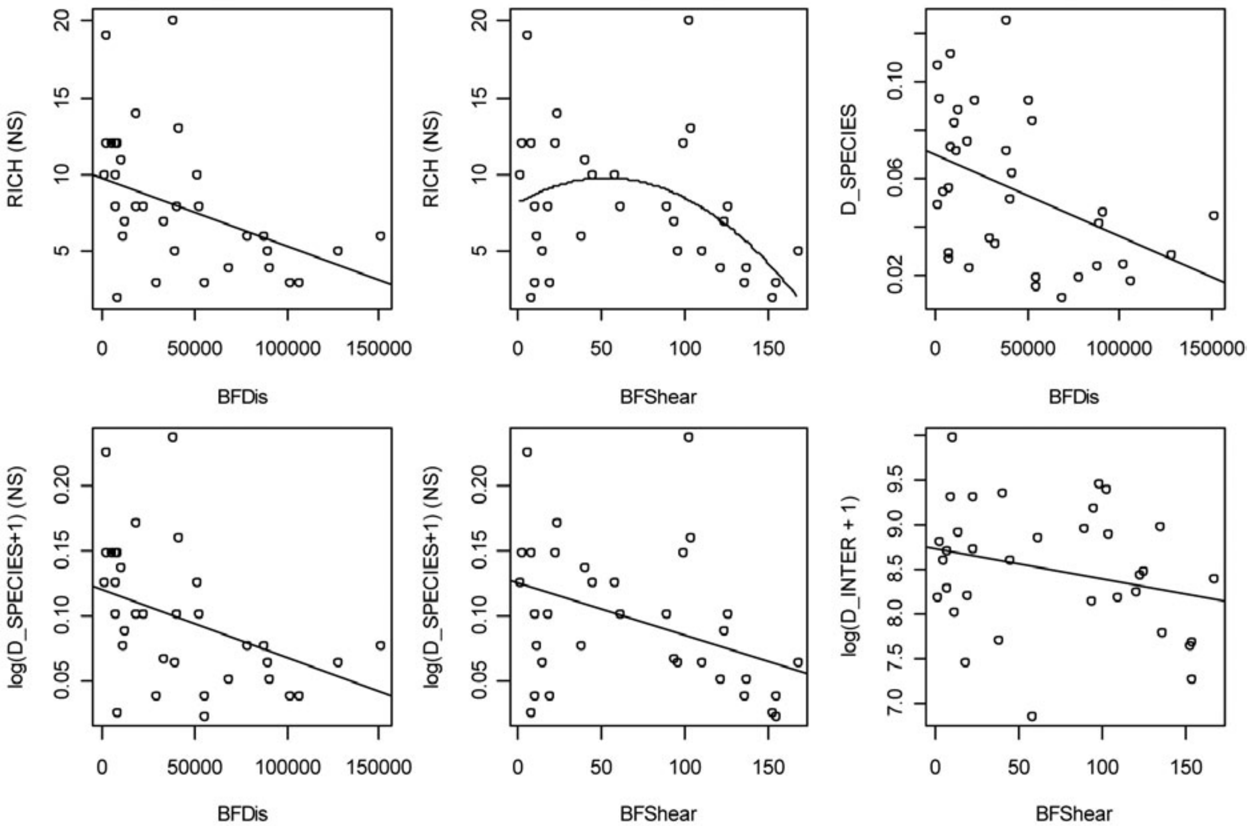
Metrics are bankfull metrics unless indicated otherwise. NS = near-stream metrics. Equation coefficients are given in Tables 6.11 and 6.12. Flow variable acronyms given in Table 6.2 and riparian metric acronyms given in Table 6.4.



Models of selected riparian metrics (RICH (NS), D_SPECIES (NS) and D_INTER) also suggested a quadratic relationship with metric values maximized at intermediate values of bankfull shear stress (BFShear) (Figure 6.12). However, a significant quadratic relationship was only found between near-stream species richness (RICH (NS)) and BFShear. A negative linear relationship was also found between bankfull discharge (BFDis) and three of the riparian metrics (RICH (NS), D_SPECIES (NS) and D_INTER) (Figure 6.12).

Figure 6.12: Scatter graphs showing significant linear or quadratic relationships between variables describing bankfull flow conditions (BFShear and BFDis) and selected riparian vegetation metrics

Metrics are bankfull metrics unless indicated otherwise. NS = near-stream metrics. Equation coefficients are given in Tables 6.11 and 6.12. Flow variable acronyms given in Table 6.2 and riparian metric acronyms given in Table 6.4.



Hypothesis 1 and 2 summary

- The results of the analyses support Hypothesis 1. Flow appears to play a role in structuring the riparian vegetation assemblages of the SEQ region. Both nMDS and partial CCA suggest a suite of flow metrics including those describing high flow and flood condition, low and average flow conditions and variation in flows were important in structuring tree and shrub assemblages. A metric of particular importance appear to be the coefficient of variation in dry season flows (CVDry) which was identified in both the nMDS and partial CCAs.
- The results of the partial CCA analyses suggest flow accounted for approximately 14% of the variation in bankfull tree and shrub riparian assemblage structure independent of other environmental and land use influences.
- The independent influence of flow appeared to be greater for bankfull vegetation (14.1 % variance explained) relative to its influence on near-stream vegetation assemblage structure (9.5% variance explained).
- A number of flow metrics were identified as important in influencing riparian vegetation metrics in particular CVDry.
- Riparian vegetation metrics exhibited some patterns across selected flow metric gradients. A number of riparian vegetation metrics showed relationships with CVDry, BFDis and BFShear. These relationships were found to be both linear (CVDry) and quadratic (CV and BFShear) in shape.
- Results of the analyses support Hypothesis 2. Environmental influences other than flow explained approximately 16% of the variance in riparian tree and shrub assemblages whilst land use explained around 5%. The analysis suggested around 7% of variance was shared between the hydrology and other environmental variables.Patterns between riparian vegetation metrics and flow metrics were more mixed. No flow metrics were correlated with the nMDS ordinations of riparian metrics, with only a small subset of other environmental influences (CMA_Temp, A_RAIN and DECLAT). Random forests regressions, however, suggested that flow did play a roll (although usually a lesser roll compared to other environmental and land use influences). Variables describing flow variation (CVDry and CV) appeared to be particularly important.

6.4.3 Hypothesis 3

ANOSIM revealed significant differences across HFCs (Global ANOSIM *R* = 0.294, P=0.001) and across the RFCs (Global ANOSIM *R* = 0.211, P=0.002) in bankfull tree and shrub assemblage data (Table 6.13). Results of the permutational Multivariate Analysis of Variance (PERMANOVA) also support the hypothesis of significant differences across RFCs and HFCs (not presented here). Group dispersions were not significantly different, suggesting that significant differences were due to difference in group means.

ANOSIM of near-stream tree and shrub assemblages revealed significant differences across HFCs (Global ANOSIM *R* = 0.228, P=0.001) but no significant differences across the RFCs (Global ANOSIM *R* = 0.074, P=0.096) (Table 6.13).

Table 6.13: ANOSIM pairwise comparison p values across RFCs and HFCs for bankfull and near-stream tree and shrub assemblage data

Significant differences are in bold text.

Pairwise comparisons	Bankfull trees and shrubs		Near-stream trees and shrubs	
	Reference	Historic	Reference	Historic
1 vs 2	**0.018**	**0.006**	No significant differences	0.074
1 vs 3	0.379	0.108		**0.025**
1 vs 4	0.082	**0.022**		0.413
1 vs 5	0.051	**0.001**		**0.001**
2 vs 3	0.5	0.553		0.528
2 vs 4	0.639	**0.002**		0.097
2 vs 5	**0.002**	**0.001**		**0.001**
3 vs 5	0.467	**0.002**		**0.004**
4 vs 3	**0.045**	0.113		0.082
4 vs 5	**0.003**	**0.021**		**0.016**

Indicator species analysis suggests that with the exception of HFC 5, there were relatively few species with significant indicator values (Table 6.14). HFC 1 and RFC 4 were indicated by only a few species (*Melaleuca bracteata*, and the exotic tree species *Celtis sinensis*) typically associated with lower rainfall inland sites. HFC 5 was associated with a suite of largely rainforest species including a number of species not generally considered obligate riparian. HFC 3 and RFCs 1, 2, 3 and 5 did not have any significant indicator species.

Table 6.14: Indicator species for bankfull tree and shrub assemblages (using Dufrene–Legendre Indicator Species Analysis) for RFCs and HFCs

Only indicator species where p≤0.05 are shown. Species acronyms are given in Attachment 6.1.

Class	Species	Flow class	Indicator value	p
Historic flow classes	Mel_bra	1	0.4406	0.003
	Cas_cun	1	0.3461	0.027
	Cel_sin	1	0.314	0.046
	Cal_vim	2	0.357	0.018
	Tri_lau	4	0.5812	0.001
	Ela_obo	4	0.3977	0.015
	Gui_sem	4	0.3778	0.012
	Arc_spp	5	0.625	0.001
	Cin_oli	5	0.594	0.002
	Cin_cam	5	0.5861	0.001
	Neo_dea	5	0.5301	0.001
	Eur_fal	5	0.5	0.003
	Cry_obo	5	0.4946	0.001
	End_pub	5	0.4499	0.004
	Slo_aus	5	0.448	0.003
	Ard_cre	5	0.3879	0.01
	Myr_var	5	0.3819	0.006
	Her_tri	5	0.3097	0.034
	End_dis	5	0.2783	0.047
	Dio_pen	5	0.2571	0.05
Reference flow classes	Euc_spp	4	0.5382	0.002
	Cel_sin	4	0.5333	0.034
	Mel_bra	4	0.5299	0.022
	Cal_vim	4	0.367	0.045

Random forests models showed that tree and shrub composition data were poor indicators of flow class. Prediction errors were worse for shrub and tree data separately, so only the combined tree and shrub confusion matrices are shown for bankfull vegetation (Table 6.15) and near-stream vegetation (Table 6.16).

Error rates were relatively similar for Reference and Historic flow classes for bankfull vegetation, but error rates were higher for RFCs compared with HFCs for near-stream vegetation only. For bankfull vegetation HFC 5 had the lowest prediction error and HFCs 1 and 4 the highest prediction error.

Prediction for the RFCs from the bankfull vegetation was lowest for RFC 2 and highest for RFCs 3 and 4 where none of the sites were correctly classified (error rate 100%). For the near-stream vegetation, HFC 3 had the lowest prediction error whilst HFC 4 had the highest error rate. Predictions for the RFCs from near-stream vegetation were similar to those of the bankfull vegetation in that RFC 2 had the lowest prediction error and RFC 3 the highest prediction error (100%).

Table 6.15: Confusion matrices for bankfull tree and shrub species cover random forests models

The confusion matrix shows how well each biotic dataset can be used to predict the HFC and RFC membership of sites. The number of trees grown for each model was 1000 and, 16 taxa (Historic) or 14 taxa (Reference) were used at each split. OOB error rate estimate for the Historic model was 47.7% and for the Reference model 47.5%.

Observed flow class	HFC predicted by random forests model					Error (%)
Tree and shrub composition	HFC1	HFC2	HFC3	HFC4	HFC5	
HFC1	2	0	4	0	0	66.7
HFC2	1	3	4	0	0	62.5
HFC3	3	1	9	0	3	43.7
HFC4	1	0	2	2	1	66.7
HFC5	0	0	1	0	7	12.5
Observed flow class	**RFC predicted by random forests model**					**Error (%)**
Tree and shrub composition	**RFC1**	**RFC2**	**RFC3**	**RFC4**	**RFC5**	
RFC1	4	3	0	0	3	60.0
RFC2	3	10	0	0	1	28.6
RFC3	0	1	0	1	0	100
RFC4	0	4	0	0	0	100
RFC5	1	2	0	0	7	30.0

Table 6.16: Confusion matrix for NS tree and shrub species cover random forests models

The confusion matrix shows how well each biotic dataset can be used to predict the HFC and RFC membership of sites. The number of trees grown for each model was 1000 and, 16 taxa (Historic) or 14 taxa (Reference) were used at each split. OOB error rate estimate for the Historic model was 43.2% and for the Reference model 65%.

Observed flow class	HFC predicted by random forests model					Error (%)
Tree and shrub composition	HFC1	HFC2	HFC3	HFC4	HFC5	
HFC1	2	1	3	0	0	66.7
HFC2	1	6	1	0	0	25.0
HFC3	0	0	14	1	1	12.5
HFC4	1	0	3	0	2	100
HFC5	0	0	5	0	3	62.5
Observed flow class	**RFC predicted by random forests model**					**Error (%)**
Tree and shrub composition	**RFC1**	**RFC2**	**RFC3**	**RFC4**	**RFC5**	
RFC1	3	4	0	0	3	70.0
RFC2	5	8	0	1	0	42.9
RFC3	0	2	0	0	0	100
RFC4	0	3	0	1	0	75.0
RFC5	5	3	0	0	2	80.0

Kruskal–Wallis tests indicated that nine vegetation metrics varied significantly between HFCs (Figure 6.13) whilst twelve vegetation metrics varied significantly across RFCs (Figure 6.14). Pairwise comparisons, however, suggested only one significant difference between classes. Species richness (RICH) was significantly higher in HFC 5 compared with HFC 1.

Fewer differences were found across the HFCs and RFCs for the near-stream vegetation in comparison to the bankfull vegetation metrics (data not shown). RICH, D_SPECIES, EXOTICPER, NATIVEPER, BA_LATE and BA_LATE were significantly different across the HFCs, but pairwise comparisons did not reveal significant differences. Only D_SPECIES differed significantly across the RFCs.

Figure 6.13: Box and whisker plots of riparian metrics for individual HFCs where Kruskal–Wallis tests showed significant differences in bankfull metric values between HFCs

Multiple comparison tests (Tukey's HSD) are also shown (Bonferroni-corrected significance for each test is α/10 = 0.005) where applicable. See Table 6.4 for metric acronyms.

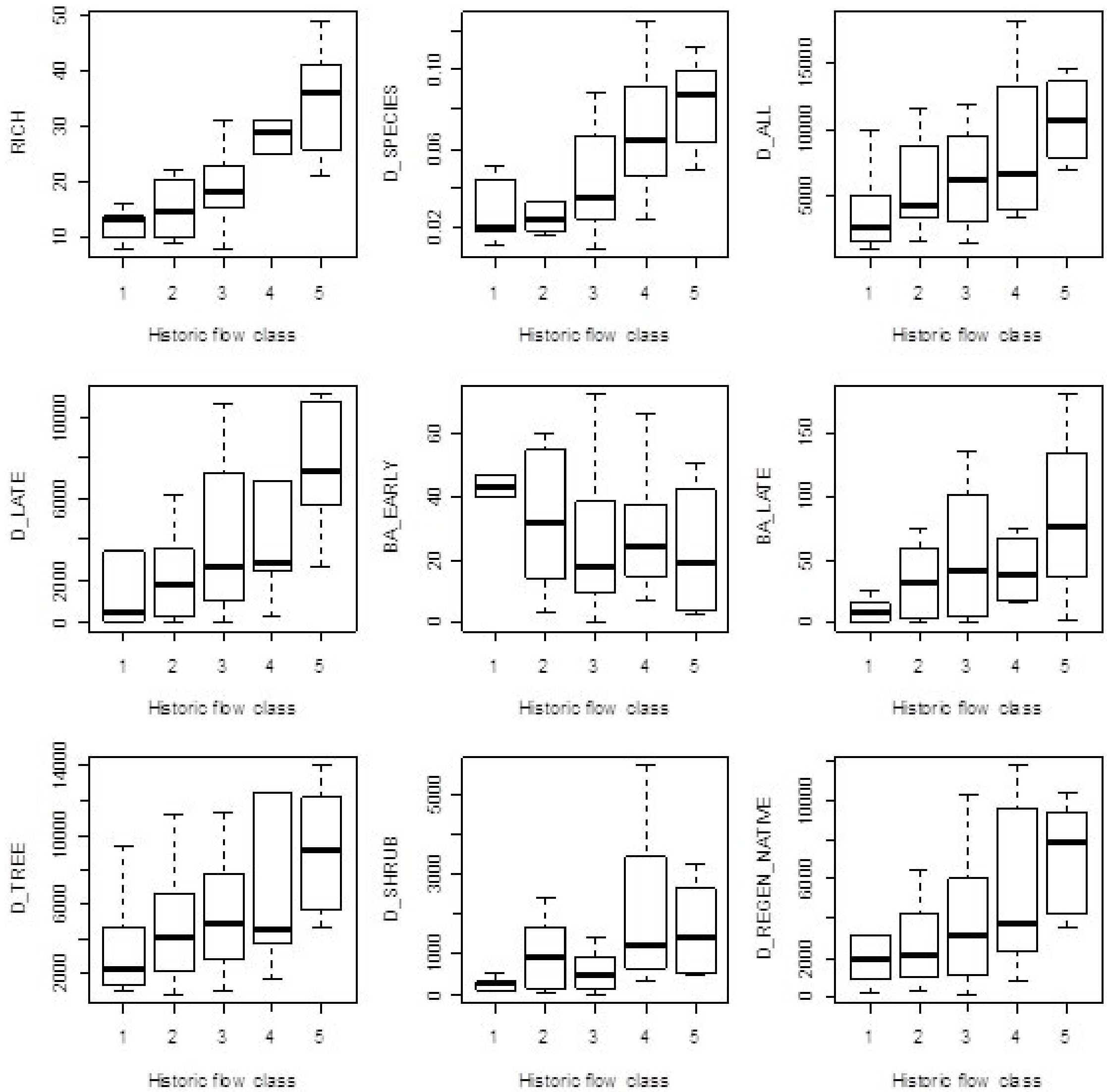
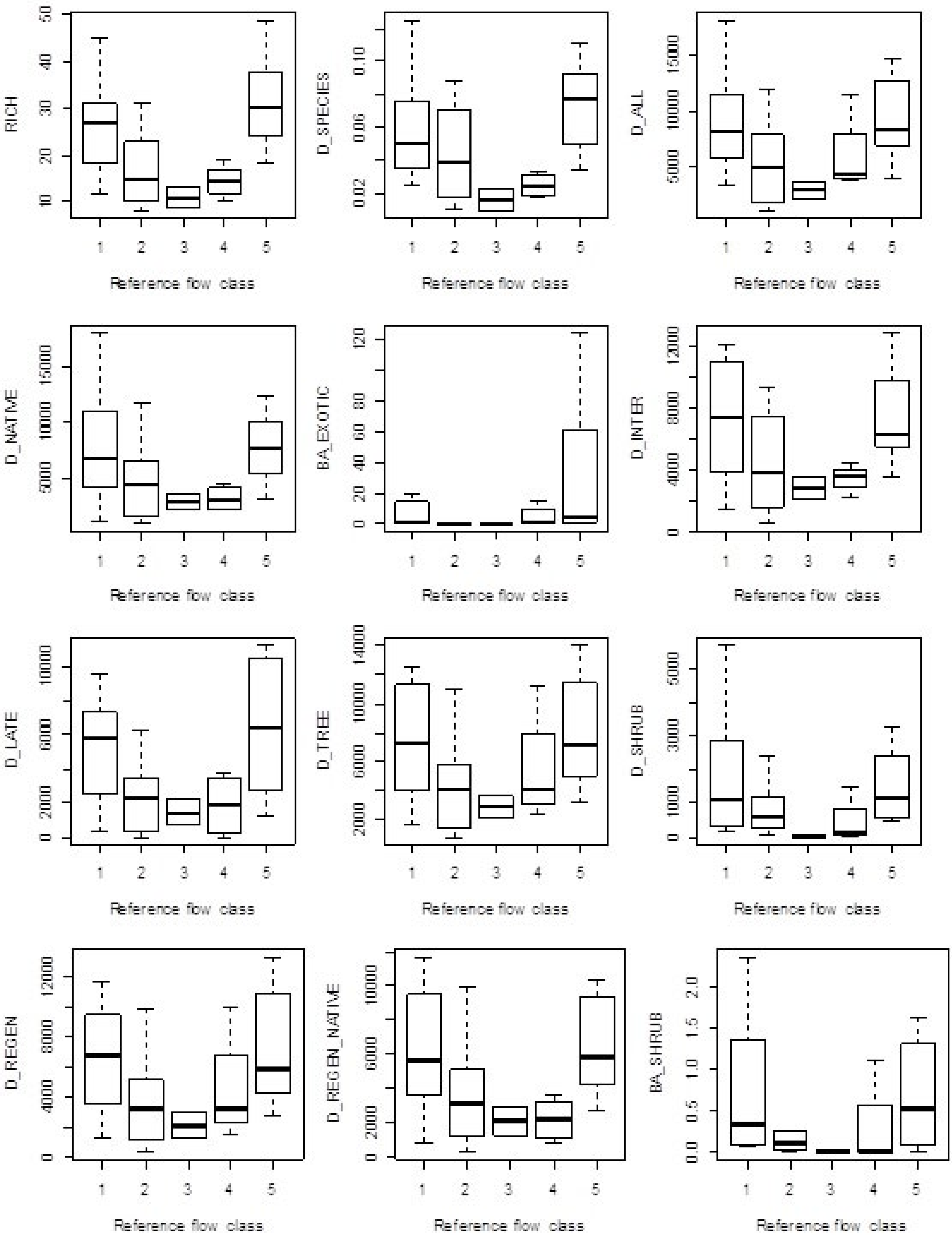


Figure 6.14: Box and whisker plots of bankfull riparian metrics for individual RFCs where Kruskal–Wallis tests showed significant differences in metric values between RFCs

Multiple comparison tests (Tukey's HSD) (Bonferroni-corrected significance for each test is α/10 = 0.005) did not return any significant differences between classes for any of the metrics. See Table 6.4 for metric acronyms.



Hypothesis 3 summary

- Results of the analyses provide mixed support for Hypothesis 3. Differences in bankfull tree and shrub assemblages across both RFCs and HFCs were evident. However, significant differences in near-stream tree and shrub assemblages were found for HFCs but not reference flow classes. Furthermore, tree and shrub assemblage data (bankfull and near-stream) was a relatively poor indicator of flow class (Historic and Reference).
- A number of the riparian bankfull and near-stream metrics also differed significantly across both HFCs and RFCs.
- Differences across flow classes may, however, be attributed to other environmental influences. In particular, climate variables such as the coldest monthly average temperature (CMA_TEMP) and annual rainfall (A_RAIN) both differed significantly across classes and were also identified as important variables in the random forests regression analyses.

6.4.4 Hypothesis 4

ANOSIM analysis revealed no significant differences in tree and shrub assemblages between regulated/supplemented and unregulated/unsupplemented sites for bankfull vegetation (Global ANOSIM R = -0.074, P=0.807) or near-stream vegetation (Global ANOSIM R = 0.029, P=0.32) overall. For riparian vegetation metrics, significant differences between regulated/supplemented and unregulated/unsupplemented were observed only for the riparian bankfull metrics D_SPECIES and BA_LATE when sites strongly impacted by flow regulation were compared (thus excluding sites 12, 13, 15 and 16 which are only weakly impacted by regulation). No significant differences were found between any metrics for either bankfull or near-stream vegetation when all regulated/supplemented sites were included in the analyses.

Within specific flow classes for which comparisons between regulated/supplemented and unregulated/unsupplemented could be made (Table 6.7), significant differences in bankfull tree and shrub assemblages were observed between regulated/supplemented and unregulated/unsupplemented for RFC 5 (Global ANOSIM R = 0.504, p = 0.022) and HFC 2 (Global ANOSIM R = 0.5, p = 0.036).

Other flow classes (RFCs 1 and 2 and, HFCs 1, 3 and 4) were not found to differ significant in assemblage structure between regulated/supplemented and unregulated/unsupplemented sites. No significant differences in near-stream tree and shrub assemblages between regulated/supplemented and unregulated/unsupplemented sites could be detected amongst the RFCs (1, 2 and 5) or HFCs (1, 2, 3 and 4).

The effect of flow regime on five bankfull riparian vegetation metrics (species richness (RICH), species density (D_SPECIES), Native vegetation densities (D_NATIVE), rushes, reeds and sedges (D_LOMAND) and the density of native tree regeneration (D_REGEN_NATIVE) was determined using PLS regression (Table 6.18). Reference models based on non-regulation sites only were generally poor (Table 6.17, Figure 6.16) with R^2 values less than 0.5 for all five metrics.

Table 6.17: PLS regression of species richness (RICH), species density (D_SPECIES), density of natives (D_NATIVE), density of reeds, rushes and sedges (D_LOMAND) and density of native regeneration (D_REGEN_NATIVE) in relation to environmental characteristics (non flow related) and land use

Comp = component. Independent variable acronyms are given in Table 6.3.

Dependent variable	R^2	Comp	Climate			Topography							Substrate		Land use			
			A_RAIN	A_TEMP	CMA_TEMP	DEC-LAT	ELEV	B_SLOPE	BFWIDTH_DEP	CAT_AREA	CAT_ELONG	CAT_RELI	FELSIC	UNC-CAT	PDA	IU	PIA	PNE
RICH[1]	0.49	1	**0.50**	0.14	**0.48**	na	na	na	na	-0.23	na	na	-0.34	**0.49**	0.35	0.29	na	na
		2	0.15	**-0.57**	**-0.38**	na	na	na	na	**-0.52**	na	na	-0.33	-0.36	-0.17	-0.41	na	na
		3	-0.10	**0.58**	**0.50**	na	na	na	na	0.56	na	na	**-0.46**	**-0.51**	-0.14		na	na
		4	**0.49**	-0.32		na	na	na	na	-0.16	na	na	**0.62**		-0.45	**0.60**	na	na
D_SPECIES[2]	0.42	1	**0.42**	na	**0.46**	na	na	0.26	-0.24	-0.21	na	na	-0.38	**0.43**	0.39	na	na	na
D_NATIVE[2]	0.46	1	0.30	0.39	**0.55**	**-0.40**	na	**0.21**	na	na	na	na	na	na	na	0.40	-0.18	-0.31
D_LOMAND[2]	0.36	1	na	**0.53**	**0.50**	na	**-0.53**	na	na	na	-0.20	-0.43	na	na	na	na		na
D_REGEN_NATIVE[2]	0.40	1	0.28	**0.53**	**0.66**	**-0.52**	na	na	na	na	na	na	na	na	na	na	na	na
		2	**0.75**	**-0.50**		**0.48**	na	na	na	na	na	na	na	na	na	na	na	na

Notes: The values indicate the variable loadings. The three highest loadings in each model are in bold text. Na indicates that the independent variable was not used in the pls model for that particular dependent variable.1. Richness (RICH) metric square root transformed. 2. Species density (D_SPECIES), density of natives (D_NATIVE), density of reeds, rushes and sedges (D_LOMAND) and density of native regeneration log10(x)+1 transformed.

Figure 6.16: Reference models (from PLS) for selected transformed riparian vegetation metrics: species richness (RICH) (square root transformed) and species density (S_DENSITY), density of natives (D_NATIVES), density of reeds, rushes and sedges (D_LOMAND) and density of native regeneration (D_REGEN_NATIVE) (log10(x)+1 transformed) using unregulated/unsupplemented sites only

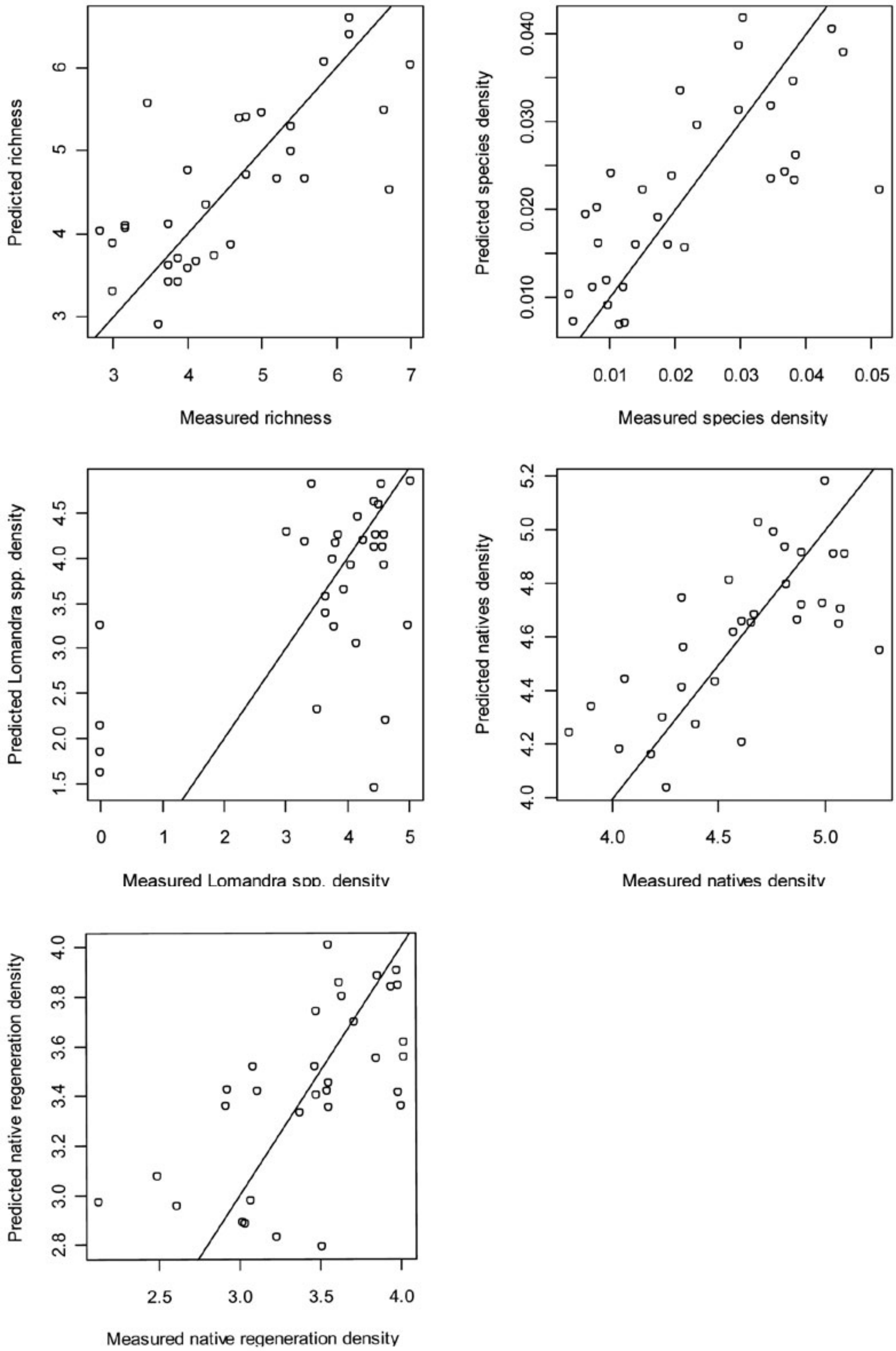


Figure 6.17: Predicted (based on PLS Reference model) versus observed for regulated/supplemented sites for selected riparian vegetation metrics: species richness (RICH) (square root transformed) and species density (S_DENSITY), density of natives (D_NATIVES), density of reeds, rushes and sedges (D_LOMAND) and density of native regeneration (D_REGEN_NATIVE) (log10(x)+1 transformed). Regulated/supplemented site labels: Nerang River (4 and 6), Yabba Creek (10 and 11), Obi Obi (12 and 13), Six Mile (15 and 16), Reynolds Creek (20 and 21) and Burnett Creek (27 and 28).

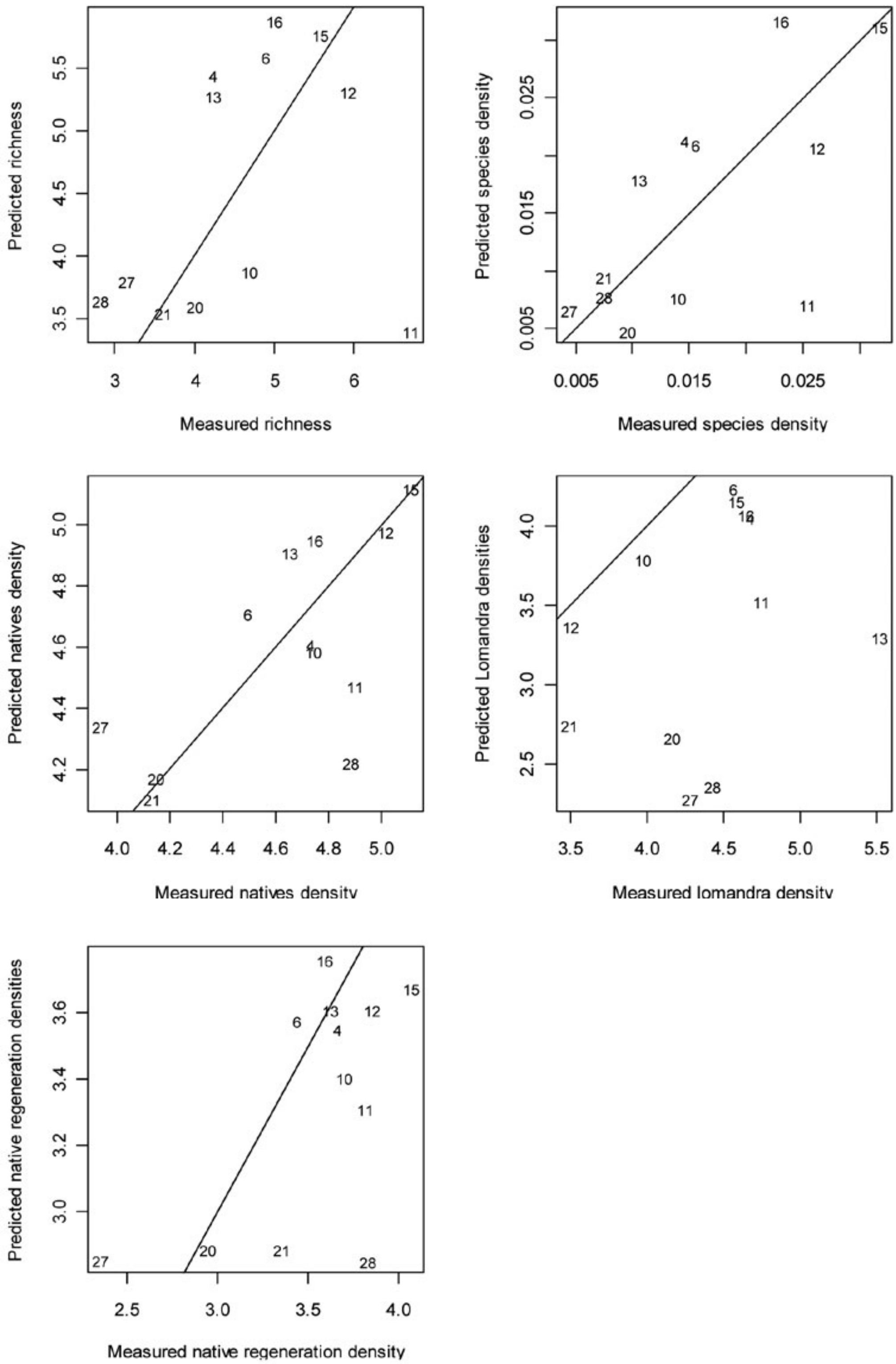


Figure 6.18: Plots of lowess smoothers showing relationships between the effect of flow regulation on selected riparian metric and the Gower metric (an indicator of flow regime change from natural). Two sites were surveyed downstream of each dam and each dam is represented by a pair of points.

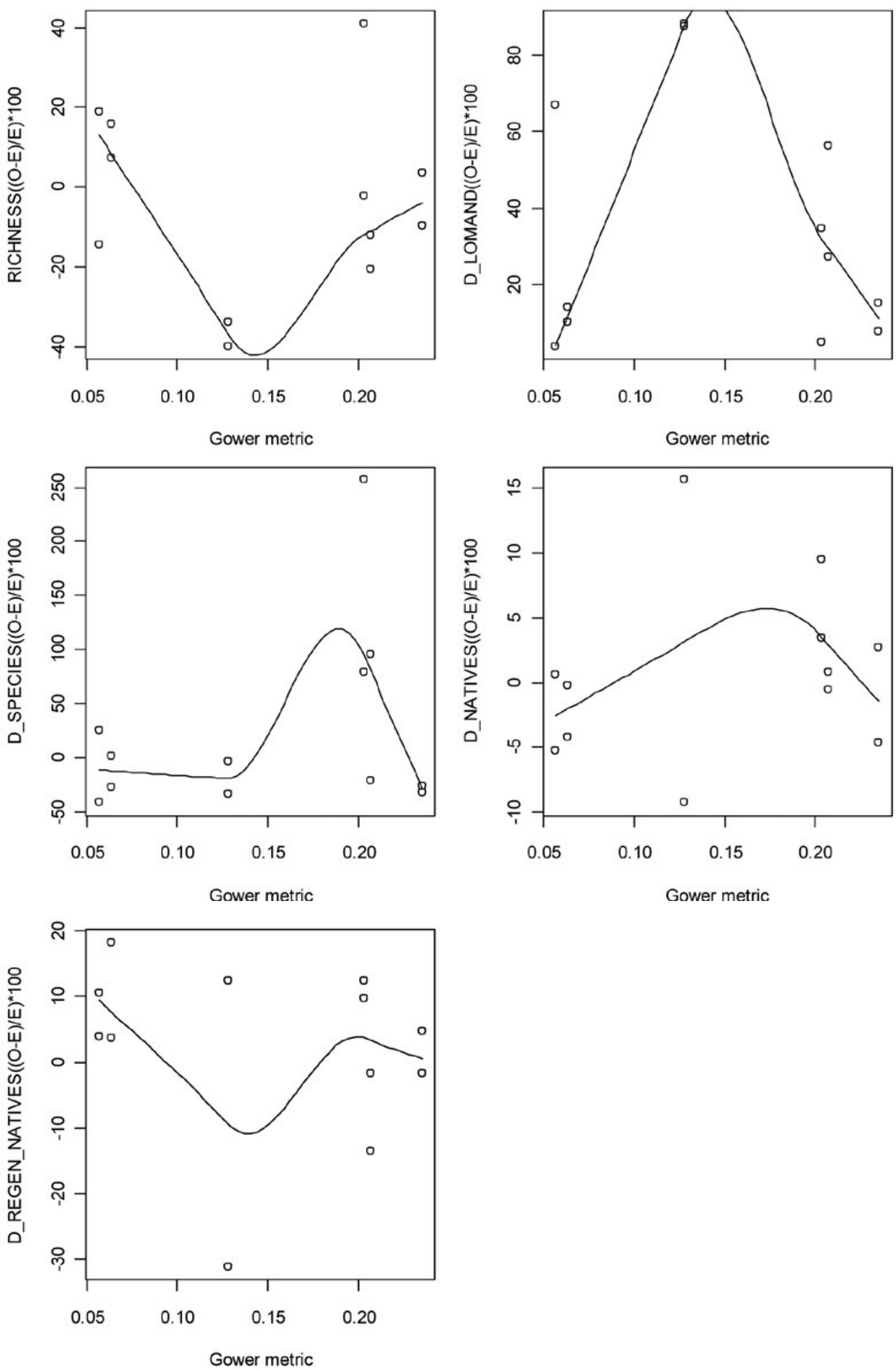


Table 6.18: The effect of flow regulation (Effect) calculated as ((Observed-Predicted)/Predicted)*100 based on PLS models for selected bankfull riparian metrics

Species richness (RICH) (square root transformed), species density (S_DENSITY) (log10(x)+1 transformed), density of natives (D_NATIVES) (log10(x)+1 transformed), density of reeds, rushes and sedges (D_LOMAND) (log10(x)+1 transformed) and the density of native tree regeneration (D_REGEN_NATIVE) (log10(x)+1 transformed).

Metric	Effect	Effect detected?
RICH	-3.99 ± 12.99	No
D_SPECIES	22.88 ± 48.67	No
D_NATIVE	0.68 ± 3.81	No
D_LOMAND	34.70 ± 18.03	Yes
D_REGEN_NATIVE	2.21 ± 7.63	No

Using the Reference models to predict the metric values for regulation sites we only found a significant effect of flow regulation for the density of reeds, rushes and sedges (D_LOMAND). Densities were found to be higher in all regulated/supplemented sites than predicted from the unregulated/unsupplemented sites (Figure 6.17 and Table 6.18).

Species density (D_SPECIES: richness standardised by sampling area) was significantly lower in strongly regulated/supplemented sites (so excluding sites on Obi Obi and Six Mile Creek), but an effect of flow regulation could not be detected from the PLS regression models as the 95% confidence interval included zero (Table 6.18). No effects of regulation could be detected for species richness (RICH), the density of natives (D_NATIVES) or the density of native regeneration (D_REGEN_NATIVE).

Figures showing relationships between the effect of flow regulation versus the Gower metric (an indicator of flow regime change from natural) (Figure 6.18) did not support the hypothesis that increasing flow regime change will result in increasing biotic change.

Hypothesis 4 summary

- Results of the analyses provided mixed support for hypothesis 4.
- The hypothesis that for a given RFC a regulated/supplemented site should be more DISSIMILAR to the unregulated/unsupplemented sites, and conversely a regulated/supplemented site should be SIMILAR to the unregulated/unsupplemented sites within HFCs, was only supported for RFC 5. Sites on the Nerang River (4 and 6) were significantly different in their tree and shrub assemblage structure to the unregulated/unsupplemented sites. Furthermore, significant differences were not found between Nerang sites (4 and 6) and unregulated/unsupplemented sites within its HFC 3.
- Differences between regulated/supplemented and unregulated/unsupplemented site near-stream assemblages, however, were not detected for any of the HFCs or RFCs.
- An effect of flow regulation was detected for the metric D_LOMAND (densities of reeds, rushes and sedges). Sites with regulation had higher densities of these species than predicted from unregulated/unsupplemented sites.
- The hypothesis that increasing flow regime change will result in increasing biotic change was not supported, although the number of dams available for testing this relationship (six) was low.

6.5 Discussion

The ELOHA framework is underpinned by several concepts. This report tested several hypotheses related to these concepts as a means of validating the framework for riparian vegetation.

Hypothesis 1: *The structure and composition of riparian assemblages in the SEQ region will be influenced by stream flow.*

The ELOHA method is underpinned by the concept that flow is a key determinate of the ecological community (Arthington et al. 2006). Whilst there is significant evidence to suggest links between stream flows and riparian vegetation for other regions of Australia and internationally, this link has not been made for riparian vegetation of SEQ. We found limited evidence to support the hypothesis that riparian vegetation assemblages of streams in SEQ are influenced by stream flows. Under this hypothesis a number of sub-hypothesis were proposed and these are dealt with individually below.

Flood and high flow disturbance are a major control on the composition and structure of riparian vegetation.

Evidence was found to support this sub-hypothesis. A number of high flow and flood metrics were significantly correlated with the bankfull vegetation NMDS ordinations. However there were some anomalies with this observation as bankfull shear stress (BFShear) and bankfull discharge (BFDis) only correlated significantly with the near-stream vegetation nMDS ordination. CCA and partial CCA analysis identified only one high flow and flood related metric (Bankfull duration; BFDur) as significant in structuring riparian vegetation and again this variable was only significant for the bankfull vegetation.

Bankfull variables were relatively important in regression random forest models of riparian vegetation metrics with bankfull shear stress (BFShear) identified as an important variable in a number of the riparian metric models for both bankfull and near-stream vegetation. Regressions of BFShear and BFDis suggested significant relationships with metrics describing riparian diversity (RICH and D_SPECIES) and the density of intermediate successional species (D_INTER).

Baseflows, low flows and average flows are a major control on the composition and structure of riparian vegetation.

Limited evidence was found to support this sub-hypothesis. Measures of average flow conditions, median annual flow (MEDAnnual), mean wet season flow (MDFWet) and mean dry season flow (MDFDry) correlated with the near-stream vegetation ordination but were not correlated with the bankfull vegetation nMDS ordinations. Only a limited suite of low flow andbaseflow variables (LSDur, BFI and MDBF) were significantly correlated with the nMDS ordinations. However, the CCA and partial CCA analyses identified both low spell number (LSnum) and low spell duration (LSDur) as significant variables.

Base flows and low flows were relatively unimportant in regression random forests models of riparian vegetation metrics. Variables describing average flow conditions (particularly Med and MEDAnnual), however, were identified as important variables in the random forests models of bankfull riparian vegetation metrics.

Variability in stream flows will drive variability in riparian vegetation.

Evidence was found to support this hypothesis. The CVDry flow was the singularly most important hydrology variable identified across both analyses of riparian vegetation assemblage data and metrics. CVDry was strongly correlated with both nMDS ordinations and was a significant variable identified from the CCA analyses of both bankfull and near-stream riparian vegetation communities. This metric was also identified as an important variable in 6 out of the 11 random forests models for selected bankfull riparian vegetation metrics. GLS regression analyses suggested a negative linear relationship between a number of riparian vegetation metrics and CVDry.

It is suggested here that variation in dry season flows may be critical as riparian vegetation may be more reliant on stream flows during the dry season when rainfall is lower. Furthermore, variation in flows at this period of the year may result in frequent spells during which the streams cease to flow, resulting in a dropping of the local riparian groundwater table.

Hypothesis 2: *The structure and composition of riparian assemblages in the SEQ region will be influenced by interactions between flow variables, and other natural factors and anthropogenic disturbances.*

Interactions between flow variables, and other natural factors and anthropogenic disturbances appeared to be relatively unimportant in structuring riparian vegetation assemblages. Partial CCA suggested a relatively low component of variation shared between hydrology and either other environmental influences or land use. Other environmental influences independently explained the highest proportion of variation in both the bankfull and near stream vegetation. Climatic gradients (in particular A_RAIN, A_TEMP and CMA_TEMP) were strongly associated with the nMDS ordinations of both bankfull and near-stream vegetation communities. Furthermore CCA analyses suggested the coldest monthly mean temperature (CMA_TEMP) was important in influencing both near-stream and bankfull vegetation assemblages.

Local and catchment topography appeared to be relatively unimportant in structuring riparian vegetation communities. Few topographic variables correlated with the nMDS ordinations. However, catchment relief ratio was significantly correlated with both the nMDS ordinations of bankfull and near-stream vegetation and, critically basin shape influences stream hydrology and the shape of a stream hydrograph (Gordon et al. 2005). Hence, relationships between catchment relief and vegetation structure may suggest an underlying link between riparian vegetation and stream hydrology

Substrate type appeared to have some importance in structuring riparian communities with the proportion of unconsolidated catchment (UNC_CATCH) strongly correlated with the nMDS ordination space. Furthermore, the proportion of FELSIC igneous geology within the catchment was identified as a significant variable in the CCA and partial CCA analyses of both the bankfull and near-stream vegetation assemblages.

Regression random forests models of individual riparian vegetation metrics support the above findings. Climate variables (A_TEMP, CMA_TEMP, HMA_TEMP and A_RAIN) were in the top ten important variables for 7 out of the 11 models.

Land use independently explained only a small proportion of the variation in riparian assemblage structure (5–7%). However, the proportion of land use under dryland agriculture and plantations (PDA) was the most important variable in the random forests models for the riparian metrics species richness (RICH) and species density (D_SPECIES).

Hypothesis 3: *Riparian assemblage structure in streams of SEQ will differ across the IQQM and gauge hydrological flow classes.*

There was mixed evidence to support this hypothesis. Analyses across flow classes suggest that riparian assemblage structure does vary significantly across both the RFCs and HFCs. Both the univariate and multivariate data revealed differences across the classes. Assemblage data analyses suggest the flow classes are associated with different riparian assemblages, although differences tended to be more distinct across the HFCs with a greater number of significant pairwise comparisons between HFCs than RFCs. Random forests models, however, indicated that tree and shrub composition data were generally poor indicators of flow class. Furthermore, near-stream tree and shrub assemblages were worse predictors of flow class membership relative to the bankfull vegetation tree and shrub assemblages.

Given the importance of many landscape factors (climate, geology, topography and soils) in controlling the hydrological regimes, this result is not surprising. These analyses do not reveal whether the flow regimes are direct controls on the distribution, abundance and diversity of riparian vegetation or whether they are simply correlates for other overarching environmental landscape factors that also differ across the flow groups. In particular, climate variables such as the coldest monthly average temperature (CMA_TEMP) and annual rainfall (A_RAIN) both differed significantly across classes and were also identified as important variables in the ordinations and random forests regression analyses.

Hypothesis 4: *Changes in stream flow regimes will alter the distribution, abundance and diversity of plants on stream and river banks.*

A review of the evidence for the impacts of flow regime alteration on riparian vegetation suggested a number of potential effects (Appendix 1). These are discussed below:

Changes to near-stream vegetation density where flow regimes are altered:

Some evidence was found to support the hypothesis that there have been changes in near-stream vegetation densities. An effect of flow regulation was detected for the bankfull riparian vegetation metric D_LOMAND. Whilst this metric is calculated cross the entire transect (and hence not only near-stream – the near-stream metric could not be modelled adequately), these vegetation types tend to be found in greatest abundances near stream edges.

There is substantial national and international evidence to suggest that a reduction in high in-channel flows and flood flow components will result in the encroachment of vegetation into the main channel itself and a subsequent reduction in active channel width. The Mary River Water Resource Plan (Brizga et al. 2004) suggests that riparian vegetation thickening may have taken place downstream of Cedar Pocket Dam on Deep Creek, while encroachment of riparian vegetation into the main channel has been reported on the Nerang River below Hinze Dam (Brizga et al. 2006b), on Reynolds Creek below Moogerah Dam (Brizga et al. 2006c) and on the Brisbane River below Wivenhoe Dam (McCosker 2000).

Our analyses suggest that these large herbaceous vegetation groups are denser in regulated/supplemented streams. However, the Reference model was relatively poor with a low R^2 so this result should be treated with some caution.

Reduction in the regeneration of native species in the bankfull channel where high flows and flood disturbance are reduced:

No evidence was found to support the notion that native regeneration has been reduced in regulated/supplemented sites. No differences in vegetation regeneration densities could be found overall between regulated/supplemented and unregulated/unsupplemented sites. No effect of regulation could be detected from PLS modelling and predicted values tended to be lower than measured values contrary to the original hypothesis.

A reduction in the proportion of species characteristic of early successional stages where flood disturbance is reduced:

No evidence was found to support the hypothesis that the proportion of species characteristic of early successional stages is reduced where flood disturbances have been reduced. No differences could be found overall between regulated/supplemented and unregulated/unsupplemented sites.

Reductions in species diversity where flow variability is reduced:

Evidence was mixed regarding the hypothesis that riparian vegetation species diversity is reduced in regulated/supplemented sites. Overall, species diversity (measured as species richness per unit area) was significantly lower in strongly regulated/supplemented sites compared to unregulated/unsupplemented sites.

However, PLS regression models did not reveal an effect of regulation. Analysis of various riparian vegetation metrics against the coefficient of variation in the dry season flows using GLS regression suggested that highest species richness was actually associated with lowest values for CVDry rather than high values of CVDry.

Increased proportion of exotic species with flow regime change:

We could not find any evidence to support the hypothesis that there is an increase in the proportion of exotic species with flow regime change. Overall, exotic species densities were not significantly different between regulated/supplemented and unregulated/unsupplemented sites. PLS models predicting the densities and proportions of exotic species were poor and predicted models could not be constructed for these metrics.

6.6 Implications

Application of the ELOHA framework to riparian vegetation in SEQ has shown that key concepts of the framework (i.e. streams with different flow regimes will have different riparian floras and increasing flow regime divergence from natural will be associated with increasing biotic change) were not fully substantiated.

Hydrologic metrics were considered to be relatively good predictors of riparian assemblage structure. Of the hydrological metrics studies here, the coefficient of variation in dry season (CVDry) (May to October) was particularly important both to the riparian assemblage structure and a number of riparian metrics. It is suggested here that variation in dry season flows may be particularly critical as riparian vegetation may be more reliant on stream flows during the dry season when rainfall is lower.

However, if flow was a primary driver of vegetation patterns one would expect the importance of flow to be greater (relative to other environmental and land use influences) for vegetation situated nearer the stream edge, as these vegetation assemblages are more likely to be impacted by regular high flow and flood conditions and have greater contact with the riparian water table.

Our results indicate that this was not the case as relationships between various hydrological variables and both riparian assemblage structure and metrics for near-stream vegetation, were generally weaker relative to the bankfull relationships. Hence the results should be taken with some caution as the significant hydrology variables may be simply correlates for other overarching drivers of vegetation structure.

Riparian vegetation composition and riparian vegetation metrics varied significantly across HFCs and RFCs. However, it is unknown to what extent flow regime itself is the primary driver of these patterns. Patterns in vegetation metrics across flow classes may be responding to gradients in other environmental factors (particularly climate) that also occur across the flow classes.

The analyses suggest that vegetation at some regulated/supplemented sites (i.e. sites 4 and 6 on Nerang River) has undergone shifts in composition as a result of regulation, as these sites differed significantly in tree and shrub assemblages to unregulated/unsupplemented sites in the same RFC (class 5).

Furthermore, these regulated/supplemented sites were not significantly different from unregulated/unsupplemented sites in the same HFC (class 3). A significant effect of regulation was also found on the density of reeds, rushes and sedges (D_LOMAND) with densities of these species significantly higher in regulated/supplemented sites than predicted from the unregulated/unsupplemented sites.

This finding supports a number of reports from the SEQ region suggesting that near-stream vegetation thickening has taken place downstream of some dam structures due to a decrease in flows capable of scouring vegetation. The implications of these findings for stream managers and water planners are fourfold:

- Firstly, regulation may be having an effect on the riparian vegetation of SEQ streams. Given that riparian vegetation are generally long lived relative to the time dams have been in place for most of the region, it is perhaps not surprising that the effects of regulation have been detected for groups of plants with shorter life histories (reeds, rushes and sedges). Hence sufficient time has elapsed since the construction of the dams to allow a response in these vegetation types to be manifested.
- Secondly, variation of stream flows during the dry season (May to October) are likely to be particularly critical to the health and long-term persistence of riparian vegetation and hence particular attention should be focused on the affects of regulation on flow variation in this season.
- Thirdly, for riparian vegetation environmental flows need to be estimated in relation to their hydraulic consequences particularly in terms of flood characteristics, i.e. bankfull shear stress, bankfull discharge and bankfull duration.

- Fourthly, these results do not rule out a significant impact of regulation on other components of the riparian flora. Insufficient time may have elapsed for an effect from regulation to be detected amongst slower growing vegetation types (even at the regeneration stage as tested here).

This study has demonstrated that riparian vegetation assemblages respond to natural landscape gradients (e.g. climate), catchment land use and components of the flow regime. Given the likelihood of further flow regime change by water resource development and climate change, flow regulation impacts need to be incorporated into the monitoring of stream ecosystem health. Riparian vegetation should be used as indicators of flow regime change, as well as being indicators of broader catchment, riparian and channel health.

6.7 Attachments

Attachment 6.1: Species list for sample sites

Successional stages followed the terminology of Kanowski et al. (2010) as Early (E), Intermediate (M) or Late (L) and combinations of these stages where the species occurred in more than one successional stage (i.e. EM, ML or EML).

** indicates exotic species*

Botanical name	Species code	Growth form	Successional stage
Acacia bakeri	Aca_bak	tree	EM
Acacia fimbriata	Aca_fim	tree	EM
Acacia disparrima	Aca_spp	tree	E
Acronychia oblongifolia	Acr_obl	tree	ML
Ailanthus triphysa	Ail_tri	tree	ML
Alangium villosum	Ala_vil	tree	L
Alchornea ilicifolia	Alc_ili	shrub	ML
Alectryon tomentosus	Ale_tom	tree	ML
Alphitonia excelsa	Alp_exc	tree	EM
Alyxia ruscifolia	Aly_rus	shrub	ML
Angophora spp.	Ang_spp	tree	?
Aphananthe philippinensis	Aph_phi	tree	ML
Araucaria cunninghamii	Ara_cun	tree	EML
Archidendron muellerianum	Rrc_mue	tree	ML
Archirhodomyrtus beckleri	Arc_bec	tree	ML
Archonotophoenix spp	Arc_spp	tree	L
*Ardisia crenata**	Ard_cre	shrub	L
Argophyllum nullumense	arg_nul	shrub	?
Arytera distylis	Ary_dis	tree	L
Arytera divaricata	Ary_div	tree	L
Atalaya salicifolia	Ata_sal	tree	?
Atractocarpus chartaceus	Atr_cha	tree	L
Backhousia myrtifolia	Bac_myr	tree	ML
Beilschmiedia obtusifolia	Bei_obt	tree	ML
Brachychiton spp	Bra_spp	tree	ML
Breynia oblongifolia	Bre_obl	shrub	EM
Briedelia exaltata	Bri_exa	tree	ML
Briedelia leichhardtii	Bri_lei	tree	ML
Bursaria incana	Bur_inc	shrub	?
Callistemon salignus	Cal_sal	tree	ML
Melaleuca viminalis (Callistemon viminalis)	Cal_vim	tree	EM
Canarium australasicum	Can_aus	tree	ML
Capparis arborea	Cap_arb	shrub	ML
Carissa ovata	Car_ova	shrub	M
Castanospermum australe	Cas_aus	tree	ML
Casuarina cunninghamiana	Cas_cun	tree	EM
Casuarina littoralis (Allocasuarina littoralis)	Cas_lit	tree	ML
*Celtis sinensis**	Cel_sin	tree	EML
*Cestrum nocturnum**	Ces_noc	shrub	?
*Cinnamomum camphora**	Cin_cam	tree	ML
Cinnamomum oliveri	Cin_oli	tree	L
Citronella moorei	Cit_moo	tree	L
Citrus X taitensis	Cit_X t	shrub	M
Cleistanthus cunninghamii	Cle_cun	tree	ML
Clerodendrum floribundum	Cle_flo	tree	EM
Commersonia bartramia	Com_bar	tree	EM
Cordyline spp	Cor_spp	shrub	ML
Croton acronychioides	Cro_acr	shrub	L
Cryptocarya bidwillii	Cry_bid	tree	L
Cryptocarya glaucescens	Cry_gla	tree	ML
Cryptocarya laevigata	Cry_lae	shrub	L
Cryptocarya macdonaldii	Cry_mac	tree	L
Cryptocarya obovata	Cry_obo	tree	L
Cryptocarya sclerophylla	Cry_scl	tree	L
Cryptocarya triplinervis	Cry_tri	tree	EML
Cupaniopsis anacardioides	Cup_ana	tree	ML
Cupaniopsis newmannii	cup_new	tree	L
Cupaniopsis serrata	Cup_ser	tree	L
Cyclophyllum coprosmoides (Canthium coprosmoides)	cyc_cop	tree	ML
Daphnandra apatela	Dap_apa	tree	L
Daphnandra tenuipes	Dap_ten	tree	M
Diospyros australis	Dio_aus	tree	ML
Diospyros ellipticifolia	Dio_ell	tree	ML
Diospyros fasciculosa	Dio_fas	tree	ML
Diospyros geminata	Dio_gem	tree	ML
Diospyros pentamera	Dio_pen	tree	ML
Diploglottis australis	Dip_aus	tree	EML
Dissiliaria baloghioides	Dis_bal	tree	?
Drypetes deplanchei	Dry_dep	tree	ML
Dysoxylum rufum	Dys_ruf	tree	ML
Elaeocarpus grandis	Ela_gra	tree	EML
Elaeocarpus obovatus	Ela_obo	tree	ML
Elattostachys nervosa	Ela_ner	tree	L
Elattostachys xylocarpa	Ela_xyl	tree	L
Endiandra discolor	End_dis	tree	L
Endiandra globosa	End_glo	tree	L
Endiandra pubens	End_pub	tree	L
Endiandra sieberi	End_sie	tree	ML
Endiandra virens	End_vir	tree	?
Erythrina species 'Croftby'	Ery_spe	tree	?
Eucalyptus spp	Euc_spp	tree	EM
*Eugenia uniflora**	Eug_uni	shrub	EM
Eupomatia bennettii	Eup_ben	shrub	ML
Eupomatia laurina	Eup_lau	tree	ML
Euroschinus falcata	Eur_fal	tree	EM
Ficus coronata	Fic_cor	tree	EM
Ficus fraseri	Fic_fra	tree	EM
Ficus obliqua	Fic_obl	tree	L
Ficus opposita	Fic_opp	tree	?
Ficus racemosa	Fic_rac	tree	?
Ficus virens	Fic_vir	tree	L

Botanical name	Species code	Growth form	Successional stage
Ficus watkinsiana	Fic_wat	tree	L
Flindersia schottiana	Fli_sch	tree	EML
Glochidion ferdinandi	Glo_fer	tree	EM
Grevillea robusta	Gre_rob	tree	M
Guioa semiglauca	Gui_sem	tree	EM
Helicia glabriflora	Hel_gla	tree	ML
Heritiera trifoliolata (Argyrodendron trifoliolatum F.Muell.)	Her_tri	tree	L
Hibiscus heterophyllus	Hib_het	shrub	EM
hicksbeachia pinnatifolia	Hic_pin	tree	L
hodgkinsonia ovatiflora	Hod_ova	tree	ML
Hymenosporum flavum	Hym_fla	tree	EM
Ixora beckleri	Ixo_bec	tree	L
Jacaranda mimosifolia	Jac_mim	tree	EM
Jagera pseudorhus var. pseudorhus	Jag_pse	tree	EM
*Lantana camara**	Lan_cam	shrub	EM
Lepiderema pulchella	lep_pul	tree	L
Leucaena leucocephala	Leu_leu	tree	E
*Ligustrum lucidum**	Lig_luc	tree	ML
*Ligustrum sinense**	Lig_sin	shrub	ML
Linospadix monostachya	Lin_mon	tree	L
Lophostemon spp	Lop_spp	tree	?
Lophostemon confertus	Lop_con	tree	EML
Lophostemon suaveolens	Lop_sua	tree	?
Macadamia tetraphylla	Mac_tet	tree	L
Mallotus claoxyloides	Mal_cla	tree	M
Mallotus discolor	Mal_dis	tree	EM
Mallotus philippensis	Mal_phi	tree	EM
Medicosma cunninghamii	Med_cun	tree	L
Melaleuca spp	Mel_spp	tree	?
Melaleuca bracteata	Mel_bra	tree	?
Melaleuca quinquenervia	Mel_qui	tree	E
Melia azedarach	Mel_aze	tree	M
Micromelum minutum	Mic_min	tree	ML
Mischarytera lautereriana	Mis_lau	tree	L
Mischocarpus australis	Mis_aus	tree	ML
Mischocarpus pyriformis	Mis_pyr	tree	L
Morus spp	Mor_spp	tree	?
Myrsine variabilis (Rapanea variabilis)	Myr_var	shrub	ML
Nandina spp	Nan_spp	shrub	?
Neolitsea dealbata	Neo_dea	tree	ML
Notelaea microcarpa	Not_mic	shrub	M
Notelaea longifolia	Not_lon	tree	M
*Ochna serrulata**	Och_ser	shrub	M
Olea paniculata	Ole_pan	tree	ML
Pararchidendron pruinosum	Par_pru	tree	ML
Pavetta australiensis	Pav_aus	shrub	ML
Phyllanthus microcladus (Sauropus albiflorus subsp. Microcladus)	Phy_mic	shrub	?
Pittosporum multiflorum	Pit_mul	tree	ML
Pittosporum rhombifolium (now named Auranticarpa rhombifolia)	Aur_rho	tree	M
Pilidiostigma rhytidosperma	Pil_rhy	tree	?
Pittosporum undulatum	Pit_und	tree	E
Pleurostylia opposita	Ple_opp	tree	M
Polyscias elegans	Pol_ele	tree	EM
Pouteria queenslandica	Pou_que	tree	L
Planchonella australis (Pouteria australia)	Pla_aus	tree	L
prunus spp	Pru_spp	tree	?
Pseudoweinmannia lachnocarpa	Pse_lac	tree	L
Psychotria daphnoides	Psy_dap	shrub	ML
Psychotria loniceroides	Psy_lon	shrub	ML
Psydrax odorata	Psy_odo	shrub	?
Psychotria spp. 'shute harbour'	Psy_spp	shrub	ML
Quassia spp	Qua_spp	shrub	L
Rhodamnia argentea	Rho_arg	tree	ML
Rhodamnia rubescens	Rho_rub	tree	M
Rhodomyrtus psidioides	Rho_psi	tree	EM
Rhodosphaera rhodanthema	Rho_rho	tree	M
*Ricinus communis**	Ric_com	shrub	?
Rosaceae fruit tree	Ros_fru	tree	?
Sarcopteryx stipata	Sar_sti	tree	ML
*Schefflera actinophylla***	Sch_act	tree	M
*Senna spp.**	Sen_spp	shrub	EM
*Senna pendula**	Sen_pen	shrub	EM
*Senna septemtrionalis**	Sen_sep	shrub	EM
Senna sulfurea	Sen_sul	shrub	EM
Sloanea australis	Slo_aus	tree	L
Sloanea woolsii	Slo_woo	tree	L
*Solanum chrysotrichum**	Sol_chr	shrub	E
*Solanum mauritianum**	Sol_mau	tree	EM
*Solanum torvum**	Sol_tor	shrub	EM
Sterculia quadrifida	Ste_qua	tree	M
Streblus brunonianus	Str_bru	tree	ML
Symplocos spp.	Sym_spp	shrub	ML
Synoum glandulosum	Syn_gla	tree	ML
Syzygium australe	Syz_aus	tree	ML
Syzygium floribundum (Waterhousea floribunda)	Syz_flo	tree	ML
Syzygium luehmannii	Syz_lue	tree	L
Syzygium oleosum	Syz_ole	tree	ML
Syzygium smithii (Acmena smithii)	Syz_smi	tree	ML
Tabernaemontana pandacaqui	Tab_pan	shrub	EML
*Tecoma stans**	Tec_sta	tree	EM
*Tecoma capensis**	Tec_cap	shrub	?
Toechima tenax	Toe_ten	tree	ML
Toona ciliata (Toona australis)	Too_cil	tree	EML
Trema tomentosa	Tre_tom	tree	EM
Tristaniopsis laurina	Tri_lau	tree	ML
Turraea pubescens	Tur_pub	shrub	ML
Vitex melicopea	Vit_mel	shrub	?
Wikstroemia indica	Wik_ind	shrub	EM
Wilkiea macrophylla	Wil_mac	shrub	L
Wilkea huegliana	Wil_hue	shrub	ML

7. Aquatic vegetation

7.1 Introduction

The ELOHA framework is a new approach for assessing environmental flow requirements and establishing regional environmental flow standards (Poff et al. 2010). Hydrologic and ecological information for rivers within the region of interest is used to develop flow alteration – ecological response relationships for rivers with different flow regime types (Poff et al. 2010). If there are consistent flow alteration–ecological response relationships within each distinctive flow regime type, then detailed information is not necessary for all rivers of that type within the region of interest.

The ELOHA framework is underpinned by several concepts. These are:

1. flow regime (i.e. the temporal pattern of river discharge) is a key driver of biotic assemblage structure and ecological function in riverine environments
2. different regional flow regime types will have different ecological characteristics
3. flow regime change will cause changes in assemblage structure and other ecological response variables
4. relationships between flow regime characteristics and biological attributes can be isolated from other potential drivers of assemblage structure such as land use (Poff et al. 2010).

There is extensive evidence to show that the principal facets of the flow regime (i.e. magnitude, timing, frequency, duration and rate of change) are key drivers of vegetation structure in rivers and streams (Appendix 2). The flow regime may influence aquatic vegetation assemblage structure directly through effects on physiology and morphology (Madsen and Søndergaard 1983; Madsen et al. 1993; Biehle et al. 1998; Barrat–Segretain 2001), and indirectly through temporal variations in channel morphology, water velocity, substrate composition and water quality (Bilby 1977; Holmes et al. 1980; Cosser 1989; Rea et al. 2002; Giorgi et al. 2005).

Furthermore, several studies have identified aquatic vegetation assemblages associated with distinct flow regime types (e.g. Westwood et al. 2006a,b), suggesting the ELOHA framework is applicable to aquatic vegetation. Similarly, flow regime alteration arising from the construction and operation of dams and weirs may cause changes to aquatic vegetation, such as changes in biomass and species composition (French and Chambers 1997; Baattrup–Pederson and Riis 1999; Vanderpoorten and Klein 2000). These changes have been linked to changes in water velocity and water level fluctuations associated with flow regime alteration.

The flow regime is known to be just one of several environmental factors that control the distribution and abundance of aquatic vegetation in rivers and streams (Biggs 1996; Carr et al. 1997). Hence the direct influence of the flow regime on aquatic vegetation may be relatively unimportant compared with other environmental factors. These factors may influence aquatic vegetation at 'local' spatial scales (e.g. riparian shading and substrate composition) or at broader spatial scales (such as climate zones and site position in catchment).

Mackay (2007) summarised the key environmental factors that control the distribution and abundance of submerged vegetation in SEQ in a simple conceptual model. This model, based on the work of Biggs (1996) and Riis and Biggs (2001), is a two dimensional habitat template consisting of disturbance and resource availability axes. The disturbance axis is represented by a combination of hydraulic and hydrologic factors and the resource axis is represented by light availability (riparian canopy cover and turbidity) and alkalinity.

This model implies that hydrology (e.g. as the CV of mean daily discharge) will interact with other key environmental factors to control the distribution and species composition of aquatic vegetation assemblages.

In summary, there is evidence to support the applicability of the ELOHA framework and its concepts to aquatic vegetation. However, the ELOHA framework has not been applied to aquatic vegetation to date.

7.1.1 Hypotheses and objectives

This report presents an application of the ELOHA framework to aquatic vegetation in SEQ. Specifically the aims of this report are to:

1. identify flow and other environmental variables influencing the structure of fish assemblages in the study area
2. identify the effects of flow variability and the impact of flow regime alteration on measures or 'indicators' of fish assemblage structure and the abundance of individual species
3. identify thresholds or linear relationships between indicators of the structure of fish assemblages and the abundance of individual species and the overall gradient of flow regime alteration in the study area
4. identify thresholds or linear relationships between indicators of the structure of fish assemblages and the abundance of individual species and gradients of alteration of individual flow metrics

To assess the utility of the ELOHA framework for aquatic vegetation the following hypotheses are tested:

Hypothesis 1: *Streams with similar flow regime characteristics should be more similar in terms of macrophyte assemblage metrics (abundance, species richness, species traits) and assemblage composition than streams with different flow regime characteristics.*

This hypothesis is a statement of a key premise of the ELOHA framework. If this premise holds true in SEQ then differences in aquatic vegetation (species composition, abundance, functional attributes) should be evident between flow classes identified by flow regime classification (Chapter 3).

Tests of this hypothesis will be based on the Historic flow regime classification (i.e. the classification based on stream gauge data) as macrophytes are expected to respond to recent short-term antecedent flow regime characteristics and short-term flow events.

Hypothesis 2: *Aquatic vegetation abundance will vary inversely with discharge magnitude.*

Hypothesis 1 assumes that the flow regime will be an important driver of vegetation assemblage structure. Hypotheses 2–4 specify individual components of the flow regime that may drive vegetation assemblage patterns. Hypothesis 2 is related to Hypothesis 1 in that discharge magnitude (standardised by catchment area) is the primary difference between HFCs. Evidence to support this hypothesis is presented in Appendix 2.

Hypothesis 3: *Macrophyte abundance will vary inversely with flood frequency.*

In New Zealand catchments aquatic macrophytes are limited to rivers with 13 or less high flow disturbances per year (Riis and Biggs 2003). Flood frequency, like discharge magnitude, is a convenient hydrologic metric for examining aquatic vegetation patterns in relation to individual flow metrics. Flood frequency is a relatively easy metric to calculate when compared with hydraulic metrics, which may require channel surveys. However, flood frequency is not easily transferred to other river catchments since the effects of flooding are related to geomorphology and channel form, which may vary between catchments.

Hypothesis 4: *Aquatic vegetation abundance will be positively correlated with discharge variability.*

The coefficient of variation (CV) of mean daily discharge is a component of the disturbance axis of the aquatic vegetation conceptual model (Appendix 2). Figure 2 (Appendix 2) suggests that the relationship between daily discharge variability and aquatic vegetation cover is positive. This is due to the fact that streams with high mean daily discharge variability tend to have conditions amenable to aquatic vegetation growth – long periods of low flow, often relatively low turbidity and relatively fine substrates (see also Riis et al. 2008).

Hypothesis 5: *Aquatic vegetation abundance (as biomass or cover) will be higher in regulated sites than unregulated sites, if flow regulation results in increased discharge stability or reduced frequency of substrate mobilisation.*

Changes in substrate stability may result from flow regulation since flood frequency and magnitude will be reduced downstream of dams. This may result in reduced frequency of substrate mobilisation and hence increased macrophyte abundance. Previous work in SEQ (Section 2.4) has shown that flow regulation may result in increased macrophyte cover downstream of dams, although this response may depend upon the extent of shading by riparian canopy cover.

Hypothesis 6: *Increasing degree of flow alteration from baseline condition will produce increasing degree of change in aquatic vegetation assemblages.*

The ELOHA framework assumes that increasing degree of flow regime change (from Reference condition) is associated with increasing degree of ecological change (Poff et al. 2010). Flow regimes downstream of dams in SEQ have undergone varying degrees of change from baseline (Reference) condition (Chapter 3) hence there is scope to examine changes in aquatic vegetation assemblages over a gradient of flow regime change.

7.2 Methods

7.2.1 Study area, flow classification and site selection

The study was conducted in the coastal river catchments of SEQ. The study area and criteria for site selection are discussed in Chapters 2–5. SEQ has a relatively high density of stream gauges, stream gauge data quality is very good and a variety of flow regime types are known to occur (Kennard et al. 2010a). The region has a sub-tropical climate and diverse topography. Rainfall is summer-dominated but the influence of temperate weather systems may produce significant rainfall during autumn and winter (Pusey et al. 2004).

Selection of sites for aquatic vegetation surveys was based on classification of modelled natural and Historic (gauged) flow regimes in SEQ. Both classifications were derived using model-based clustering and 35 flow metrics (Chapter 3). The natural flow regime for individual sites was modelled using an IQQM. The natural classification is hereafter termed the Reference classification and the flow metrics used to undertake the Reference flow classification are termed the Reference flow metrics. The Historic flow regime is the flow regime actually recorded by a stream gauge.

Thus the Historic flow regime includes flow regime changes through time. Flow metrics calculated from stream gauge data are termed Historic flow metrics and the classification derived from these metrics is termed the Historic flow classification. Six RFCs and five HFCs were identified (Chapter 3). The HFCs varied principally in terms of discharge magnitude, with a secondary gradient of discharge variability and spell frequency/duration also present. Sites for aquatic vegetation surveys were selected from RFCs and HFCs (Table 7.1) according to principles described in Chapter 4.

Final site selection was based on ease of access, the condition of riparian vegetation (see below), the location of tributaries near gauges, and workplace health and safety issues. The Brisbane River was not included in this study since the effects of flow regulation by Wivenhoe Dam were the focus of a recent environmental flow study, the riparian vegetation of the main channel is highly modified and the river downstream of Wivenhoe Dam is difficult to survey due to high-volume water releases from the dam (Arthington et al. 2000).

All ELOHA study sites were as close as possible to a stream gauge operated by the Queensland D. Over half of the sites were within 4 km of a stream gauge and 10 sites were within 1 km or less from a stream gauge (Attachment 7.1).

A pair of sites was chosen in the vicinity of each stream gauge. The sites in each pair were a minimum of 2 km apart. Sites were 40–100 m long and contained a variety of hydraulic units (i.e. riffles, runs, pools) where present. Forty individual sites (20 stream gauges) were selected for aquatic vegetation surveys (Attachment 7.1).

Aquatic vegetation surveys were undertaken from June 2008 to September 2010. Four individual aquatic vegetation surveys were undertaken but the number of sites surveyed at each time varied. The first survey included 28 sites (sites 1–28 in Appendix 7.1) surveyed between June 2008 and January 2009. The second survey included all 40 sites and was conducted between June and October 2009.

The additional sites were included to better represent gradients in individual hydrologic metrics (Chapters 3–5). The third survey (January–March 2010) included 26 sites as 14 sites (i.e. seven gauges) could not be surveyed due to extensive flooding in the region. All 40 sites were surveyed during the fourth survey, completed between June and September 2010. Twenty sites were surveyed at all four sampling times (Attachment 7.1). The total number of site surveys was 132. To avoid confusion the term 'sample' refers to an individual site survey and 'site' refers to the location in the stream network where the sample was taken. Each site therefore has a maximum of four samples.

7.2.2 Aquatic vegetation survey methods

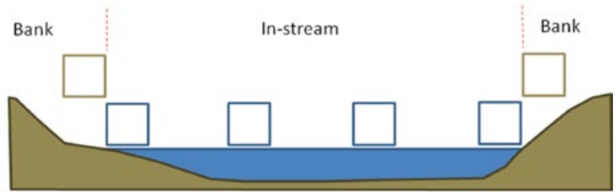
At each site aquatic vegetation was surveyed on five randomly located transects placed perpendicular to the direction of stream flow. Minimum distance between transects was 3 m to minimise disturbance to adjacent transects during sampling. Two vegetation components were surveyed: in-stream vegetation and bank vegetation.

In-stream vegetation was defined as vegetation rooted or occurring within the wetted perimeter of each transect. Bank vegetation was defined as vegetation rooted on the stream bank within 1 m of the stream edge. In-stream vegetation cover was estimated within four 1 m^2 quadrats on each transect (20 quadrats per site). Two of the four quadrats on each transect were placed at the edge of the wetted perimeter at each bank (Figure 7.1) to ensure that emergent vegetation was represented. Emergent vegetation was expected to be the dominant component at each site based on previous experience in the region.

Bank vegetation cover was estimated in two 1 m^2 quadrats on each transect. These quadrats were placed on each bank at the edge of the stream (Figure 7.1). Site cover for in-stream and bank vegetation was also estimated separately for each component. Site cover for bank vegetation (both banks combined) was estimated within a 1 m wide belt-transect extending the length of the site.

Aquatic vegetation was identified using keys (Wilson 1988; Jacobs and Frank 1997; Stephens and Dowling 2002) or by comparison with material submitted to the Queensland Herbarium.

Figure 7.1: Placement of in-stream and bank quadrats for aquatic vegetation surveys.



7.2.3 Environmental parameters

Environmental parameters are summarised in Table 7.1. These parameters describe habitat at three spatial scales – within-site, site-scale and catchment-scale. Within-site parameters were those measured at the quadrat and transect-scale describing fine-scale environmental variation. Site-scale parameters were those that relate to the site as a whole (e.g. water quality and flow metrics). Catchment-scale parameters were those describing site positions in catchment, land use, geology and climate.

Within-site parameters

Within-site parameters mostly describe the hydraulic environment (i.e. the forces exerted on the streambed by flowing water). Quadrat-scale measurements included water velocity, depth, and riparian canopy cover. The average water velocity within each quadrat was measured at 0.6 times the stream depth (Gordon et al. 2005) with a Swoffer model 2100 flow meter. The depth of each quadrat was recorded to the nearest centimetre with a staff. The riparian canopy cover above each quadrat was estimated using a spherical densiometer (Lemmon 1956).

Depth and water velocity measurements were used to calculate Froude number and Reynolds number, descriptors of flow patterns (Gordon et al. 2005). Reynolds Number (Re) is the ratio of inertial forces to viscous forces and describes whether flow is laminar (smooth) or turbulent. It is calculated from the equation $Re = VL/v$ where V is velocity (ms^{-1}), L is length (m) and v is kinematic viscosity (m^2s^{-1}). Mean depth was used as the length measure for calculating Reynolds number (Gordon et al. 2005).

Froude number (*Fr*) is the ratio of inertial to gravitational forces and is a useful measure of bulk flow characteristics. Froude number was calculated from the formula $Fr = V/\sqrt{(gD)}$ where V is mean velocity (ms^{-1}), g is acceleration due to gravity (ms^{-2}) and D is hydraulic depth (m).

Table 7.1: Environmental variables

See text for description of variables.

Spatial Scale	Parameter	Unit	Acronym
Catchment and land use	Latitude	Degrees	DECLAT
	Longitude	Degrees	DECLONG
	Catchment area upstream of site	km^2	CATAREA
	Site elevation	m	ELEV
	Site distance to source	km	DISTS
	Site distance to mouth	km	DISTM
	Bank slope	$m.m^{-1}$	B_SLOPE
	Catchment elongation ratio	No unit	CAT_ELON
	Catchment relief ratio	No unit	CAT_RELI
	Reach valley confinement	%	V_Conf
	% Felsic geology	%	FELSIC
	% Mafic geology	%	MAFIC
	% Sedimentary rock (siliclastic and undifferentiated)	%	SED_SILIC
	% Sedimentary rock (carbonates)	%	SED_CARB
	% Mixed sedimentary and igneous rock	%	MIXED
	% Unconsolidated rock (alluvium, colluviums etc.)	%	UNC_CATCH
	% Unconsolidated material for reach	%	UNC_REACH
	Production from relatively natural environments	%	PNE
	Production from dryland agriculture and plantations	%	PDA
	Production from irrigated agriculture and plantations	%	PIA
	Conservation and natural environments	%	CAN
	Intensive uses	%	IU
	Annual mean temperature	°C	A_TEMP
	Coldest month mean temperature	°C	COLD_TEMP
	Hottest month mean temperature	°C	HOT_TEMP
	Annual mean rainfall	mm	A_RAINFALL
Site	Discharge required to mobilise D50	$Ml.day^{-1}$	Q_D50
	Percentage of days prior to sampling where discharge was above the threshold required to mobilise the D50	%	FD50MOVE
	Number of days since discharge required to mobilise D50 occurred	No unit	DAYS_Q_D50
	Number of high spells based on discharge required to mobilise D50	No unit	D50_HSNum
	Mean duration of high spells (Q_D50 threshold)	Days	D50_HSDur
	Number of high spells (75th percentile)		HSNum
	M ean duration of high spells (75th percentile)	Days	HSDur
	Number of low spells (25th percentile)		LSNum
	Mean duration of low spells (25th percentile)	Days	LSDur
	Coefficient of variation of mean daily discharge	%	CVDaily
	Reference flow class	No unit	REF_CLASS
	Historic flow class	No unit	HIS_CLASS
	Median particle size	cm	D50
	Shear stress	Nm^2	SHEAR
	Critical shear stress	Nm^2	CRITSHEAR
	Substrate stability	No unit	SUBSTAB
	Bankfull shear stress	Nm^{-2}	BFSHEAR
	Bankfull substrate stability	No unit	BFSUBSTAB
	pH		PH
	Turbidity	NTU	TURB
	Bankfull depth	M	BFDEPTH
	Bankfull width	M	BFWIDTH
	Ratio of bankfull width:bankfull depth	No unit	BFWIDTH_ BFDEP
	Bedslope	$m.m^{-1}$	BEDSLOPE
	Waterslope	$m.m^{-1}$	WATERSLOPE
Within-site	Width	m	WIDTH
	Depth	m	DEPTH
	Water velocity	ms^{-1}	VELOC
	Reynolds number	No unit	REYNOLD
	Froude number	No unit	FROUDE
	Riparian canopy cover	%	RipCov

Transect-scale measurements included wetted width, median particle size and shear stress. The wetted width of each transect was measured to the nearest 0.1 m with a tape. The median particle size (D50) was used in calculations of substrate stability (see below). The median particle size was estimated by measuring the *b*-axis of 100 randomly selected substrate particles (>0.5 cm diameter) with a Vernier caliper (Wolman 1954; Gordon et al. 2005).

Where particle size was less than 0.5 cm, a PVC cylinder (internal diameter 8.6 cm, sampling area 58 cm^2) was used to collect a sediment sample to a depth of 5–10 cm from each quadrat (on the transect) for dry sieve analysis. Substrate samples for dry sieve analysis were stored in clipseal plastic bags and stored frozen. For dry sieving, samples from individual transects were thawed, thoroughly homogenised and allowed to air dry. Soil clods were broken up as much as possible as the soil dried.

The moisture content of soil samples from each transect was determined by drying approximately 60 g of soil in an aluminium crucible at 105oC for 24 hours. To determine soil particle size composition approximately 1 kg of air-dried soil was agitated for 15 minutes through a sieve stack comprising 2 mm, 710 µm, 600 µm, 425 µm, 300 µm, 150 µm and 75 µm brass sieves.

The soil retained on each sieve was weighed to the nearest milligram. A cumulative frequency distribution was plotted for each site to determine the 16th, 50th and 84th percentiles of the particle size distribution. Median particle size estimates were used to calculate site-scale estimates of substrate stability.

Shear stress (τ, Nm^{-2}), the frictional force exerted per unit area on the stream bed was calculated from the equation $\tau = \rho g R S$ where ρ is the density of water (1000 kgm^{-3}), g is acceleration due to gravity (9.8 ms^{-2}), is hydraulic radius (the ratio of cross-sectional area to wetted perimeter) and S is slope of the energy line, approximated by water slope (Gordon et al. 2005). Shear stress at bankfull discharge was calculated using the same equation, with bankfull hydraulic radius substituted for hydraulic radius and bed slope used as an approximation of the water slope (Marsh et al. 2004).

Substrate stability was estimated using equations in Grayson et al. (1996). The critical shear stress (τ_c) required to initiate movement of streambed particles was calculated from the equation $\tau_c = \theta_c gd(\rho_s - \rho)$ where θ_c is dimensionless critical shear stress, *d* is substrate size, ρ_s is sediment density (approximated as 2650 kgm^{-3}) and ρ is the density of water (1000 kgm^{-3}). θ_c was estimated from a Shields curve (θ_c *versus* Roughness Reynolds number) as described by Grayson et al. (1996).

The critical shear stress was calculated using the median value for particle size. The ratio of critical shear stress (calculated for the median particle size) to bankfull shear stress was calculated to determine whether the median particle size (i.e. half of the stream bed) would be mobilised at bankfull discharge.

Ratios greater than one indicates that the median particle size would not be mobilised at bankfull discharge and a ratio less than 1 indicates mobilisation of the median particle size at bankfull discharge. As stage discharge relationships were not developed for individual sites τ_c could not be compared with shear stresses calculated for different discharges. The bankfull shear stress was estimated using data from cross-sectional channel surveys and WINXSPRO version 3.0.

Site-scale parameters

Site-scale parameters included bed slope, water slope, water quality and hydrologic metrics.

Bed slope was measured with a dumpy and staff as the fall in height over three riffles located in the same reach as the site (Roy et al. 2003; Gordon et al. 2005). Water slope over the length of the site was also measured with a dumpy and staff.

The influence of hydrology on aquatic vegetation in the study area was examined by selecting a small suite of hydrologic metrics as potential correlates of vegetation structure. The metrics chosen were based on hypotheses drawn from published information and the aquatic vegetation conceptual model (see literature review, Appendix 2). These metrics described flood frequency, spell frequency and duration, time between spells, and discharge variability. Hydrologic metrics were calculated using the River Analysis Package (Marsh et al. 2003).

Water quality parameters included conductivity, pH, turbidity and water temperature. These parameters were measured *in-situ* using Greenspan sensors. These readings were taken at approximately noon to standardise water temperature measurements between sites. Three measurements of each parameter were taken at each site.

Catchment, geology and land use parameters

Calculation of catchment, geomorphology and land use parameters is presented in Chapter 5.

Cross sectional surveys were undertaken at each site to determine channel morphology and used in the calculation of various stream hydraulic parameters (Table 7.1). Surveys were conducted with a dumpy and staff. One cross section was surveyed per site, but additional surveys were conducted in heterogeneous channels. Cross sections were located in riffles (Harrelson et al. 1994) and were marked with a 10 cm bolt and quick dry cement at a point above bankfull discharge (Harrelson et al. 1994). Channel orientation was measured with a compass in the field.

Site topography variables (distance to source, distance to mouth and elevation) were determined for each site using GIS with detailed stream networks based on 1:25 000 and 1:100 000 scale drainage maps and 30 m DEM. Catchment area was computed using geometry functions of the GIS software.

Drainage basin shape was represented by the elongation ratio (Re; the diameter of a circle with the same area as that of the basin divided by the length of the basin) which is considered to have a reasonable correlation with stream hydrology (Morisawa 1958). Reach morphology variables, stream gradient (elevation difference for each reach divided by its length) and valley confinement (percentage of stream grid cells and their immediate neighbours that are not defined as valley bottoms) were acquired from Stein et al. (2009).

Reach mean annual temperature, reach hottest month mean temperature and reach coldest month mean temperature variables were acquired from Stein et al. (2009) for individual reaches on which sites were situated. Mean annual rainfall was supplied by the Bureau for Meteorology. Rainfall values were based on a rainfall grid generated using the ANU 3-D spline.

Substrate geological characteristics were derived for the field sites from the SEQ Region Geoscience dataset (Queensland DNRM 2002) and digital 1:100 000 scale geology maps for the region. Geological groupings were based on broad composition characteristics and followed the classes of Stein et al. (2009).

Landscape-scale land use and disturbance for field site catchments were assessed using the QLUMP dataset (Witte et al. 2006). The primary land use classes were based on the Australian Land Use and Management Classification version 6 (BRS 2002) as these represent broad land use categories differentiating conservation and relatively natural land uses from intensive land uses.

7.2.4 Data analysis

Environmental variation

Environmental variation across the study area was characterised to determine whether individual HFCs had unique environmental characteristics that could influence interpretation of the role of flow on aquatic vegetation patterns. Environmental data were checked for outliers prior to analysis. Unless otherwise specified, environmental variables were range standardised prior to analysis.

Environmental patterns across the study were visualised using nMDS as implemented by the *metaMDS* function in the vegan package for R (Oksanen et al. 2010; R Development Core Team 2010). The association matrix was generated using Euclidean distance.

Fifty random starts were used to find the configuration that minimised stress (i.e. goodness of fit between observed dissimilarities and ordination distances). The *metaMDS* function rotates the best solution (rotation to principal components) so that maximum variation is displayed on the first ordination axis (Oksanen et al. 2010). The *envfit* function was used to fit environmental vectors to the ordination space. This function finds the maximum correlation between intrinsic variables and the ordination space. Significance was assessed using a randomisation procedure and 999 permutations.

Environmental differences between HFCs were examined using ANOSIM (Primer 5, Clarke and Gorley 2001). ANOSIM is analogous to a non-parametric one-way ANOVA except that comparisons are based on ranked similarities. ANOSIM is sensitive to variation in spread or variability between groups (Anderson 2001). Homogeneity of group variances was therefore checked using the *betadisper* function in the vegan package for R, with Euclidean distance as the distance measure. This method is comparable to the Levene test for homogeneity of variances for univariate data (Oksanen et al. 2010).

Finally, a Kruskal–Wallis test was used to test the hypothesis that channel morphology did not vary significantly between HFCs. The ELOHA framework acknowledges that geomorphic differences within flow regime classes may influence biotic patterns (Poff et al. 2010). The ratio of bankfull width:bankfull depth (BFWID_BFDEP) was used as a measure of channel morphology.

Aquatic vegetation metrics

Aquatic vegetation survey data were used to calculate metrics that could be applied and understood by stream managers and were expected to respond to flow regime alteration (Appendix 2; Table 7.2). These metrics were total richness, total in-stream cover, the number of native and alien taxa (i.e. Status) recorded at each site, and functional group. The scheme of Brock and Casanova (1997) was used to assign species to one of six functional groups (Table 7.2).

For species not assigned a functional group by Brock and Casanova (1997) personal knowledge of habitat use and information from Romanowski (1998) and the National Herbarium of New South Wales (http://plantnet.rbgsyd.nsw.gov.au) was used to determine the appropriate functional group. Status was determined from Bostock and Holland (2010). Total richness and cover metrics were also calculated for emergent, submerged and floating/attached vegetation separately.

Emergent vegetation has photosynthetic parts above the water surface. Submerged vegetation has photosynthetic parts below the water surface. Floating/attached vegetation has floating leaves but is attached to the stream bed. Free floating taxa were not considered since they are often a minor component of the in-stream flora in the region (Mackay 2007). All richness metrics (total richness, Status, functional groups) were standardised by site area, which was calculated from measurements of transect width and site length.

For data analysis several taxa were grouped. Bryophyte identifications from the Queensland Herbarium were not available at the time of analysis, so mosses were grouped as bryophytes (Bryo). *Hydrocotyle* spp. were grouped as 'Hydrocotyle'. *Lomandra* spp. and *Carex* spp. were not identified to species in the field and so were grouped as 'Lomandra' and 'Carex' respectively (Attachment 7.2).

Aquatic vegetation metrics were calculated for site-scale and in-stream vegetation data only.

Aquatic vegetation patterns across Historic flow classes

Aquatic vegetation data were checked for outliers prior to analysis. Outliers were not necessarily removed for statistical analysis but were checked to determine whether the outlier was a data entry error or represented natural variation.

Aquatic vegetation patterns were generally compared across the HFCs and not the RFCs. It was assumed that aquatic vegetation would have responded to altered flow regimes given the length of time of flow regulation in the study area (most recent dam constructed was Hinze Dam in 1989, Table 3.2) and the relatively rapid turnover of plant populations.

Two separate vegetation datasets were analysed – an assemblage dataset and a vegetation metric dataset. The assemblage dataset was a site × species matrix (i.e. cover estimates for each taxon present at each site and time). The metric dataset included the value of each vegetation metric (Table 7.2) for each sample. The assemblage dataset was analysed using multivariate techniques and the metric dataset was analysed using univariate techniques.

Table 7.2: Aquatic vegetation metrics

Density metrics are richness metrics standardised by site area. SUB, ATE, ATI, ARP, ARF, TDA and TDR are functional groups described by Brock and Casanova (1997). See Attachment 7.2 for allocation of taxa to functional groups.

Type	Metric	Definition
Richness and density	TOTRICH	Species richness
	TOTDENS	Species density (TOTRICH standardised by site area)
	EMRICH	Number of emergent taxa
	EMDENS	Density of emergent taxa (EMRICH standardised by site area)
	FARICH	Number of floating/attached taxa
	FADENS	Density of floating/attached taxa (FARICH standardised by site area)
Cover	TOTCOV	% In-stream vegetation cover
	SUBCOV	% Submerged vegetation cover
	FACOV	% Attached floating vegetation cover
	EMCOV	% Emergent vegetation cover
Status	NATIVE	Number of native taxa
	NATIVEDENS	Density of native taxa
	ALIEN	Number of alien taxa
	ALIENDENS	Density of alien taxa
Functional Group	SUB	Number of submerged taxa
	SUBDENS	Density of SUB taxa
	ATE	Number of Amphibious Fluctuation-Tolerators (emergent)
	ATEDENS	Density of ATE taxa
	ATL	Number of Amphibious Fluctuation-Tolerators (low-growing)
	ATLDENS	Density of ATL taxa
	ARP	Number of Amphibious Fluctuation-Responders (morphologically plastic)
	ARPDENS	Density of ARP taxa
	ARF	Number of Amphibious Fluctuation-Responders (floating/stranded)
	ARFDENS	Density of ARF taxa
	TDA	Number of Terrestrial taxa associated with damp places
	TDADENS	Density of TDA taxa
	TDR	Number of Terrestrial taxa associated with dry places
	TDRDENS	Density of TDR

Assemblagescale vegetation patterns across the study area were first investigated using nMDS as described above for environmental data. Individual ordinations were undertaken for species cover data and presence–absence data. Presence–absence data and species cover data may produce different interpretations of assemblage-scale patterns (Cushman and McGarigal 2004).

The Bray–Curtis dissimilarity measure was used to calculate the distance matrix (Faith et al. 1987). Species cover ordinations were undertaken using raw, log-transformed ($\log_{10}x+1$ for $x>0$, i.e. zero cover values were not transformed, see Oksanen et al. 2010), and square root-transformed species cover data to compare the effects of data transformation on ordination stress. Samples without aquatic vegetation were omitted from ordination.

Differences in assemblage composition between HFCs were tested using ANOSIM, as described above for environmental data. This analysis tested the hypothesis that assemblage structure (i.e. species presence–absence and species cover) would vary between HFCs (see Hypothesis 1 in *Introduction*). Bray–Curtis dissimilarity was used to calculate the distance matrix. Similar analyses were completed for vegetation metrics (Table 7.2) using Kruskal–Wallis tests. Species cover data were $\log_{10}x+1$ transformed for $x > 0$.

Next, classification random forests (Breiman 2001) were used to determine how well samples could be allocated to the correct HFC using assemblage-scale aquatic vegetation data as criteria for sample allocation to HFCs (Cutler et al. 2007). A random forest is a decision tree method similar to classification and regression trees (CART), except that a 'forest' of trees is generated, compared with a single tree in CART methods.

Random forests models cannot be overfit (contain too many predictor variables) if the number of trees created is large (Breiman 2001) and measures of variable importance can be obtained based on misclassification rates. Each tree is grown with a bootstrapped subsample of cases representing approximately two thirds of the original dataset. The remaining one third of the dataset is the OOB sample (Cutler et al. 2007).

A random subsample of predictor variables is used to determine splitting at each node. Each tree is used to predict the OOB observation (Cutler et al. 2007). For classification, the predicted class of each OOB observation is the class with the highest number of votes. Each tree votes for a class and the class with the most votes is the predicted class for that observation. Functions within the randomForest package for R (Liaw and Weiner 2009) were used to construct random forests. The number of species used at each split was determined using the *tuneRF* function. This function constructs random forests and compares the out-of-bag (OOB) error rate when the number of metrics used at each split is varied. One thousand trees were constructed.

Variable importance (a measure of misclassification rate when the values for an individual predictor are permuted) was assessed using mean decrease in prediction accuracy (Liaw and Weiner 2009). Mean decrease in accuracy for individual predictors is assessed using the OOB samples. Values for each predictor variable are randomly permuted for the OOB samples and passed down each tree to get new predictions.

The difference in the misclassification rate between the original and modified OOB sample is averaged over all trees and normalised by the standard error (Cutler et al. 2007; Liaw and Weiner 2009). The confusion matrix was used to assess how well vegetation assemblage data could be used to predict HFC membership of sites.

Influence of hydrology *versus* other environmental factors on aquatic vegetation patterns

Partial CCA was used to determine the variation explained in species composition data by hydrologic metrics and other environmental data. CCA is a constrained ordination technique, where the variation explained in a biotic dataset is constrained by the environmental variables used. In partial CCA, the effects of a co-variable matrix are first removed and the residual variation explained by a second environmental matrix is then determined (Anderson and Gribble 1998).

This method is useful when datasets share an underlying structure and is typically used to determine variance explained by a set of environmental (predictor) variables independent of other confounding factors (Borcard et al. 1992).

Underlying structure shared by the biotic and environmental datasets can result in an overestimation of the interactions between biotic and environmental variables (Borcard et al. 1992). Through a series of partial CCAs the shared structure can be partialled out, leaving the variance explained by the environmental data matrix only (Anderson and Gribble 1998).

Three environmental datasets were used to undertake partial CCA (Table 7.3). The first matrix (HYDRO) included hydrologic metrics only, since the goal of this analysis was to determine the variation in aquatic vegetation structure explained by hydrologic data independently of other environmental parameters.

The second matrix (TEMPVARS) included environmental variables that varied through time (e.g. hydraulic variables, water quality, riparian canopy cover). The third matrix (LANDSCAPE) included environmental variables that were invariant through time or invariant over the duration of the study (e.g. catchment, land use, climate variables). Environmental parameters included in each dataset are summarised in Table 7.3.

Table 7.3: Summary of Canonical Correspondence Analyses performed (after Anderson and Gribble 1998)

Total explained variation is equal to sum of analysis steps (1+7+12), or analysis steps (2+4+12) or analysis steps (3+5+9).

Analysis step	Explanatory dataset	Co-variable dataset(s)
1	TEMPVARS	None
2	HYDRO	None
3	LANDSCAPE	None
4	TEMPVARS	HYDRO
5	TEMPVARS	LANDSCAPE
6	TEMPVARS	HYDRO + LANDSCAPE
7	HYDRO	TEMPVARS
8	HYDRO	LANDSCAPE
9	HYDRO	TEMPVARS + LANDSCAPE
10	LANDSCAPE	TEMPVARS
11	LANDSCAPE	HYDRO
12	LANDSCAPE	TEMPVARS + HYDRO

Variables included in each dataset:

1. *TEMPVARS: Width, Depth, Velocity, D50, FROUDE, REYNOLD, SHEAR, BFSHEAR, SUBSTAB, pH, BF_SUBSTAB, Temp, Conduct, Turb, RipCover.*
2. *HYDRO: CVDaily, HSNum, HSDur, LSNum, LSDur, DAYSFLOOD, FD50MOVE, Q_D50MOVE, D50_HSNum, D50_HSDur, DAYS_Q_D50, HIS_CLASS..*
3. *LANDSCAPE: BFWID_BFDEP, ORIENT, DECLAT, DECLONG, CATAREA, ELEV, DISTS, DISTM, B_SLOPE, PIA, CAN, IU, A_RAINFALL, A_TEMP, HOT_TEMP, COLD_TEMP, CAT_ELON, CAT_RELI, VAL_CONF, VAL_SLOPE, FELSIC, MAFIC, SED_SILI, MIXED, UNC_CATCH, UNC_REACH.*

Partial CCA was conducted on the species cover matrix only using the *cca* function in the vegan package for R (Oksanen et al. 2010). Species cover data were transformed ($\log_{10}x$+1 for x>0) prior to analysis (Jongman et al. 1995). Forward selection was used to select significant variables from each dataset for partial CCA. Starting with an unconstrained model, variables were added to the unconstrained model using Akaike's Information Criterion (AIC) (Oksanen 2010). Variable significance was tested using a Monte Carlo permutation test and 199 permutations.

At each step AIC was evaluated for all possible variable additions and removals (Oksanen 2010). Variance inflation factors (VIFs) were examined to determine whether remaining variables exhibited high collinearity. Variables with high VIFs (>10, Ahmadi-Nedushan et al. 2006) were excluded from further analysis. Following selection of significant variables, partial CCAs were carried out using the procedure described by Borcard et al. (1992) and Anderson and Gribble (1998). The steps in the variance partitioning process are summarised in Table 7.3.

The partial CCA approach examined the relative importance of hydrology versus other factors in explaining patterns in assemblage scale vegetation data. Regression random forests were used to investigate the relative importance of hydrology in explaining patterns in aquatic vegetation metrics (Table 7.2).

The importance of hydrologic metrics *versus* other environmental variables in explaining patterns in vegetation metrics was determined using the decrease in mean square error (as opposed to the decrease in prediction accuracy for classification random forests described above). The number of environmental variables used at each split was determined using the *tuneRF* function. One thousand trees were constructed for each model.

Vegetation patterns across individual flow metric gradients

Relationships between vegetation metrics and selected Historic flow metrics were modelled using GLS regression. The goal of this analysis was to specifically test Hypotheses 2–4 (see *Introduction*). GLS regression allows for correlated errors and unequal variances (Bolker 2008; Pinheiro 2011).

Each relationship was modelled as a univariate model (i.e. a single flow metric was used as a predictor for each vegetation metric tested, after Heikkinen and Mäkipää 2010). A GLS regression was fitted for each relationship and if the model was significant ($p<0.05$), the model fit was plotted. Residuals were examined to look for outliers and violations of model concepts, especially heteroscedastic or non-normal errors.

The modelled vegetation metrics were based on Hypotheses 2–4 (Table 7.4). Hypothesis 2 predicts that aquatic vegetation cover will be inversely proportional to discharge magnitude. Hypothesis 3 predicts that vegetation cover will be inversely proportional to flood frequency.

Vegetation metrics related to cover (TOTCOV, EMCOV and SUBCOV) were used to test this hypothesis, with HSNum, D50_HSNum and FD50MOVE used as measures of flood frequency. Hypothesis 4 predicts that aquatic vegetation cover will be positively related to discharge variability. This hypothesis was tested using TOTCOV, EMCOV and SUBCOV and CVDaily as a measure of discharge variability (Mackay 2007).

Where significant models were identified, log-likelihood ratio tests were used to ensure that the model was an improvement over the null model. The log-likelihood ratio statistic, *D*, was calculated from the equation 2x(log likelihood null model-log likelihood alternate model), where the alternate model is the univariate GLS model. The significance of   was compared against a X^2 distribution with one degree of freedom (Bolker 2008).

GLS models were fitted using the *gls* function in the nlme package for R (Pinheiro 2011).

Table 7.4: Summary of GLS regressions conducted to identify significant relationships between vegetation metrics and flow metrics

Hypotheses tested	Flow metric(s) tested	Vegetation metric(s) tested
Hypothesis 2 – vegetation cover is inversely proportional to discharge magnitude	Q_D50MOVE	TOTCOV
Hypothesis 3 – vegetation cover is inversely related to flood frequency	HSNum, D50_HSNum, FD50MOVE	TOTCOV, EMCOV, SUBCOV
Hypothesis 4 – vegetation cover is positively related to discharge variability	CVDaily	TOTCOV, EMCOV, SUBCOV

Influence of flow regime alteration on aquatic vegetation

Several methods were used to assess the impacts of flow regime alteration by dams on aquatic vegetation. Firstly, analysis of similarities (ANOSIM) was used to compare species presence–absence and species composition between regulated and unregulated sites, as described above.

This analysis provides a test of Hypothesis 5 (see *Introduction*), which predicted that vegetation cover will be higher in regulated sites if substrate stability is increased in regulated sites. The bankfull substrate stability of regulated and unregulated sites was compared using non-parametric analysis of variance (Kruskal–Wallis test).

Next, ANOSIM was used to compare assemblage structure between unregulated and regulated samples within selected RFCs (Table 7.5). This analysis examined, for each selected RFC, whether sites in that RFC that were subsequently regulated by dams diverged in terms of assemblage structure from non-regulated sites in that class.

It is assumed that if the flow regime is a major driver of assemblage structure and altered flows influence aquatic vegetation, then within a given RFC a regulated site should be more dissimilar to the unregulated sites in that RFC. Log-transformed species cover data ($\log_{10}x$ +1 for x>0) were used to calculate Bray–Curtis dissimilarities. Due to the distribution of regulated sites in RFCs this test could only be performed for RFCs 1 and 5 (Table 7.4).

RFC 1 includes sites downstream of Baroon Pocket Dam (Obi Obi Creek) and sites downstream of Six Mile Creek Dam. Six Mile Creek was excluded from this analysis as in-stream vegetation was rarely recorded at the study sites in this creek. RFC 5 included sites downstream of Hinze Dam on the Nerang River.

Table 7.5: Sites used for ANISOM comparing vegetation assemblage structure between regulated and unregulated sites within selected RFCs

RFC	Regulated sites in RFC	RFC sites for comparison
1	Obi Obi Creek downstream of Baroon Pocket Dam (sites 12, 13)	Amamoor Creek (sites 9, 22), Mary River at Moy Pocket (14, 24), Coomera River (5, 25)
5	Nerang River downstream of Hinze Dam (sites 4,6)	Eudlo Creek (sites 18, 19), North Maroochy River (35, 36), Stanley River (1, 26), Currumbin Creek (29, 30)

Partial least squares projection to latent structures (PLS) modelling was used to quantify the effect of flow regulation (if any) on vegetation metrics, using the methods of Englund et al. (1997a,b) and Zhang et al. (1998). PLS modelling is similar to principal components analysis in that many predictor variables can be summarised into a reduced set of latent components (Eriksson et al. 1995). An advantage of PLS modelling is that it can handle correlated predictor variables and also situations where the number of predictor variables greatly exceeds the number of cases (Eriksson et al. 1995).

Reference PLS models were first developed for vegetation metrics using environmental parameters not directly influenced by flow regulation as predictors. These models were based on unregulated sites only (Attachment 7.1).

Predictors were selected based on variable importance from the regression random forests models as described above. Environmental variables were standardised (zero mean, unit variance) prior to analysis (Jansson et al. 2000). Models were cross-validated using 'leave one out' cross-validation (Wehrens and Mevik 2009) and the appropriate number of components for individual models was determined from a plot of the root mean squared error of prediction versus the number of components.

The Reference model was then used to predict values for vegetation metrics at regulated sites. The 'effect' of flow regulation was calculated as [(observed value–predicted value)/predicted value] × 100 (Zhang et al. 1998). The effect of flow regulation was considered to be significant if the mean of the effects did not include zero, corresponding to $p<0.05$ (Zhang et al. 1998). PLS models were fitted using the *pls* package for R (Wehrens and Mevik 2009) and the orthogonal scores algorithm.

Finally, to test the hypothesis that increasing flow regime change causes increased divergence of assemblage structure from Reference condition (Hypothesis 6), the 'effects' from the PLS models were correlated with the Gower metric values obtained for the comparison of Reference and Historic flow regimes for individual stream gauges (Chapter 3).

If increasing flow regime change is associated with increasing divergence of biota from Reference condition then the flow regulation effect for individual sites should be correlated with the Gower metric.

7.3 Results

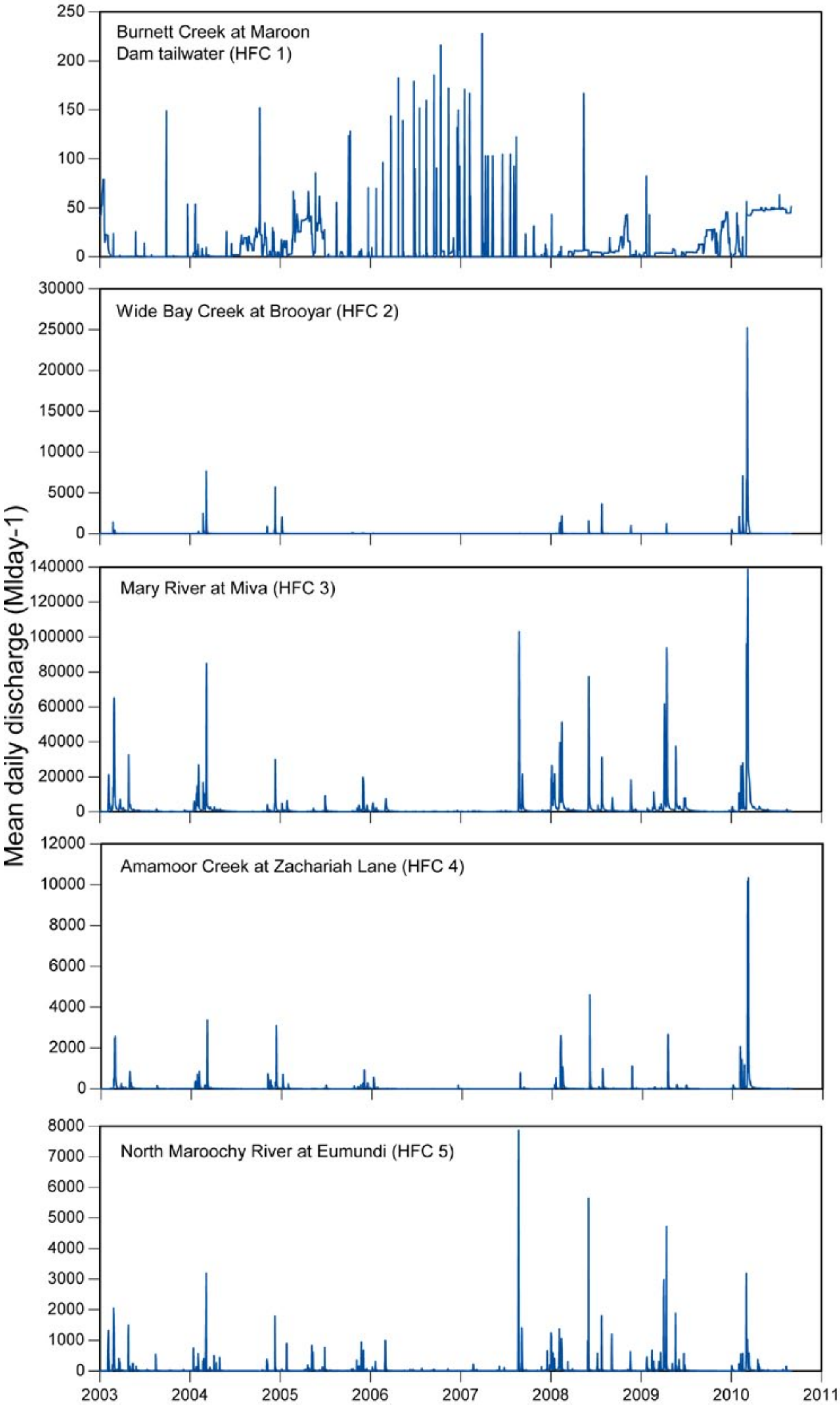
7.3.1 Flow conditions preceding aquatic vegetation sampling

The study area was influenced by drought prior to the commencement of sampling in 2008 (Figure 7.2). With the exception of the Noosa and South Coast catchments, the study area had been drought-declared prior to the commencement of field surveying in June 2007 (Queensland Government 2007).

Flooding occurred throughout the study period and the magnitude of the floods at some sites in the study area was relatively large compared with floods that had occurred in the five years preceding sampling (Figure 7.2). By the end of the final aquatic vegetation survey (September 2010) only a small area located in the south-west of the study area remained drought-declared (Queensland Government 2010).

Thus the aquatic vegetation survey period was preceded by drought but included periods of above-average rainfalls (e.g. Chandler 2009). The year 2010 was the wettest year on record for Queensland (Bureau of Meteorology 2011).

Figure 7.2: Hydrographs of flow conditions preceding and during aquatic vegetation surveys. Study period (June 2008 – August 2010) shown by arrows. Note differences in y-axis scales.



7.3.2 Environmental variation across Historic flow classes

HFCs varied in their environmental characteristics (Figure 7.3a,b). Sites in HFC 5 occurred at higher longitudes (i.e. closer to the coast) than sites in other HFCs and also occurred in areas with higher rainfall, higher riparian canopy cover and were more turbid.

HFCs 1–4 were distributed along a gradient of latitude (sites in HFC 1 were located at higher or more southern latitudes), BFWID_BFDEP, A_TEMP (annual mean temperature), flow variability (CVDaily) and days since the median particle size was mobilised (DAYS_Q_D50). Environmental differences between HFCs as shown by ordination were supported by ANOSIM (Global ANOSIM R = 0.304, p=0.001, Table 7.6). Most of the significant *post hoc* comparisons involved HFCs 1 and 5 (Table 7.6).

7.3.3 Variation in channel morphology across Historic flow classes

The ratio of bankfull width:bankfull depth (BFWID_BFDEP) was used to investigate differences in channel morphology between HFCs (Figure 7.4). The ratio of bankfull width: bankfull depth was relatively high for HFCs 1–3 (median value approximately 15) but comparatively low for HFCs 4 and 5 (median value less than 10, Figure 7.4). Thus sites HFCs 1–3 had relatively wide channels and HFCs 4 and 5 had more incised channels.

The ratio of bankfull width:bankfull depth (BWID_BFDEP) varied significantly between HFCs (Kruskal–Wallis test, P<0.001; Figure 7.5). Differences occurred between HFCs 2–4 and 3–4 (Figure 7.4). Removal of outliers (Eudlo Creek site 19, HFC 5, BFWID_BFDEP ratio 36.8; Reynolds Creek site 20, HFC 1, BFWID_BFDEP ratio 44.6) did not influence the result of the Kruskal–Wallis test or the result of multiple comparisons.

Figure 7.3: Environmental variation across HFCs as shown by nMDS (stress 0.145, three dimensions)

(a) Location of sites in ordination space as represented by HFC membership. Vectors show environmental metrics significantly correlated with the ordination (p<0.001).

(b) Distance of sites in relation to HFC centroids.

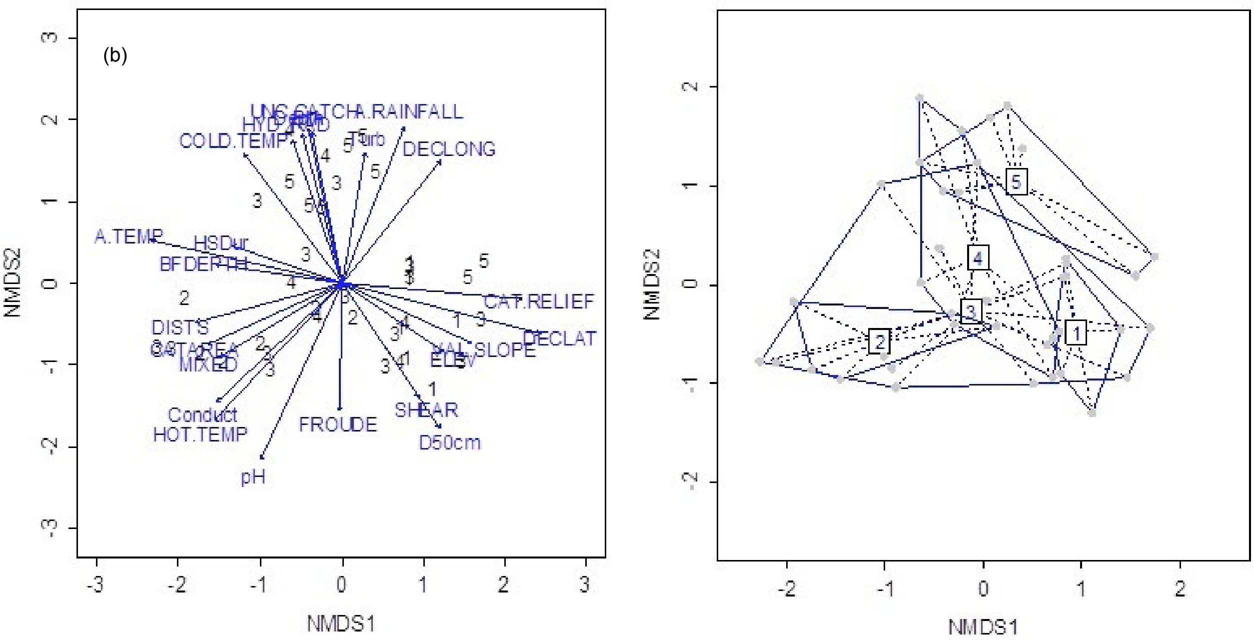


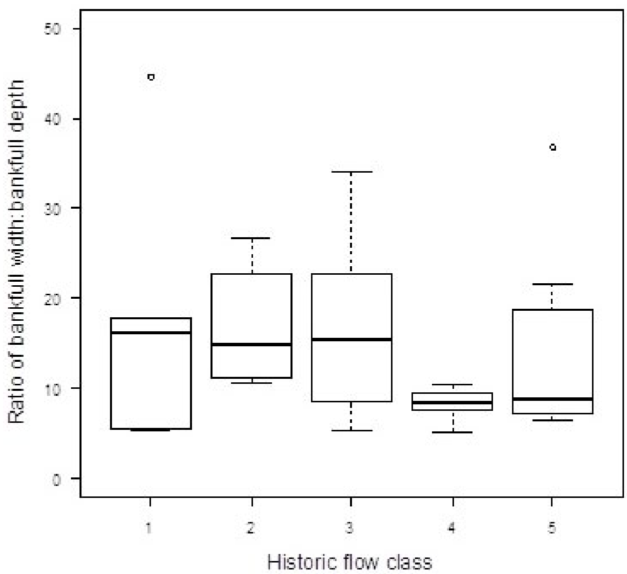
Table 7.6: Summary of ANOSIM testing the hypothesis that environmental characteristics of study sites did not vary significantly between HFCs

Global R = *0.304* (p=*0.001). Significance values for pairwise comparisons shown in brackets. Environmental data were range standardised prior to analysis.*

Factor	Global R	p	Significant pair wise comparisons
Historic flow class	0.304	0.001	1-2 (p = 0.002)
			1-4 (p = 0.005)
			1-5 (p = 0.001)
			2-5 (p = 0.001)
			3-5 (p = 0.001)

Figure 7.4: Variation in the ratio of bankfull width:bankfull depth across HFCs

HFCs with the same subscript are significantly different (Bonferroni-corrected significance value p***<0.005). The box represents the 25th (top) and 75th percentiles (bottom) and contains half of the points. The horizontal line within the box is the median. The whiskers extend to 1.5 times the interquartile range. Values outside of the range of the whiskers are shown as circles.***



Summary of environmental patterns

There were significant environmental variations across the HFCs, which were in part associated with differences in the location of HFCs within the study area. The majority of sites in HFC 1 were located in the southern part of the study area and most sites in HFC 5 were located close to the coast.

Significant differences in channel morphology occurred between HFCs, as measured by the ratio of bankfull width to bankfull depth (BFWID_BFDEP). In particular, sites in HFC 4 had significantly lower values for BFWID_BFDEP than sites in HFCs 2 and 3.

7.3.4 Aquatic vegetation

A total of 74 taxa were recorded from in-stream vegetation surveys (Attachment 7.2). The most common taxa were the submerged species *Potamogeton crispus* L., *Myriophyllum* sp. (probably mostly *M. verrucosum* Lindl.), mosses, and the emergent species *Lomandra* spp., *Carex* spp., *Hydrocotyle* spp. and *Persicaria decipiens* (R.Br.) K.L. Wilson (Figure 7.5). Eighteen taxa were recorded at a single site only and most taxa were recorded at less than 20% of sites.

Total richness ranged from zero to 18 taxa (Figure 7.5). In-stream vegetation was absent from three individual sites and 22 of the 134 samples in total. Sites at which in-stream vegetation was absent included Six Mile Creek (site 15), Eudlo Creek (site 19) and the Stanley River (site 26). These sites are all from HFC 5. Floating/attached vegetation (e.g. waterlilies) was rare in study sites. Low values were recorded for cover of floating/attached vegetation (FACOV) and number of floating/attached species (FARICH) (Figure 7.6), so these metrics are not considered further.

Total in-stream cover was generally less than 20% (median 10%, Figure 7.5) but high values (> 40%) were recorded at some sites (site 13 Obi Obi Creek, Burnett Creek downstream of Maroon Dam and Wide Bay Creek). Emergent vegetation cover (EMCOV) tended to be higher than submerged vegetation cover (SUBCOV) and therefore EMCOV contributed more to total in-stream cover than SUBCOV (Figure 7.5, 7.6).

Alien taxa comprised 27% of the in-stream taxa recorded (Figure 7.5). The most commonly recorded alien taxa were Watercress (*Rorippa nasturtium-aquaticum* (L.) Hayek), Mist Flower *(Ageratina riparia* (Regel) R.M. King & H. Rob.) and *Cyperus eragrostis* Lam. Alien taxa such as *Ageratum* spp. (Billygoat Weed), Conyza spp. (Fleabane) and *Cardamine* spp. (Bittercress) are not aquatic species but are associated with disturbed habitats.

Submerged (SUB) and Amphibious Fluctuation-Tolerators (ATE) were the most common functional groups (Brock and Casanova 1997) recorded from the study area (Figure 7.5). ATL (Amphibious Fluctuation-Tolerators – Low Growing) and TDR (Terrestrial – Dry Places) were rare in the study area (Figure 7.6) and were not considered further.

Figure 7.5: Frequency of occurrence of in-stream vegetation species at 40 sites in the SEQ study area

See Attachment 7.2 for species acronyms. Alien taxa are shown in red text. Insets show the occurrence of each Brock and Casanova functional group as a percentage of all species recorded at each site, and box and whisker plots of total richness and total cover values (all sites and times combined). Functional group codes: SUB, Submerged; ATE, Amphibious FluctuationTolerators (Emergent); ATL, Amphibious Fluctuation-Tolerators (Low Growing); ARF, Amphibious Fluctuation-Responders (Floating/Stranded); ARP, Amphibious Fluctuation-Responders (Morphologically Plastic); TDA, Terrestrial (Damp Places); TDR, terrestrial (Dry Places).

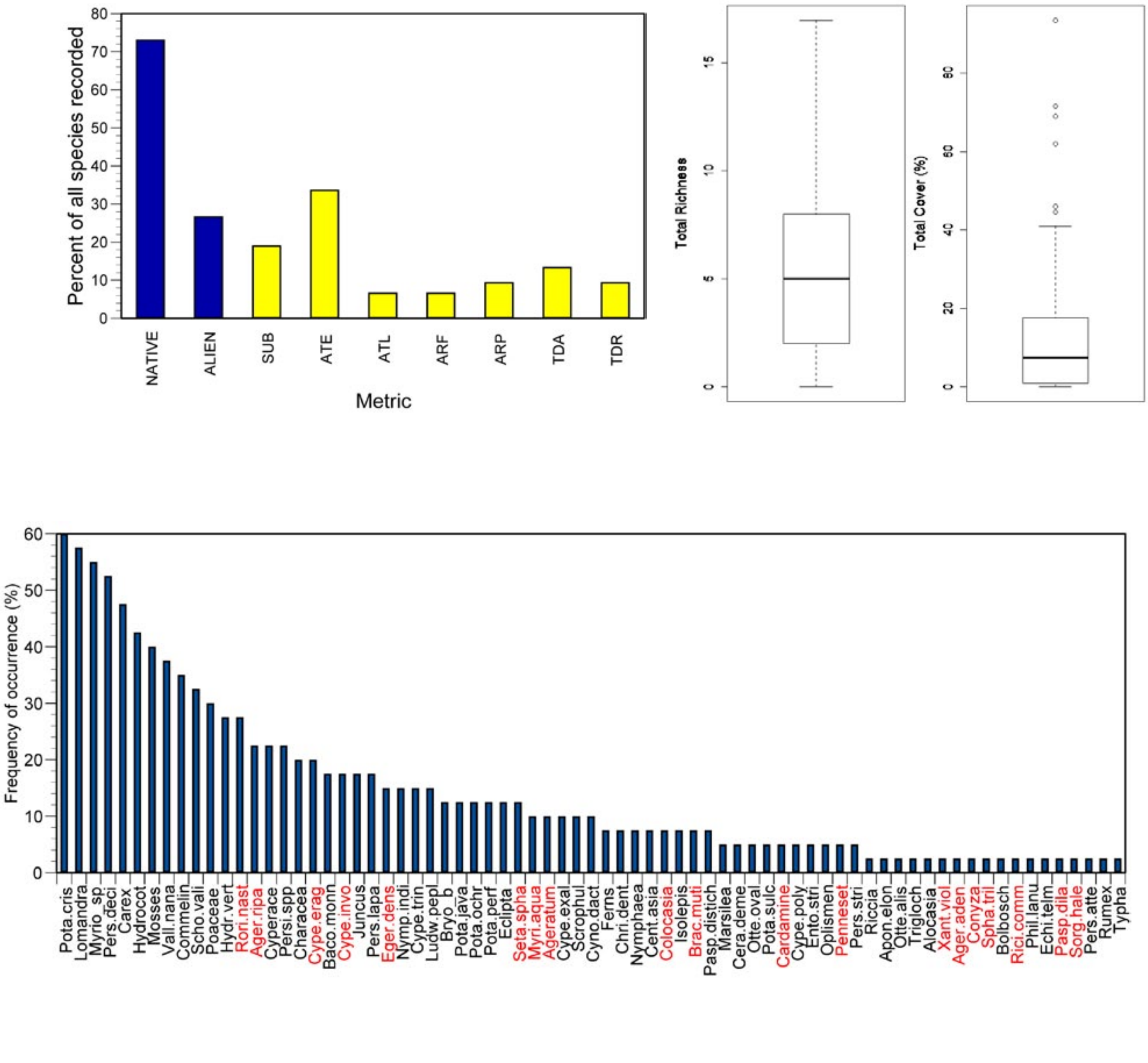
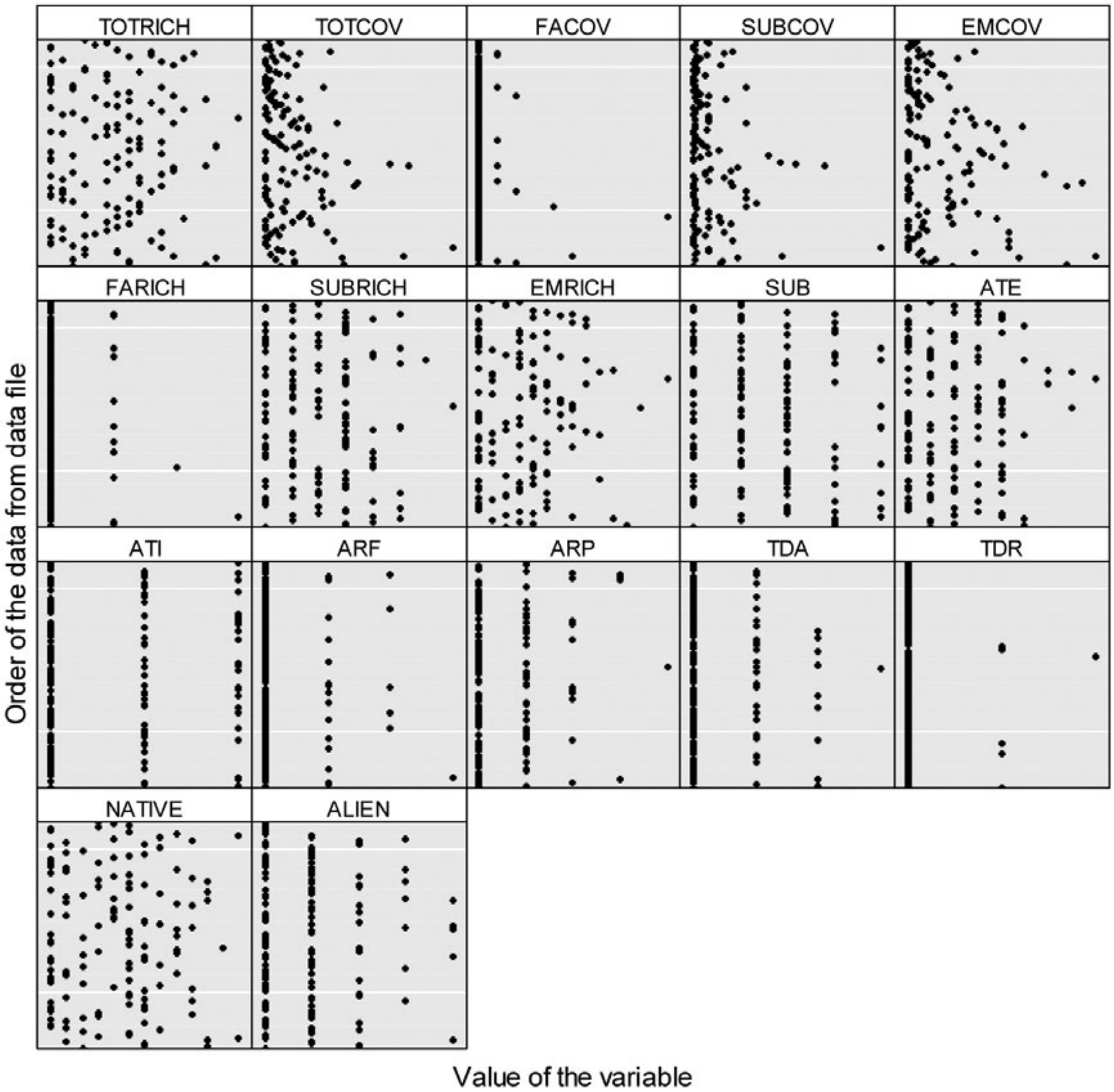


Figure 7.6: Cleveland dotplots showing individual values for 17 vegetation metrics

The** x**-axis represents the values for individual metrics and the y-axis counts for each metric value. See Table 7.2 for description of vegetation metrics.

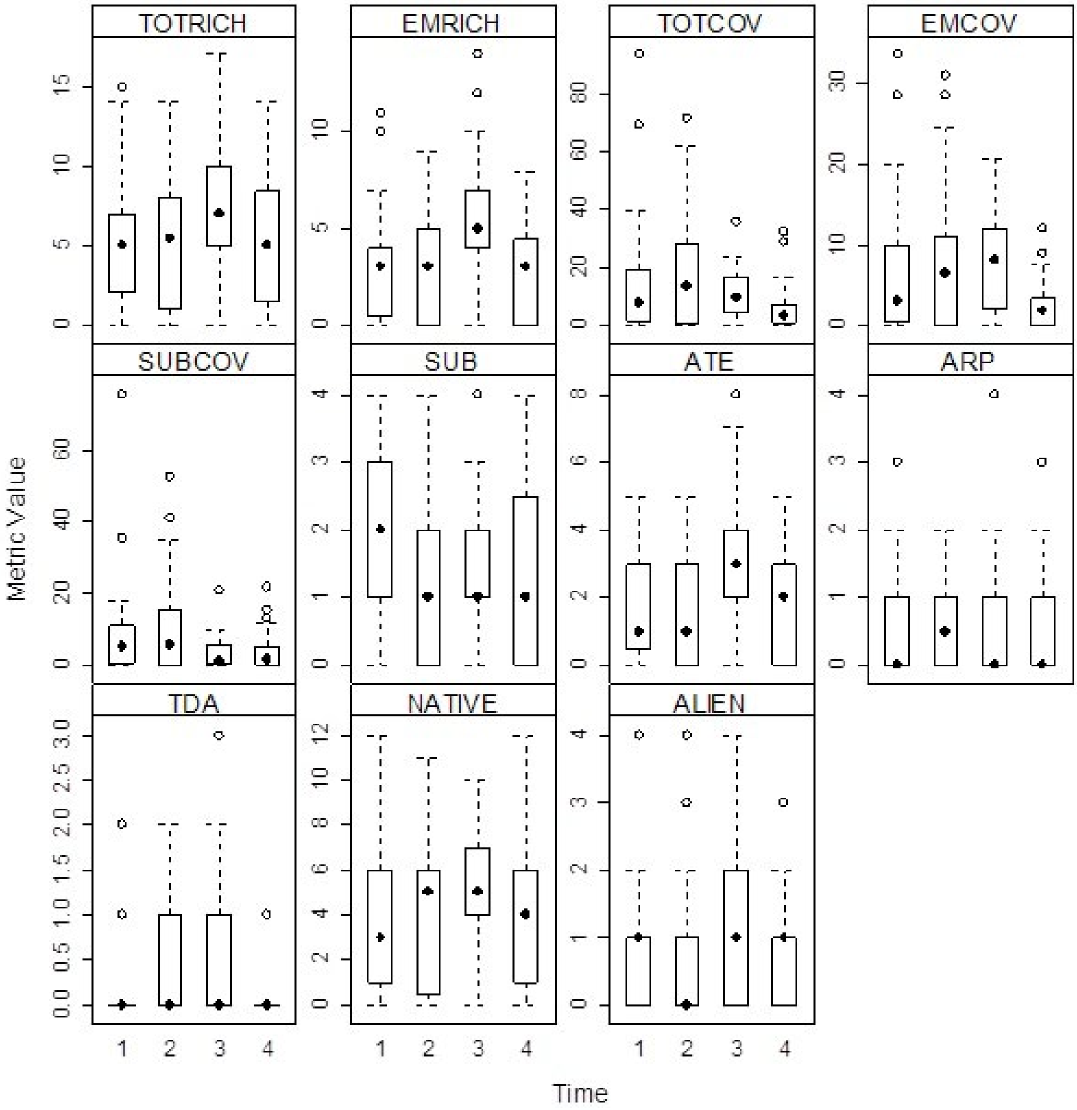


Most vegetation metrics did not vary appreciably over time (Figure 7.7). Total richness (TOTRICH) and emergent species richness (EMRICH) peaked in survey 3 and both decreased in value at the fourth survey. Total Cover (TOTCOV) peaked in survey 2 and then decreased over the remaining two surveys. In contrast, emergent species cover (EMCOV) peaked in survey 3 (Figure 7.7). SUBCOV also peaked at the second survey, decreased at the third survey (post flooding) but increased slightly at the fourth survey.

The most variable metrics over the sampling period were metrics describing vegetation cover i.e. TOTCOV, EMCOV and SUBCOV (Figure 7.7). These metrics had slightly different temporal trajectories. EMCOV was not apparently affected to the same extent as SUBCOV by the flooding experienced in the study area prior to the third survey in early 2010, as shown by the interquartile ranges for these metrics (i.e. the length of the box). The substantial reduction in EMCOV at the fourth survey does not correspond with a period of flooding (Figure 7.2).

Figure 7.7: Box and whisker plots of values for vegetation metric across the four survey times

See Table 7.2 for description of vegetation metrics and Figure 7.4 for description of box and whisker plots.



Summary

Seventy-four taxa were recorded in the study area. In-stream vegetation was absent from three sites, all belonging to HFC 5. The maximum site species richness recorded was 18 species. Patterns in total richness were driven more by emergent species richness than submerged species richness. Total in-stream cover was generally low (median value less than 10% cover) but varied considerably across the study area.

Of the 17 metrics trialled in this study FACOV, FARICH, TDR, SUB, ATL and ARF did not vary sufficiently across the study area to be useful measures of in-stream vegetation response to flow regime change.

7.3.5 Vegetation patterns across Historic flow classes

Ordination of presence–absence and species cover data were compared using Procrustes rotation. Since both ordinations were highly correlated (p<0.001) only results for the species cover ordination are discussed here.

Ordination of samples based on log-transformed species cover data identified three assemblage types present in the study area (Figure 7.8). The first assemblage was characterised by *Lomandra* spp. and occurred in sites with high riparian canopy cover, high Reynolds number and low water conductivity (Figure 7.8a,b).

The second assemblage was dominated by bryophytes, *Persicaria* spp., *Commelina* spp. and *Ageratina riparia* (Mistflower). This assemblage was dominated by non-vascular taxa (bryophytes) and emergent taxa and occurred in sites at higher latitude (i.e. more southerly regions of the study area) and higher elevations with coarse substrates (higher D50) and catchments with largely natural environments (PNE). The third assemblage was dominated by a mix of submerged (*Hydrilla verticillata, Potamogeton crispus, Vallisnera nana*) and amphibious species (*Hydrocotyle* spp., *Myriophyllum* spp. *Ludwigia peploides* subsp. *montevidensis*). This assemblage was associated with higher water conductivities, low riparian canopy cover and low Reynolds number.

While three broad aquatic vegetation assemblage types were identified by ordination there were few differences in assemblage composition associated with the HFCs (Figure 7.8c). The most obvious difference is that the centroid for HFC 1 is slightly displaced from the other flow class centroids in ordination space (Figure 7.8c). However, ANOSIM indicated that there were significant differences in assemblage composition between the HFCs (Table 7.7).

The results should be interpreted with caution due to the low global R value (0.106 – a value of zero indicates randomness). Most pairwise comparisons (7 out of 10) were significantly different, with the exception of HFCs 1–3, 2–3 and 3–4. Non-significant pairwise comparisons were similar to the non-significant pairwise comparisons undertaken for the ANOSIM with environmental data ANOSIM (Table 7.6).

Figure 7.8: Non-metric MDS ordination of sites based on log*x*+1 transformed species cover data (*x*>0), stress = 0.191, three dimensions

(a) Position of sites in ordination space as represented by HFC membership. Vectors show taxa significantly correlated with the ordination (p=0.01).

(b) Environmental variables significantly correlated with the ordination (p≤0.01).

(c) Distance to group centroids for sites in each HFC. See Table 7.1 for environmental variable acronyms and Attachment 7.2 for species acronyms.

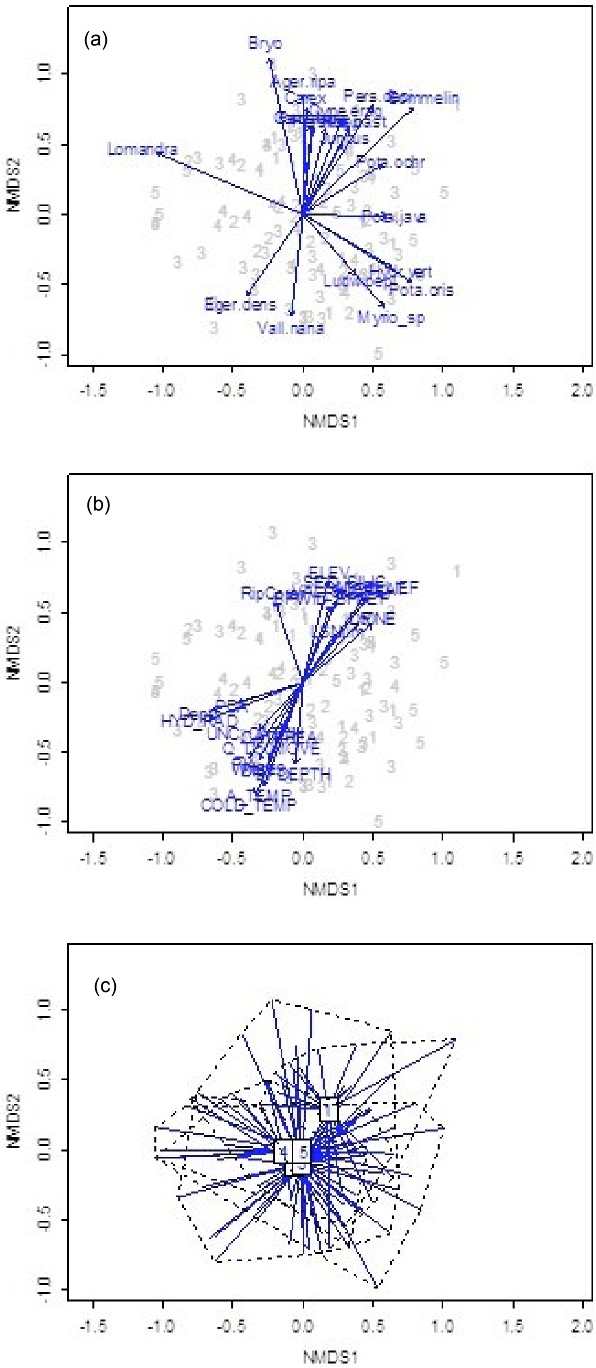


Table 7.7: Summary of ANOSIM results testing the hypothesis of no difference in aquatic vegetation composition between HFCs

Vegetation data logx+*1 transformed (for* x>0), *Bray–Curtis dissimilarity used to calculate the association matrix. Analysis carried out on samples with aquatic vegetation present* (n=132).

Factor	Global R	p	Significant pairwise comparisons (Bonferonni adjusted)
Historic flow class	0.106	0.002	1-2 (*p* = 0.001)
			1-4 (*p* = 0.002)
			1-5 (*p* = 0.001)
			2-4 (*p* = 0.001)
			2-5 (*p* = 0.001)
			3-5 (*p* = 0.002)
			4-5 (*p* = 0.002)

Random forests models showed that presence–absence and species composition data were poor indicators of HFC, as shown by the high OOB error rate (Table 7.8). The presence–absence model was a slightly superior model to the species cover model in terms of error rates. Classification errors from the presence–absence model for individual HFCs were lower than or equal to the classification errors from the species cover model for four out five HFCs. The species cover model performed better than the presence–absence model in allocating samples to HFC 1, comprised mostly of sites influenced by flow regulation.

For both models HFC 3 had the lowest prediction error and HFC 4 the highest prediction error. Most of the erroneously allocated sites were allocated to HFC 3, which had broadly similar flow regime attributes to HFC 4 (Chapter 3). ANOSIM showed no significant differences in aquatic vegetation composition between HFCs 3 and 4 (Table 7.7). For HFC 1, errors in class allocation were mostly associated with allocation of samples to HFC 3 (and *vice versa* for HFC 3). For HFC 2, erroneously allocated samples were mostly allocated to HFC 3.

The random forests models suggest several points. Firstly, presence–absence data performed slightly better than species cover in allocating samples to HFCs. Secondly, prediction error varied substantially between the HFCs. The lowest errors of prediction were associated with HFCs 3 and 5. The principal sources of error for allocation of samples to HFCs were:

- HFC 1 – allocation of samples to HFC 3
- HFC 2 – allocation of samples to HFC 3
- HFC 3 – allocation of samples to HFC 1.

Table 7.8: Confusion matrix for presence–absence and species cover random forests models

The confusion matrix shows how well each biotic dataset can be used to predict the HFC membership of sites. The number of trees grown for each model was 1000 and eight taxa were used at each split. OOB error rate estimate for the presence–absence model was 36.4% and for the species composition model 38.2%.

Observed HFC	HFC predicted by random forests model					
Presence–absence	**HFC1**	**HFC2**	**HFC3**	**HFC4**	**HFC5**	**Error (%)**
HFC1 (*n*=21)	13	1	6	1	0	38.1
HFC2 (*n*=15)	0	8	6	0	1	46.7
HFC3 (*n*=46)	7	0	37	1	1	19.6
HFC4 (*n*=15)	1	1	10	2	1	86.7
HFC5 (*n*=13)	2	0	1	0	10	23.1
Species composition	**HFC1**	**HFC2**	**HFC3**	**HFC4**	**HFC5**	**Error (%)**
HFC1 (*n*=21)	15	1	4	1	0	28.6
HFC2 (*n*=15)	0	6	7	0	2	60.0
HFC3 (*n*=46)	7	0	36	2	1	21.7
HFC4 (*n*=15)	0	0	13	1	1	93.3
HFC5 (*n*=13)	2	0	1	0	10	23.1

Constancy values were used to determine if there were aquatic vegetation taxa with indicator value for individual HFCs. Constancy is the number of times a species occurs in the sites in each HFC. An individual taxon has indicator value if it occurs in a high proportion of members within a flow class but in no other flow classes.

No taxon had indicator value since most occurred in too few samples within an individual HFC, or occurred in a relatively high proportion of samples across several HFCs (Table 7.9).

However, several taxa were very common in a single HFC. *Ageratina riparia* (Mistflower) and *Commelina* spp. were common in HFC 1. *Marsilea* sp. occurred only in HFC 2, although at a relatively low frequency of occurrence (Table 7.9). Although *Vallisneria nana* occurred at high frequencies across several HFCs, it was especially common in samples in HFC 2 (75% occurrence, Table 7.9).

Table 7.9: Constancy values for aquatic vegetation taxa

Shown are taxa occurring in at least 20% of the samples in at least one HFC. Occurrences in samples of 1–19% represented by'+'. Blank cells indicate absence from the flow class.

Taxon	HFC1	HFC2	HFC3	HFC4	HFC5
Ageratina riparia	0.50		+	+	+
Commelina spp.	0.50	+	+		+
Carex spp.	0.63	0.31	0.34	0.52	+
Persicaria decipiens	0.45	0.43	0.32	0.28	+
Potamogeton crispus	0.45	0.37	0.39	0.28	+
Bryophyta	0.40	0.31	0.23	0.28	+
Marsilea sp.		0.25			
Bacopa monnieri		0.37	+		
Cynodon dactylon	+	0.37			
Potamogeton perfoliatus		0.25	0.23		
Ludwigia peploides		0.25	+	+	+
Cyperus involucratus		0.31	+	0.23	
Vallisneria nana	+	0.75	0.30	0.28	
Hydrocotyle spp.	+	0.62	0.36	0.14	+
Lomandra spp.	+	0.50	0.32	0.47	0.22
Egeria densa			0.26	+	
Myriophyllum sp.	0.22	0.43	0.47	0.33	+
Schoenoplectus validus	+	0.25	0.21	0.42	+
Potamogeton octandrus			+	+	0.22
Hydrilla verticillata	+		+	0.23	0.25

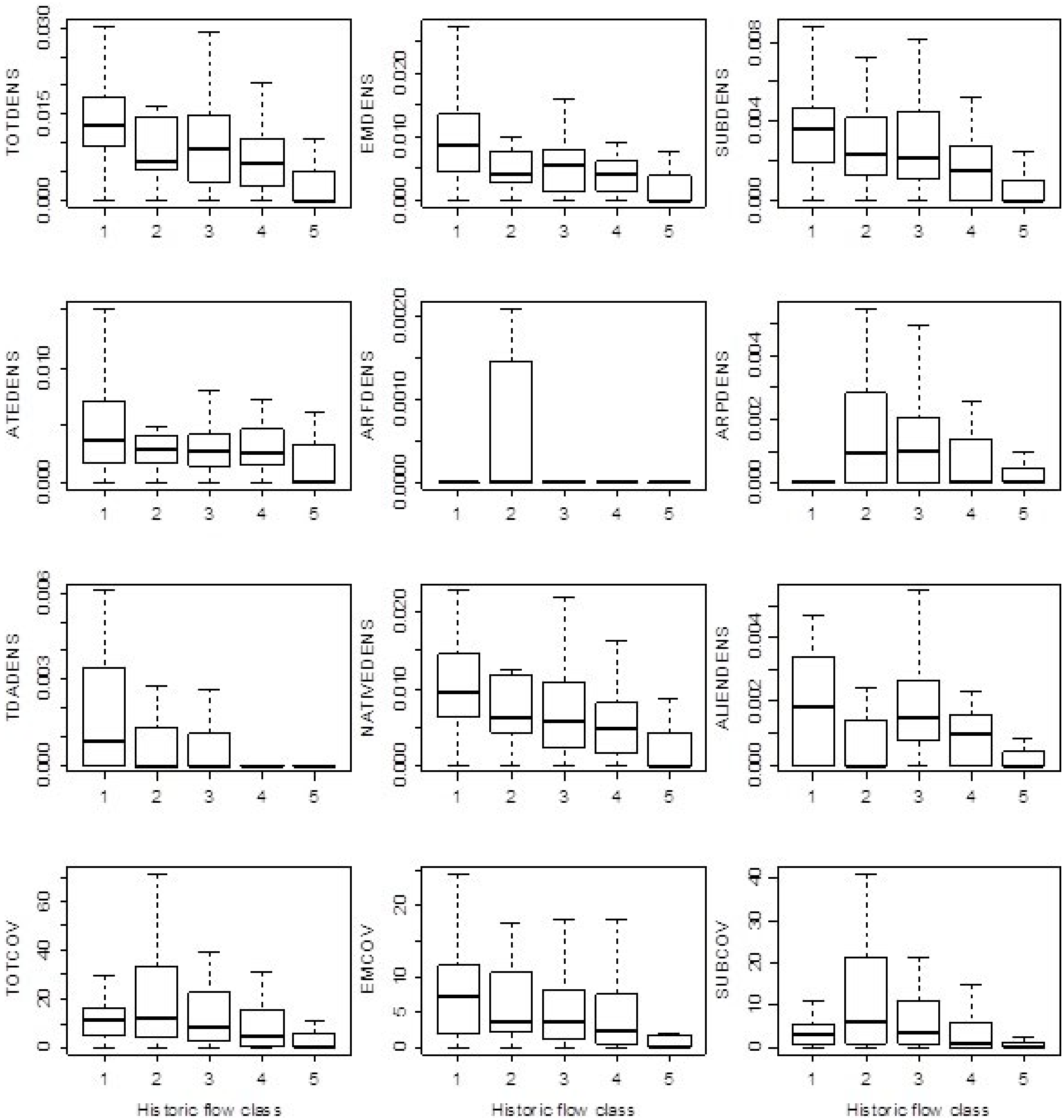
Patterns in aquatic vegetation metrics

Kruskal–Wallis tests indicated that seven vegetation metrics varied significantly between HFCs (Figure 7.9). Six of these significant results were due to differences between HFCs 1 and 5. In general, median values for individual vegetation metrics were highest for HFC 1 and lowest for HFC 5.

Exceptions to this trend were ARP (density of Amphibious Fluctuation-Responders – morphologically plastic), ARF (Amphibious Fluctuation-Responders – floating/stranded) and SUBCOV (submerged vegetation cover). Densities of amphibious taxa (ARF and ARP) were highest in HFCs 2 and 3 respectively. SUBCOV was highest in HFC 2 (Figure 7.9).

Figure 7.9: Box and whisker plots of in-stream vegetation metrics for individual HFCs

The results of multiple comparison tests (Tukeys HSD) are shown where Kruskal–Wallis tests showed significant differences in vegetation metrics across HFCs (Bonferroni-corrected significance p ***= 0.005). See Table 7.2 for description of vegetation metric.***



Summary

Three aquatic vegetation assemblages were identified in the study area. The first assemblage was characterised by Lomandra spp. and occurred in sites with high riparian canopy cover, high Reynolds number and low water conductivity. The second assemblage was dominated by bryophytes, *Persicaria* spp., *Commelina* spp. and *Ageratina riparia* (Mistflower).

This assemblage was dominated by non-vascular taxa (bryophytes) and emergent taxa and occurred in sites at higher latitude (i.e. more southerly regions of the study area) and higher elevations with coarse substrates (higher D50) and catchments with largely natural environments (PNE). The third assemblage was dominated by a mix of submerged (*Hydrilla verticillata, Potamogeton crispus, Vallisnera nana*) and amphibious species (*Hydrocotyle* spp., *Myriophyllum* spp. *Ludwigia* peploides subsp. *montevidensis*). This assemblage was associated with higher water conductivities, low riparian canopy cover and low Reynolds number.

Although significant differences occurred in aquatic vegetation assemblages across HFCs (ANOSIM), random forests models showed that species assemblage data had little capacity to correctly allocate samples to HFCs. Supporting this analysis, constancy values showed that no taxon had value as an indicator of HFC.

Seven aquatic vegetation metrics varied significantly across HFCs. Highest values for aquatic vegetation metrics (i.e. highest vegetation cover and highest density of functional groups) were generally recorded in HFC 1 (flow regulated sites) and lowest values in HFC 5 (coastal streams).

7.3.6 Relative importance of hydrology versus other environmental factors in structuring aquatic vegetation

Environmental variables used partial CCA explained 42.9% in vegetation assemblage data (Table 7.10). The unexplained variation may be due to the limitations of the environmental variables in explaining aquatic vegetation patterns at the assemblage scale (i.e. potentially important environmental variables were not included) and/or the influence of stochastic (random process), for example chance colonisation events. The ‘pure’ variation explained by three environmental datasets (i.e. TEMPVARS, HYDRO, and CATCHMENT) was 6.7%, 4.1% and 23.0% respectively (Table 7.10). Thus catchment, land use and climate variables (CATCHMENT) explained a greater proportion of the pure environmental variation than TEMPVARS and HYDRO combined (Table 7.10).

Table 7.10: Results of variance partitioning by partial Canonical Correspondence Analysis

Environmental data were range standardised and biotic data were logx+1 transformed (for x>0) prior to analysis. Samples without aquatic vegetation were excluded from analysis. Not all components of variation are shown, as interpretation of these components with respect to the scheme of Anderson and Gribble (1998) is difficult because two of the three datasets (TEMPVARS and HYDRO) were temporal datasets.

Component of variation	Variation explained
TEMPVARS	6.7%
HYDRO	4.1%
CATCHMENT	23.0%
Total explained variation	42.9%
Unexplained variation	57.1%

Significant variables for each dataset (as identified by forward selection):

TEMPVARS: Conduct, Turb, RipCover, pH, REYNOLD.

HYDRO: HIS_CLASS, CVDaily, DAYS_Q_D50, Q_D50MOVE.

CATCHMENT: A_RAINFALL, DISTS, MIXED, CAN, MAFIC, IU, DECLONG, PDA, BFWID_BFDEP, VAL_SLOPE, UNC_CATCH, PIA, ORIENT, CAT_ELONG, B_SLOPE.

Regression random forests models for aquatic vegetation metrics indicated that D50, RipCover, PIA, PDA, UNC_CATCH and BFSHEAR were the most important environmental parameters describing patterns in vegetation metrics (Figure 7.10). In particular, D50 was especially important, being the most important environmental variable (i.e. removal of this variable caused the largest change in mean square error) in six of the eight random forests models. Variation in dependent variables explained by the random forests models ranged from 13% for the ALIEN model to 65.5% for the TOTCOV model (Table 7.11). Most models explained greater than 40% of the variation in individual vegetation metrics.

Hydrologic metrics were considered to be relatively unimportant in the regression random forests models, as determined by the increase in mean squared error. The most important hydrologic metrics (in terms of change in mean square error) were DAYS_Q_D50 (SUBCOV) and D50_HSDur (TOTCOV). Flow metrics had the greatest influence on SUBCOV (two flow metrics ranked in the 10 most important environmental parameters).

Figure 7.10: Environmental variable importance (as percentage change in mean squared error, MSE) for vegetation metric random forests models

For clarity only the 15 most important variables are shown. See Table 7.1 for environmental variable acronyms.

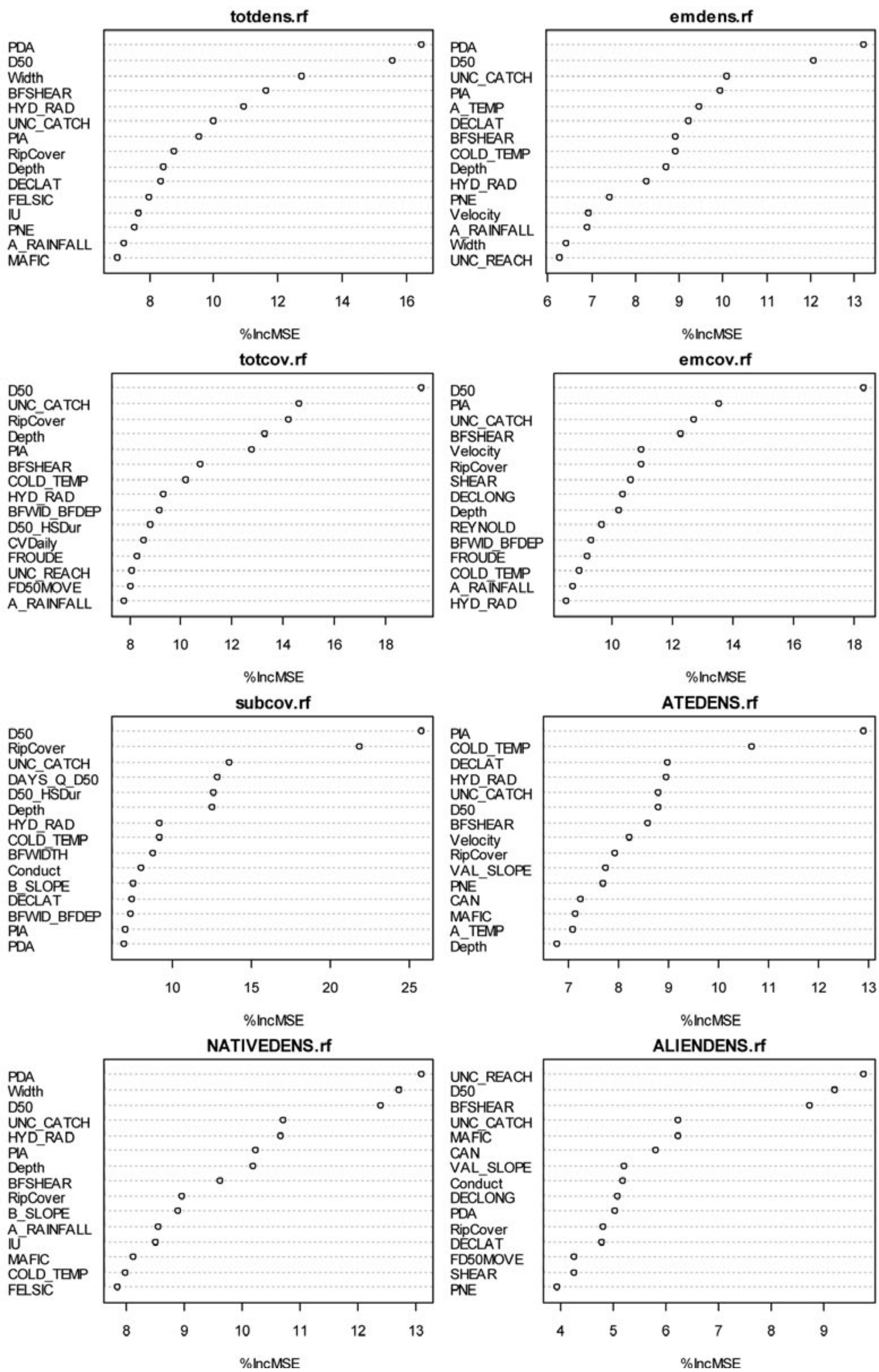


Table 7.11: Summary of regression random forests for vegetation metrics, and important flow metrics for each model (as determined by change in mean square error, shown in brackets)

Metric importance determined from the percentage increase in mean square error when that metric is removed. Total number of environmental variables used was 57.

Metric	R^2	Most important flow metric (% increase in mean square error)
TOTDENS	0.559	No flow metric in 30 most important environmental variables
EMDENS	0.429	No flow metric in 30 most important environmental variables
TOTCOV	0.655	D50_HSDur (10)
EMCOV	0.605	FD50MOVE (20)
SUBCOV	0.571	DAYS_Q_D50 (4)
ATEDENS	0.416	No flow metric in 30 most important environmental variables
NATIVEDENS	0.606	DAYS_Q_D50 (25)
ALIENDENS	0.130	FD50MOVE (13)

Summary

Regression random forests models for aquatic vegetation metrics indicated that D50, RipCover, PIA, PDA, UNC_CATCH and BFSHEAR were the most important environmental parameters describing patterns in vegetation metrics. Hydrologic parameters were relatively unimportant in describing variation in aquatic vegetation assemblages, when compared with hydraulic and catchment-scale parameters such as land use. This pattern was observed for species composition data and aquatic vegetation metrics. TOTCOV and SUBCOV showed the strongest relationships with hydrologic metrics.

7.3.7 Patterns in vegetation metrics across flow metric gradients

Relationships between selected vegetation metrics and flow metrics were modelled using GLS regression. Hydrologic metrics were identified as significant predictors in five of the models tested (Table 7.12). However, log-likelihood ratio tests indicated that only three of these models were a better fit than the null model (Table 7.12). The vegetation metrics for which significant relationships were found were TOTCOV (Hypotheses 2 and 3, flow metrics Q_D50MOVE and log(FD50MOVE)) and EMCOV (Hypothesis 3, flow metric log(FD50MOVE)).

The frequency of occurrence of the discharge required to mobilise the median particle size in the 12 months prior to sampling (FD50MOVE) was a significant predictor in both of the GLS models relating cover estimates (TOTCOV) to flood frequency. The relationships between vegetation cover and FD50MOVE were negative, indicating lower values for these metrics at higher frequencies of substrate mobilisation (Table 7.12; Figure 7.11).

GLS models with CVDaily as the predictor variable were not significant (Table 7.12), indicating no support for Hypothesis 4 (vegetation cover predicted to be positively related to discharge variability).

Table 7.12: Summary of GLS regression models describing relationships between vegetation metrics and selected flow metrics

D *is the value of the log-likelihood ratio test comparing model fits (i.e. the null model compared to the alternate model). Significance for* D *is determined by comparison with* X^2 *with one degree of freedom (3.841). Significance: * 0.01 ≥ p > 0.001; **p ≤ 0.001.*

Vegetation metric	Flow metric	Coefficient	Intercept	Log-likelihood (flow model)	Log-likelihood (null model)	*D*
Hypothesis 2						
TOTCOV	log(Q_D50MOVE)	1.746 ± 0.270 **	-0.107	-513.466	-541.951	56.97*
Hypothesis 3						
TOTCOV	HSNum	0.163 ± 0.316	11.432	-406.626	-406.524	—
TOTCOV	log(FD50MOVE)[1]	-2.060 ± 0.767 *	13.950	-166.275	-171.200	-9.85*
EMCOV	HSNum	-0.132 ± 0.189	6.771	-446.582	-446.079	—
EMCOV	log(FD50MOVE)[2]	-1.202 ± 0.354 **	6.456	-197.209	-202.471	10.52*
TOTDENS	HSNum	-0.0006 ± 0.0002 *	0.013	442.539	443.718	-2.358
TOTDENS	FD50MOVE[3]	-0.00008 ± 0.00002 **	0.009	225.802	225.740	0.124
EMDENS	HSNum	-0.0004 ± 0.0001	0.009	484.271	488.796	—
EMDENS	FD50MOVE x>0[3]	$-5 \times 10^5 \pm 9 \times 10^6$**	0.005	251.797	250.979	1.636
Hypothesis 4						
TOTCOV	CVDaily	-0.001 ± 0.006	12.860	-495.761	-491.586	—
EMCOV	CVDaily (all sites)	-0.00016 ± 0.004	5.829	-450.764	-446.079	—
TOTDENS	CVDaily	$1 \times 10^6 \pm 4 \times 10^6$	0.008	432.308	443.718	—
EMDENS	CVDaily	$2 \times 10^7 \pm 3 \times 10^6$	0.006	476.978	488.796	—

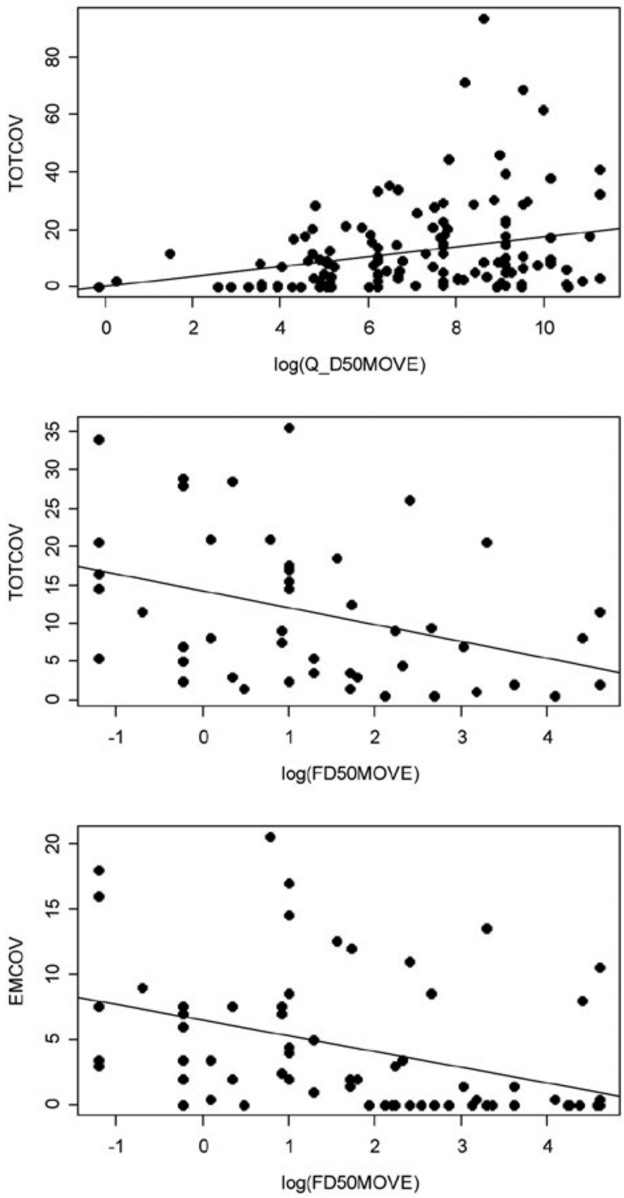
[1] *For values FD50MOVE > 0 and TOTCOV > 0*

[2] *For values FD50MOVE >0 and EMCOV > 0*

[3] *For values FD50MOVE > 0 and TOTDENS > 0, outlier (site 42) deleted.*

Figure 7.11: Plots of model fits for significant GLS models describing relationships between vegetation metrics and selected hydrologic metrics

See Table 7.13 for GLS model summaries.



Summary

The results of GLS modelling support Hypothesis 2 (vegetation cover will be proportional to discharge magnitude) and Hypothesis 3 (aquatic vegetation cover will be inversely proportional to flood frequency). However, FD50MOVE (percentage of days in the 12 months prior to sampling when discharge was above the threshold required to mobilise the median particle size) was a better measure of flood frequency than the number of floods greater than the median flow (HSNum). There was no evidence that aquatic vegetation was limited to sites with 13 or less floods per year.

There was no support for Hypothesis 4 that aquatic vegetation abundance will be positively correlated with discharge variability, measured as the coefficient of variation of mean daily discharge (CVDaily).

7.3.8 Effects of flow regime change on aquatic vegetation

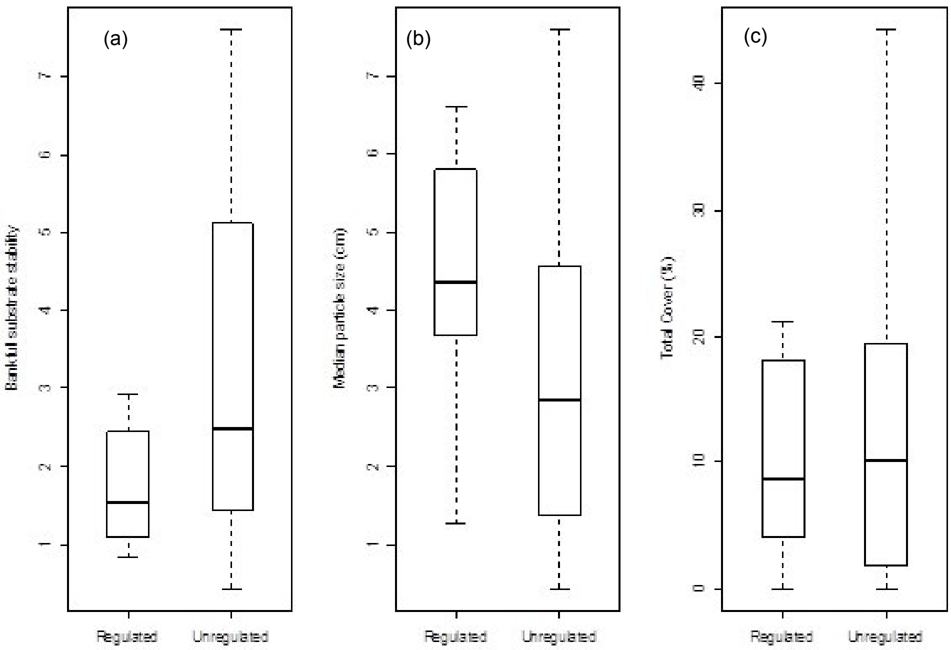
Hypothesis 5 predicted that aquatic vegetation cover would be higher in regulated sites compared with unregulated sites, assuming substrate stability in regulated sites was higher than unregulated sites. There was no difference in the median particle size between regulated and unregulated sites ($p>0.05$) but bankfull substrate stability (the ratio of bankfull shear stress to the critical shear stress required to mobilise the median particle size, BF_SUBSTAB) was significantly higher in unregulated sites (Kruskal-wallis test, $p<0.05$, Figure 7.12). However, since the median value for BF_SUBSTAB in regulated sites was greater than 1 it is evident that the median particle size is being mobilised at bankfull discharge in regulated sites.

Figure 7.12: Box and whisker plots of

(a) bankfull substrate stability for flow regulated (R), unregulated (U) sites

(b) median particle size for regulated and unregulated sites

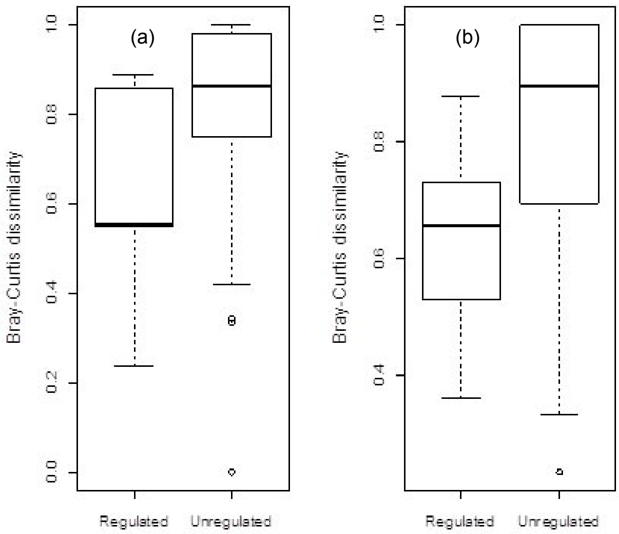
(c) total cover for regulated and unregulated sites.



ANOSIM showed that vegetation composition differed between regulated and unregulated sites within RFCs 1 and 5 (Figure 7.13). RFC 1 includes Obi Obi Creek at Kidaman, subsequently regulated by Baroon Pocket Dam. RFC 5 includes the Nerang River at Glenhurst, subsequently regulated by Hinze Dam. The ANOSIM statistic for the RFC 1 comparison was significant but low (R = 0.133, p=0.035). The ANOSIM statistic for the RFC 5 comparison was also significant (R = 0.247, p=0.027). Figure 7.13 suggests that there is greater dissimilarity in vegetation composition in unregulated sites compared with regulated sites within RFCs 1 and 5.

Figure 7.13: Box and whisker plots of ranked Bray–Curtis dissimilarities for comparisons of vegetation composition between regulated sites (R) and unregulated sites (U) in RFC 1 (a) and RFC 5 (b)

n=*28 for RFC 1 and* n=*19 for RFC 5.*



The effect of flow regulation on five vegetation metrics was determined using PLS regression. Reference models for each metric were constructed from predictor variables unaffected by flow regulation. The Reference model was then used to predict vegetation metric values at regulated sites.

The TOTCOV Reference model had the highest R^2 of the five vegetation metrics modelled (0.682, Table 7.13). Only one model (EMCOV) had an R^2 less than 0.6.

The effect of flow regulation on individual vegetation metrics was determined as ((observed value–predicted value)/predicted value)×100, using the Reference PLS models to predict vegetation metric values at regulated sites (Zhang et al. 1998). An effect of flow regulation was determined for TOTCOV (Table 7.15). Figure 7.14 shows that TOTCOV was higher than predicted at three sites and lower than predicted at nine sites. However, four sites had values for TOTCOV close to predicted, as shown by their position in relation to the 1:1 line (Figure 7.14). In general, values for aquatic vegetation metrics were lower than predicted at regulated sites.

Table 7.13: Summary of PLS model fits for selected vegetation metrics

The effect of flow regulation ('effect') is calculated as ((Observed–Predicted)/Predicted)×100. Vegetation metrics were log-transformed prior to analysis.

Metric	R^2	Components	Predictors	Effect
TOTDENS	0.612	1	PDA, PIA, PNE, IU, UNC_CATCH, DECLAT, FELSIC, MAFIC, RipCover, A_RAINFALL	-1.6 ± 22.4
TOTCOV	0.682	2	UNC_CATCH, PIA, BFWID_BFDEP, RipCover, UNC_REACH, A_RAINFALL, COLD_TEMP	20.0 ± 13.9
SUBCOV	0.644	5	DECLAT, RipCover, UNC_CATCH, B_SLOPE, COLD_TEMP, BFWIDTH, PIA, PDA, BFWID_BFDEP	353.2 ± 499.6
EMCOV	0.409	1	PIA, UNC_CATCH, DECLONG, RipCover, BFWID_BFDEP, COLD_TEMP, A_RAINFALL,	42.8 ± 164.0
NATIVE	0.600	5	PDA, PIA, B_SLOPE, A_RAINFALL, IU, FELSIC, MAFIC, UNC_CATCH, COLD_TEMP, RipCover	961.5 ± 1222.5

The effect of flow regulation for the five vegetation metrics was plotted against the Gower metric (Figure 7.15). This tested the ELOHA premise that increasing degree of flow regulation is associated with increasing biotic change. If this concept was valid then the effect of flow regulation should increase with increasing value for the Gower metric. While there are only six dams represented in Figure 7.15, it is evident that there is no consistent relationship between the effect of flow regulation and the extent of overall flow regime change, as measured by the Gower metric.

However, the effect of flow regulation on vegetation cover (TOTCOV) was significantly negatively correlated with the Gower metric. In general, there was little difference in the effect of flow regulation on aquatic vegetation downstream of dams where flow regime changes were high (i.e. high Gower metric value), as shown by the distance between sites downstream of each dam. The greatest differences in flow regulation effect occurred for sites downstream of Baroon Pocket Dam (Obi Obi Creek) and Six Mile Creek Dam (Six Mile Creek) where flow regime changes have been relatively minor (Figure 7.15).

The effect of flow regulation on flow regimes downstream of dams in the study area varies between dams (Chapter 3). Four main types of flow regime alteration downstream of dams in the study area were identified in Chapter 3. However, there was no evidence to suggest that the aquatic vegetation response to flow regime change (as measured by the effect of flow regulation) was consistent within each type of flow regime alteration. For example, Obi Obi Creek, Six Mile Creek and the Nerang River have similar flow regime changes but the vegetation response within this group clearly varies among sites (Figure 7.15).

Next, the effect of flow regulation for total in-stream cover (TOTCOV) was plotted against the change in value (i.e. change from Reference flow regime to Historic flow regime) for selected flow metrics (Figure 7.16). The selected flow metrics were those that had changed downstream of dams in the study area or discriminated between HFCs (Chapter 3). There were no significant correlations between the effect of flow regulation on TOTCOV and the change in flow metric values from Reference to Historic (Spearman correlation coefficients, $p>0.05$).

Figure 7.14: Plots of predicted versus measured values for PLS Reference models (left hand column) and PLS test models (right hand side)

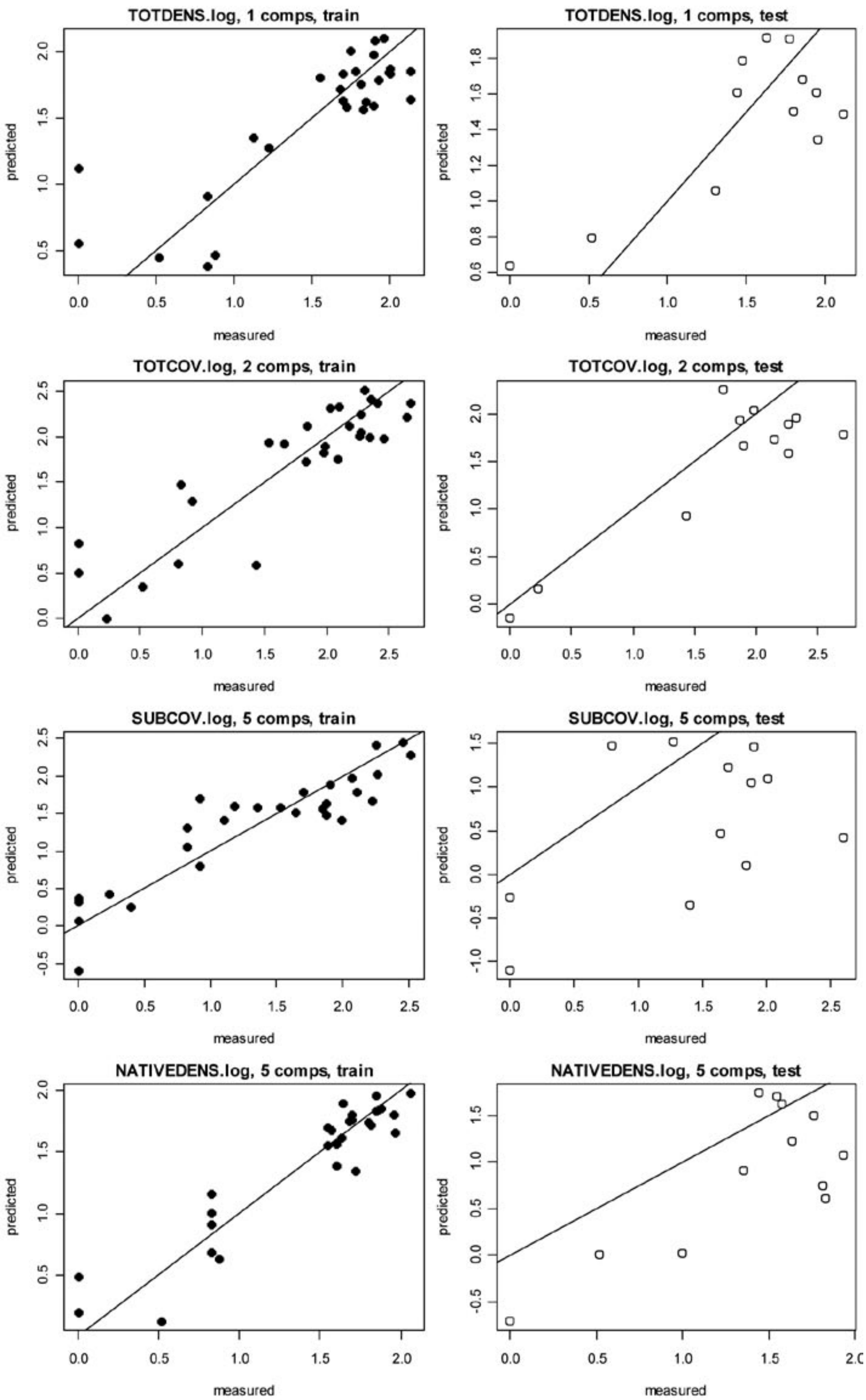


Figure 7.15: Scatterplots showing relationships between the effect of flow regulation versus the Gower metric (an indicator of overall flow regime change) for sites downstream of dams in the study area

Two sites were surveyed downstream of each dam and each dam is represented by a pair of points (the average of flow regulation effects for all samples). Site codes: Obi – Obi Obi Creek; Six – Six Mile Creek; Bur – Burnett Creek; Ybb – Yabba Creek; Rey – Reynolds Creek; Nrg – Nerang River.

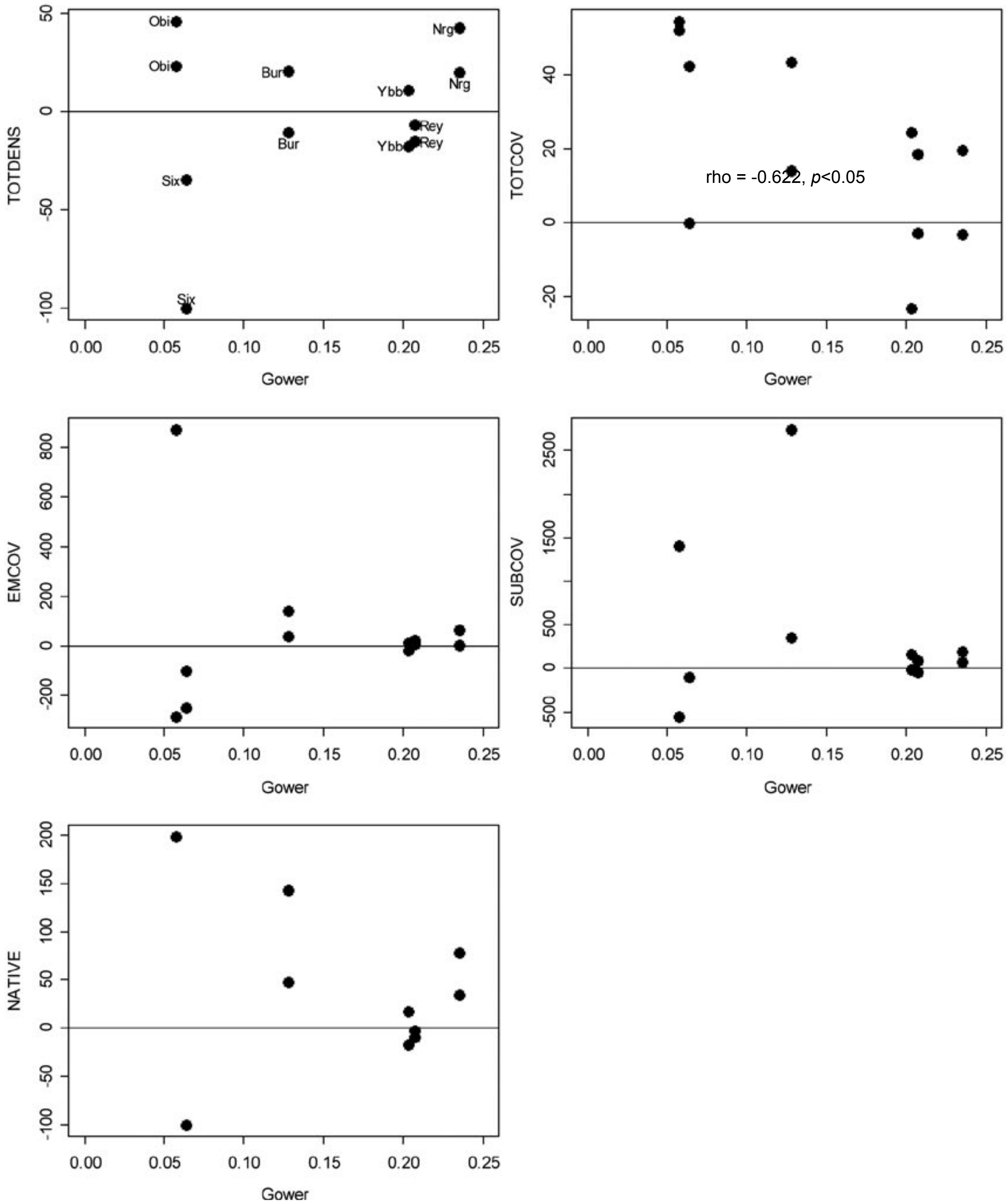
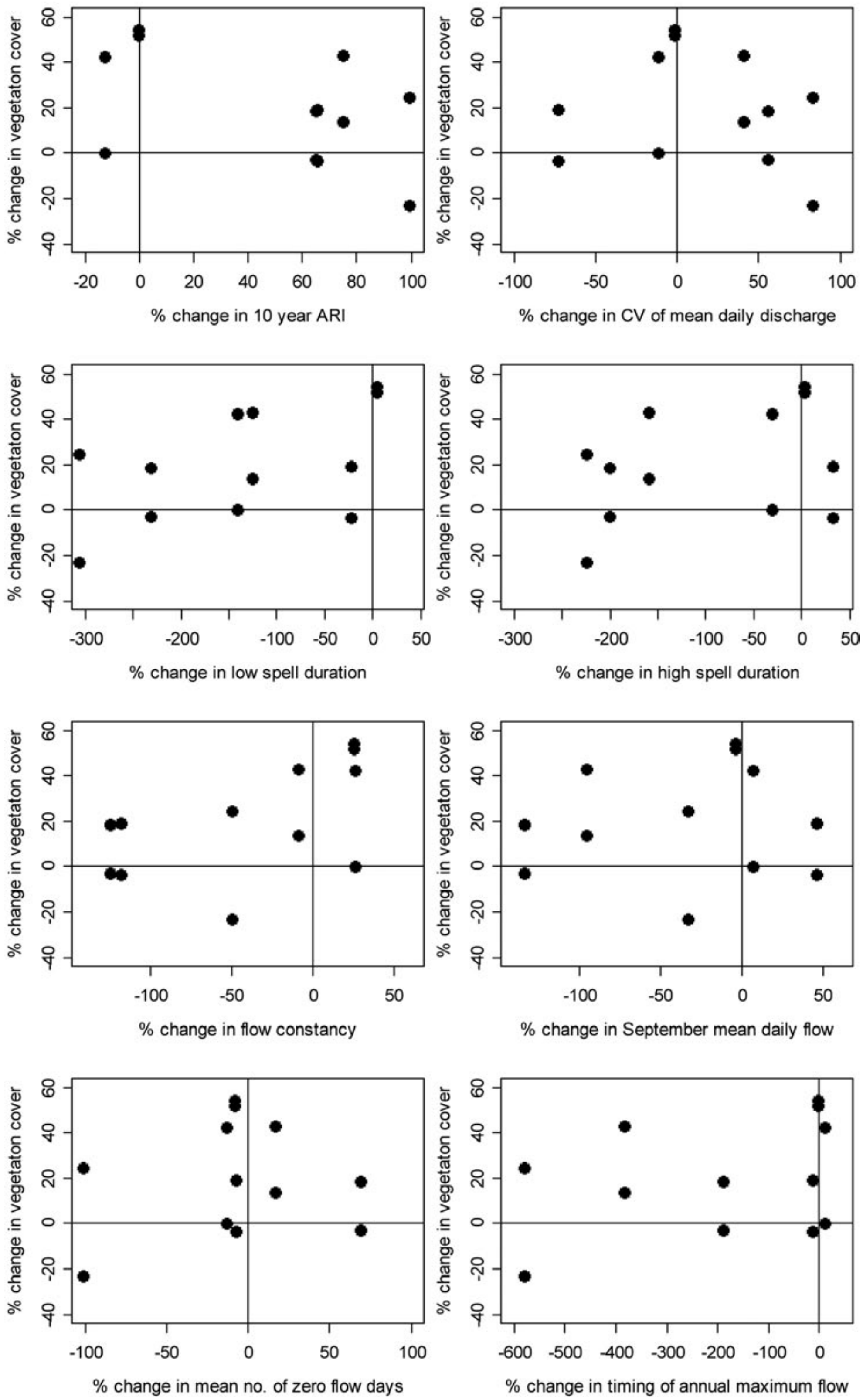


Figure 7.16: Scatterplots of change in total cover as predicted by PLS models (effect of flow regulation) *versus* %



Summary

Sites in RFCs 1 and 5 that were consequently subjected to flow regulation were found to have different aquatic vegetation assemblages compared to the unregulated sites in those RFCs. This implies that flow regime changes have caused changes in aquatic vegetation assemblage structure.

An effect of flow regulation was detected for total cover (TOTCOV) using PLS regression, although TOTCOV did not differ significantly between regulated and unregulated sites in the study area.

The Hypothesis that increasing flow regime change will result in increasing biotic change was not supported. In contrast, the effect of flow regulation on total cover was inversely related to the magnitude of flow regime change, as measured by the Gower metric. Thus relatively large flow regime changes were associated with small changes in total cover.

7.4 Discussion

The ELOHA framework is underpinned by several concepts. This report tested several hypotheses related to these concepts as a means of validating the framework for aquatic vegetation.

7.4.1 Hypothesis 1

Hypothesis 1 stated that streams with similar flow regime characteristics should be more similar in terms of aquatic vegetation attributes than streams with different flow regime characteristics. There was mixed evidence to support this hypothesis.

Assemblage composition (i.e. species cover data) varied significantly across HFCs, as shown by ANOSIM. Most pairwise comparisons were significant, except for comparisons between HFCs 1–3, 2–3 and 3–4. However, it should be noted that the Global *R* value for this analysis was low (0.106) and therefore the results must be interpreted with caution. Ordination of species cover data showed considerable variation and overlap in assemblage composition within HFCs (Figure 7.8).

Aquatic vegetation metrics also varied significantly across HFCs. Vegetation metrics showed consistent patterns across HFCs. In general, total richness and total cover were significantly higher in HFC 1 when compared with HFC 5. However, this pattern also matches to some extent the variation in channel morphology across HFCs.

It is therefore difficult to attribute significant differences in vegetation metrics across HFCs to the nature of the flow regime alone. Regression random forests models for vegetation metrics also showed that D50, riparian canopy cover and land use (particularly PDA and PIA) were more important in describing variation in vegetation metrics than hydrologic variables.

Interpretation of the contribution of the flow regime to the differences observed in vegetation assemblage and metric data across HFCs is dependent upon whether other environmental differences unrelated to hydrology occurred across the HFCs. ANOSIM showed that environmental differences unrelated to hydrology did occur across the HFCs. Only HFC 4 was not significantly different from HFC 5 in terms of environmental conditions (Table 7.6). Furthermore, significant differences occurred in the ratio of bankfull width:bankfull depth for HFCs 2–3 and 3–4, which coincided with differences in aquatic vegetation assemblages determined by ANOSIM (see above). Based on environmental differences it is unlikely that the flow regime was a primary cause of the aquatic vegetation differences detected across the HFCs.

Support for the concept that differences in aquatic vegetation across HFCs are due primarily to environmental differences not related to the flow regime comes from the random forests models testing the capacity of species presence–absence and composition data to predict HFCs. Presence–absence and species composition data had low capacity to allocate sites to the correct HFC, although HFC 3 and 5 had relatively low errors of prediction (21.7% and 23.1% respectively).

Much of the error with the random forests models was associated with HFC 2 (60% error) and HFC 4 (93% error). This shows that the predictive capacity of species composition data to predict HFC varied considerably. Many of the samples from HFCs 2 and 4 that were classified erroneously by the random forests models were allocated to HFC 3. HFCs 2–4 occupy an intermediate position on flow metric gradients (with the exception of CVDaily – Chapter 3), suggesting some hydrologic similarities.

It is interesting to note that samples belonging to HFC 3 that were classified erroneously by the random forests models were not (in most cases) allocated to HFCs 2 or 4 (only two samples in HFC 3 were allocated to HFC 4 – Table 7.6). A possible explanation for this is that the aquatic floras of HFCs 2 and 4 are subsets (or nested) within HFC 3, but HFCs 2 and 4 have higher median values for total richness (plots not shown). An alternative explanation may be the greater number of samples in HFC 3 compared with HFCs 2 and 4 (Table 7.8).

Several factors may have obscured assemblage-scale patterns across HFCs. Firstly the in-stream species recorded in the study area vary in their affinity for aquatic habitats and can be expected to vary in their response to inundation and exposure. Few species can be considered truly aquatic, that is requiring water for successful completion of life cycles. A small proportion of taxa recorded in the study area were classified as TDR (Terrestrial – Dry Places) or TDA (Terrestrial – Damp Places).

These taxa were included in multivariate analyses. While these taxa may be simply indicative of human activity, we considered these species may have had indicator value if flow regime change had allowed the colonisation of near stream habitats by these species. Non-aquatic taxa may indicate anthropogenic disturbances (grazing, roads) but also could have been associated with flow regime change.

Secondly, the HFCs may not have differed sufficiently in flow regime characteristics to generate differences in assemblage composition across these classes. Kennard et al. (2010a) identified 12 flow regime classes across Australia. While it is difficult to align our flow classes with the continental flow classes, 11 study reaches were aligned with continental flow classes 7 and 11 (Chapter 3 – four continental flow classes occurred in the SEQ study area). This suggests that flow regime variability across our study area was low relative to continental-scale flow variability.

Finally, the Historic flow regime classification (Chapter 3) may have excluded flow regime metrics relevant to aquatic vegetation. The Historic flow classification is a hydrologic (statistical) classification where redundant variables (if they occur) are removed. For instance, CVDaily was excluded from the Historic flow classification as it had low loadings on PCA axes. However, the aquatic vegetation conceptual model has CVDaily as a key predictor of vegetation structure. This raises the issue of whether a flow classification should be a statistical classification, where the classification follows statistical criteria, or an ecological classification, where potentially (statistically) redundant variables are included because they have ecological relevance.

Summary

Hypothesis 1 is rejected. Although assemblage composition and vegetation metrics varied across HFCs, it was unlikely that the flow regime itself was the primary driver of these patterns. Patterns in vegetation metrics across HFCs were more apparent but were similar to the gradient in channel morphology (bankfull width:bankfull depth) and latitudinal gradients that occurred across HFCs.

7.4.2 Hypothesis 2

Hypothesis 2 predicted that aquatic vegetation cover would vary inversely with discharge magnitude. The discharge required to mobilise the median particle size (Q_D50MOVE) was chosen as a measure of discharge magnitude potentially relevant to aquatic vegetation, since substrate mobilisation is a key mechanism of biomass loss during high flow events (Riis and Biggs 2003). GLS regression showed a significant positive relationship between total in-stream vegetation cover and the log of Q_D50MOVE. This indicates that total cover is higher where the discharge required for substrate mobilisation is high. Thus total cover is higher in streams where the likelihood of the median particle size being mobilised is low, since it can be assumed that larger discharge events are less likely than smaller discharge events. Note that this does not indicate that sandy streams are less stable than rocky streams in terms of discharge stability. Stability can be considered a function of particle size, as well as the probability of the occurrence of the discharge required to mobilise that particle size. Sandy streams could be considered 'stable' if the discharge required to mobilise the substrate occurs infrequently (e.g. if the interspate length is sufficiently high).

Summary

Hypothesis 2 is accepted. However, the nature of the relationship between total cover and Q_D50MOVE is positive, suggesting aquatic vegetation cover is higher in streams where the probability of occurrence of the discharge required to mobilise the median particle size occurring is low.

7.4.3 Hypothesis 3

Hypothesis 3 predicted that aquatic vegetation abundance would vary inversely with flood frequency. This hypothesis was based on the work of Riis and Biggs (2003), who found that aquatic vegetation in New Zealand streams was limited by flood frequency. Several metrics were trialled as measures of flood frequency – the number of high spells (HSNum), the frequency with which the discharge required to mobilise the median particle size occurred (FD50MOVE), the number of days since the discharge required to mobilise the median particle size occurred (DAYS_Q_D50), and days since last flood (DAYFLOOD). These metrics represent simple counts of flood frequency and mechanisms by which floods impact upon aquatic macrophytes (through sediment mobilisation).

Regression modelling identified significant relationships between vegetation and flood frequency metrics. FD50MOVE was a significant predictor in four models, whereas HSNum was a significant predictor in one model. FD50MOVE is a mechanistic descriptor of flood frequency since it is known that biomass loss during flooding is more likely to be from substrate mobilisation and uprooting, rather than through shedding of biomass (Riis and Biggs 2003). Regression models showed that vegetation occurred in samples where up to 16 floods occurred in the 12 months prior to sampling, whereas Riis and Biggs (2003) estimated that macrophytes were limited to New Zealand streams with 13 or less floods per year.

Summary

Hypothesis 3 is accepted. Aquatic vegetation abundance varied inversely with flood frequency. A measure of flood frequency related to mobilisation of the median particle size (FD50MOVE) was a better measure of flood frequency than the number of high spells (HSNum).

7.4.4 Hypothesis 4

Hypothesis 4 predicted that aquatic vegetation abundance would be positively correlated with discharge variability. This was based on earlier work in the Mary River catchment (Appendix 2). Regression modelling showed that while there was a trend for aquatic vegetation abundance to increase with the coefficient of variation in mean daily discharge, this relationship was not significant (Table 7.13).

The HFCs that represent the extremes of the CVDaily gradient were HFC 1 (mostly flow regulated sites with low CVDaily) and HFC 2 (high CVDaily, located in drier parts of the study area). Flow regulation (where discharge variability is reduced) is thought to promote aquatic vegetation growth since disturbance is reduced (Rørslett et al. 1989; French and Chambers 1997; Howell and Benson 2000; Arthington et al. 2000). This could override any relationships between CVDaily and total cover evident in unregulated sites. However, re-analysis of the relationship with HFC 1 samples improved the model fit but the relationship was not significant (analysis not shown).

Variable discharge regimes, such as the discharge regime of HFC 2, should favour vegetation taxa with traits that enhance survival or persistence in such environments. The flow regime of HFC 2 is characterised by long duration of high spells, long periods of zero discharge and low discharge constancy (Chapter 3).

Aquatic vegetation dominating this flow class may be expected to be tolerant of desiccation and exposure, either through establishment of seed banks or morphologic variation. In particular, amphibious taxa should be especially dominant. Most Brock and Casanova (1997) functional groups occurred with insufficient frequency to draw conclusions about the association of these traits with particular HFCs. However, ARF (Amphibious Fluctuation-Responders – floating/stranded) were especially dominant in HFC 2 and ARP (Amphibious Fluctuation-Responders – morphologically plastic) were rare in HFC 1 (Figure 7.6).

Summary

The hypothesis that aquatic vegetation abundance would be positively correlated with discharge variability is rejected.

7.4.5 Hypothesis 5

Hypothesis 5 predicted that aquatic vegetation abundance would be higher in sites subjected to flow regime alteration than unregulated sites, if flow regulation resulted in increased discharge stability or reduced frequency of substrate mobilisation. This hypothesis was not supported by the data but changes in vegetation structure associated with flow regulation occurred in the study area.

Bankfull substrate stability was significantly higher in unregulated than regulated sites. However, the median value for bankfull substrate stability in regulated sites was greater than 1, indicating that bankfull shear stresses in regulated sites were sufficient to mobilise the median particle size. This implies that within the study area, substrate mobilisation would be expected to cause aquatic vegetation biomass loss in regulated sites. Accordingly, total in-stream vegetation cover (TOTCOV) did not differ significantly between regulated and unregulated sites. This is in line with the Hypothesis, since substrate stability has not been reduced enough by flow regulation to prevent substrate mobilisation during floods.

PLS regression modelling detected an effect of flow regulation which caused a reduction in TOTCOV at most sites. Since TOTCOV did not differ significantly between regulated and unregulated sites and the sites in Figure 7.14 do not deviate substantially from natural (as shown by the 1:1 line), it is expected that the effect of flow regulation on TOTCOV is relatively minor. This follows the work of Mackay (2007) who found a small effect of flow regulation on submerged macrophytes in the Brisbane River downstream of Wivenhoe Dam.

The likelihood of detecting substantial changes in aquatic vegetation related to flow regime change was limited by its lack of severity in the study area (Chapter 3). It was found that stream gauges could be allocated to their correct RFC by a random forests model, even for those gauges affected by flow regime alteration. Furthermore, most flow metrics have only changed by 20% or less under the Historic flow regime compared with the Reference flow regime (Chapter 3).

Further evidence for an influence of flow regulation on aquatic vegetation was provided by ANOSIM comparing vegetation structure at regulated and unregulated sites within individual RFCs. This analysis suggests that flow regulation is associated with a divergence in assemblage structure of sites in RFCs 1 and 5. In this respect, results for the study area are consistent with predictions of the ELOHA framework (Poff et al. 2010).

Summary

Flow regulation was associated with changes in aquatic vegetation cover in sites influenced by flow regulation. Hypothesis 5 could not be disproved since the bankfull substrate stability of flow regulated sites, while lower than unregulated sites, was still high enough to ensure substrate mobilisation and hence periodic biomass removal. The effect of flow regulation in the study area was to cause a reduction in total vegetation cover.

7.4.6 Hypothesis 6

Hypothesis 6 predicted that increasing degree of flow alteration from baseline condition will produce increasing degree of change in aquatic vegetation assemblages. This hypothesis was difficult to test as sites were surveyed downstream of six different dams in the study area. The hypothesis was tested by correlating the effect of flow regime alteration calculated from PLS models with the Gower metric, a measure of overall change between the Reference flow regime and Historic flow regime. The effect for TOTCOV was significantly correlated with the Gower metric but in contrast to the predictions of the ELOHA framework, the effect of flow regulation on TOTCOV decreased with increasing flow regime change from natural.

This may be due to the fact that flow regime changes downstream of dams in the study area vary from dam to dam (Figure 3.15). For example, discharge constancy (Colwell 1974) has increased downstream of Moogerah and Dams, but has decreased downstream of Six Mile Creek and Baroon Pocket Dams.

Poff et al. (2007) suggested that dams homogenise faunas through changes in flow variability. It is clear from the scatterplots in Figure 7.14 that the effects of flow regulation vary from site to site within a given reach. Thus dams are not necessarily homogenising aquatic floras but; rather, the effects of flow regime changes are dependent upon within-site habitat features such as riparian canopy cover and substrate size. This implies that the effects of environmental flows on aquatic vegetation would vary substantially within a reach.

Summary

Hypothesis 6 is not accepted. While the magnitude of flow regime change was correlated with the magnitude of change in total vegetation cover, this relationship was negative (i.e. greater flow regime changes were associated with small change in total cover). The effect of flow regime change varies between sites within a given reach.

7.4.7 Implications

Application of the ELOHA framework to aquatic vegetation in SEQ has shown that key concepts of the framework (i.e. streams with different flow regimes will have different aquatic floras, and increasing flow regime divergence from natural will be associated with increasing biotic change) were not fully substantiated. However, several aquatic vegetation metrics were found to be significantly related to hydrologic metrics, providing mechanisms for managing stream flows in relation to the flow requirements of aquatic vegetation.

Although aquatic vegetation composition and aquatic vegetation metrics varied across HFCs, it was unlikely that the flow regime itself was the primary driver of these patterns. Patterns in vegetation metrics across HFCs were similar to the gradients in channel morphology (bankfull width:bankfull depth) and latitude that occurred across HFCs.

The study has shown that hydrologic metrics are, in general, poor direct predictors of vegetation assemblage structure. This should not be construed to imply that the flow regime is unimportant to aquatic vegetation. The flow regime is critical for at least two reasons. Firstly, this study and others (Riis and Biggs 2003; Mackay 2007) have demonstrated the importance of substrate stability as a driver of vegetation assemblage patterns.

Substrate stability is a function of several factors, particularly stream bed particle size and shear stress. While shear stress is not directly determined by hydrology, two of the parameters in the equation (hydraulic radius and slope) are expected to vary with varying discharge (Gordon et al. 2005). Thus the flow regime is an important driver of aquatic vegetation structure through well-established mechanisms of action on hydraulic habitat characteristics and their stability, as predicted in the first hydro-ecological principle of Bunn and Arthington (2002) and the aquatic vegetation conceptual model (Appendix 2). Secondly, the variable presence of water is vital as the medium that supports submerged vegetation, hence the importance of water depth for a range of species and functional groups.

Catchment characteristics and land use variables explained 23% of variation in aquatic vegetation structure. Amongst these, riparian cover and adjacent land use (irrigated agriculture, dryland agriculture and plantations) accounted for some of the spatial variation in aquatic vegetation assemblages. These factors influence aquatic vegetation through the resource axis of the aquatic vegetation conceptual model (Appendix 2).

The implications of these findings for stream managers and water planners are that:

1. waterway management requires management of the catchment, stream channel and riparian zone as a whole
2. the effects of flow regime change by dams and other factors need to be interpreted in terms of stream hydraulics and habitat structure.

The results of this study also have implications for environmental flow development and the monitoring and management of steam ecosystem health. For in-stream aquatic vegetation, environmental flows need to be estimated in relation to their hydraulic consequences. For instance, environmental flows to maintain vegetation assemblages should be based on natural patterns and frequencies of substrate mobilisation. The frequency of occurrence of high flows that mobilise stream substrates should be maintained to support aquatic plant diversity and cover. Loss of high flows capable of mobilising substrates could have impacts on in-stream vegetation.

The ecological health of streams is a function of its biodiversity and aquatic vegetation is an important component of that diversity, as habitat and refuge for aquatic biota (Pusey et al. 2004; Raynor et al. 2008) and as spawning sites for fish of conservation significance (Kemp 1984; Humphries 1995; Rea et al. 2002). This study has demonstrated that aquatic vegetation assemblages respond to natural landscape gradients (climate and geomorphology), catchment land use, the condition of the riparian zone, stream channel characteristics, and components of the flow regime.

Given the likelihood of further flow regime change by water resource development and climate change, flow regulation impacts need to be incorporated into the monitoring of stream ecosystem health. Aquatic vegetation should be used as indicators of flow regime change, as well as being indicators of broader catchment, riparian and channel health.

This study has demonstrated relationships between flows and simple aquatic vegetation metrics (i.e. total richness and total cover). These metrics are easily estimated in the field without requiring detailed knowledge of aquatic vegetation taxonomy, since they are based on counts and estimates of substrate coverage. Aquatic vegetation can therefore provide an easily applied and useful indicator of a wide range of pressures on catchments and streams, including flow regulation.

7.5 Attachments

Attachment 7.1: Details of sites surveyed for aquatic vegetation

Site number and name	RFC	HFC	Site length (m)	Habitat type	Distance to gauge	No. of surveys
1. Stanley River at Cove Road	5	5	80	Pool-Run	15.0 km	3
2. Burnett Creek downstream of gauge 145018a	No class	3	80	Riffle	4.4 km	4
3. Burnett Creek upstream of gauge 145018a	No class	3	100	Riffle-Pool	0.7 km	4
4. Nerang River at Grand Manor Golf Course	5	3	90	Pool-Run	3.3 km	3
5. Coomera River at Coomera Scouts Hall	1	4	100	Riffle	4.4 km	4
6. Nerang River at Weber Court	5	3	100	Pool	1.0 km	4
7. Teviot Brook near Brennan Road[1]	No class	3	100	Riffle	4.8 km	3
8. Teviot Brook at Croftby	No class	3	100	Pool	1.7 km	4
9. Amamoor Creek at Harrys Creek Road[2]	1	4	100[1]	Pool	7.1 km	4
10. Yabba Creek at Stirling Crossing	2	2	100	Riffle-Run	8.9 km	4
11. Yabba Creek at No. 8 Crossing	2	2	100	Pool	1.3 km	4
12. Obi Obi Creek downstream of number 2 crossing	1	3	45	Pool	2.0 km	3
13. Obi Obi Creek upstream of number 2 crossing	1	3	100	Riffle	4.1 km	3
14. Mary River downstream of Walker Road Bridge	1	3	55	Pool	3.1 km	3
15. Six Mile Creek at Old Noosa Road	1	4	50	Pool	0.7 km	3
16. Six Mile Creek at Grahams Road	1	4	40	Pool	7.5 km	3
17. Glastonbury Creek at Greendale Road Crossing	2	3	100	Pool	11.4 km	4
18. Eudlo Creek at gauge site	5	5	40	Pool	30 m	4
19. Eudlo Creek upstream of Bruce Highway	5	5	55	Run-Pool	4.5 km	4
20. Reynolds Creek at Yarramalong camp ground	2	1	100	Riffle	2.5 km	4
21. Reynolds Creek at downstream of Purdons Bridge	2	1	50	Pool-Riffle	4.7 km	4
22. Amamoor Creek at Zachariah Lane	1	4	100	Riffle-Run	0.2 km	4
23. Glastonbury Creek at 2 km from Mary River confluence	2	3	100	Run	6.2 km	4
24. Mary River at Moy Pocket (north of quarry)	1	3	100	Pool-Run	0.9 km	3
25. Coomera River at Tucker Lane	1	4	100	Riffle-Pool	8.0 km	4
26. Stanley River at gauge 143303a	5	5	100	Pool-Run	0.1 km	3
27. Burnett Creek at 2 km downstream of Maroon Dam	2	1	100	Riffle	2.0 km	4
28. Burnett Creek at Splityard Creek Road	2	1	50	Pool	3.5 km	4
29. Currumbin Creek at Currumbin Valley Primary School	5	5	100	Riffle-Pool	6.5 km	3
30. Currumbin Creek at Fordyce Court	5	5	100	Riffle-Pool	2.7 km	3
31. Wide Bay Creek downstream of gauge 138002c	4	2	100	Riffle-Run	1.8 km	2
32. Wide Bay Creek upstream of gauge 138002c	4	2	60	Pool	0.5 km	2
33. Munna Creek at gauge 138004b	4	2	100	Riffle	0.02 km	2
34. Munna Creek downstream of gauge 138004b	4	2	90	Pool	1.2 km	2
35. North Maroochy River at Eumundi	5	5	60	Pool	4.0 km	4
36. North Maroochy River at North Arm-Yandina Creek Road	5	5	100	Run	4.2 km	4
37. Mary River at Bauple-Woolooga Road	3	3	80	Riffle-Run	14.8 km	2
38. Mary River at Orphants Road	3	3	90	Riffle-Run	10.0 km	2
41. Logan River at Running Creek Road	2	1	100	Riffle-Run	0.8 km	3
42. Logan River at upstream Tilleys Bridge	2	1	65	Pool	3.3 km	3

[1] This site was dry for the first survey.

[2] Site length reduced to 30 m after flooding prior to the fourth survey made much of the site too deep to sample.

Attachment 7.2: List of plant species collected

Alien taxa indicated by an asterisk (*). '?' indicates uncertain identification.

Class	Family	Species name	Common name	Growth form	Functional Group	Acronym
Bryophyta	Amblystegiacaeae	*Cratoneuropsis* spp?		SUB	SUB	Bryo
	Brachytheciaceae	*Rhynchostegium tenuifolium* (Hedw.) Reichardt?		SUB	SUB	Bryo
	Fissidentaceae	*Fissidens berteroi* (Mont.) Müll.Hal.		SUB	SUB	Bryo
		Fissidens (*linearis*?)		SUB	SUB	Bryo
		Fissidens (*pungens*?) Müll.Hal. & Hampe		SUB	SUB	Bryo
	Hypnaceae	*Taxiphyllum taxirameum* (Mitt.) M.Fleisch.		SUB	SUB	Bryo
	Hypopterygiaceae	*Hypopterygium tamarisci* (Sw.) Brid. ex Müll. Hal.		SUB	SUB	Bryo
	Ricciaceae	*Riccia* sp.	Liverwort	SUB	SUB	Riccia
Pteridophyta	Marsileaceae	*Marsilea* sp.	Marsilea, Nardoo	EM	ARP	Marsilea
	Thelypteridaceae	*Christella dentata* (Forssk.) Brownsey & Jermy	Binung	EM	TDA	Chri.dent
Angiospermae	Amaranthaceae	*Alternanthera philoxeroides* (Mart.) Griseb.*	Alligator weed	EM	TDA	Alte.phil
	Apiaceae	*Centella asiatica* (L.) Urb.	Indian Pennywort	EM	ATI	Cent.asia
		Hydrocotyle peduncularis R.Br. ex A.Rich?		EM	ATI	Hydrocot
		Hydrocotyle tripartita R.Br. ex. A.Rich	Pennywort	EM	ATI	Hydrocot
		Hydrocotyle verticillata Thunb.	Shield Pennywort	EM	ATI	Hydrocot
	Aponogetonaceae	*Aponogeton* sp.	Aponogeton	FA	SUB	Apono
	Araceae	*Alocasia brisbanensis* (F.M. Bailey) Domin	Cunjevoi, Spoon Lily	EM	ATE	Aloc.bris
		Colocasia esculenta (L.) Schott*	Taro	EM	ATE	Colo.escu
		Xanthosoma violaceum Schott*	Blue Taro	EM	ATE	Xant.viol
	Asteraceae	*Ageratina adenophora* (Spreng.) R.M.King & H.Rob.*	Crofton weed	EM	TDA	Ager.aden
		Ageratina riparia (Regel) R.M.King & H.Rob*	Mist flower	EM	TDA	Ager.ripa
		Ageratum houstonianum Mill.*	Billygoat Weed	EM	TDR	Ageratum
		Bidens pilosa (L.)*	Cobbler's Pegs	EM	TDR	Bide.pilo
		Conyza spp.*	Fleabane	EM	TDA	Conyza
		Eclipta prostrata (L.) L.	Eclipta	EM	ATE	Ecli.pros
		Sphagneticola trilobata (L.) Pruski*	Singapore Daisy	EM	TDA	Spha.tril
	Brassicaceae	*Cardamine hirsuta* L.*	Flickweed	EM	TDA	Card.hirs
		Rorippa nasturtium aquaticum (L.) Hayek*	Watercress	EM	ATI	Rori.nast
	Ceratophyllaceae	*Ceratophyllum demersum* (L.)	Hornwort	SUB	SUB	Cera.deme
	Cyperaceae	*Bolboschoenus* sp.		EM	ATE	Bolbosch
		Carex appressa R.Br.		EM	ATE	Carex
		Carex gaudichaudiana Kunth		EM	ATE	Carex
		Carex polyantha F.Muell.		EM	ATE	Carex
		Cyperus eragrostis Lam.*		EM	ATE	Cyp.erag
		Cyperus exaltatus Retz.	Giant Sedge	EM	ATE	Cyp.exal
		Cyperus involucratus Rottb.*	Umbrella Sedge	EM	ATE	Cype.invol
		Cyperus polystachyos Rottb. var *polystachyos*	Bunchy Sedge	EM	ATE	Cype.poly
		Cyperus trinervis R.Br.		EM	ARP	Cype.trin
		Eleocharis pallens S.T.Blake	Pale Spike-Sedge	EM	ATE	Eleo.pall
		Isolepis sp.		EM	ARP	Isolepis
		Schoenoplectus validus (Vahl.) A.Love & D.Love	River Clubrush	EM	ATE	Scho.vali
	Euphorbiaceae	*Ricinus communis* L.	Castor Oil Plant	EM	TD	Rici.comm

Attachment 7.2: List of plant species collected (continued)

Class	Family	Species name	Common name	Growth form	Functional Group	Acronym
Angiospermae	Haloragaceae	*Myriophyllum aquaticum* (Vell.) Verdc.*	Parrot's Feather	SUB	ARP	Myri.aqua
		Myriophyllum verrucosum Lindl.	Red Water Milfoil	SUB	ARP	Myrio.
	Hydrocharitaceae	*Egeria densa* Planch.*	Dense Waterweed	SUB	SUB	Eger.dens
		Hydrilla verticillata (L.f.) Royle	Hydrilla	SUB	SUB	Hydr.vert
		Ottelia alismoides (L.) Pers.	Water Lettuce	SUB	SUB	Otte.alis
		Ottelia ovalifolia (R.Br.) Rich.	Swamp Lily	FA	ARP	Otte.oval
		Vallisneria nana R.Br.	Ribbonweed	SUB	SUB	Vall.nana
	Juncaceae	*Juncus prismatocarpus* R.Br.	Rush	EM	ATE	Juncus
	Juncaginaceae	*Triglochin* sp.	Water Ribbons	SUB	ARF	Trigloch
	Laxmanniaceae	*Lomandra hystrix* (R.Br.) L.R.Fraser & Vickery	Mat-rush	EM	ATE	Lomand
		Lomandra longifolia Labill.	Spiny-headed Mat-rush	EM	ATE	Lomand
	Menyanthaceae	*Nymphoides indica* (L.) Kuntze	Water Snowflake	FA	ARF	Nymp.indi
	Nymphaeaceae	*Nymphaea* sp.	A Waterlily	FA	ARF	Nymphaea
	Poaceae	*Axonopus* sp.?	Carpet grass	EM	TDA	Axonopus
		Cynodon dactylodon (L.) Pers	Couch	EM	TDR	Cyn.dact
		Echinochloa telmatophila P.W.Michael & Vickery	Swamp Barnyard Grass	EM	ATE	Echin.telm
		Entolasia stricta (R.Br.) Hughes	Wiry Panic	EM	TDR	Ento.stri
		Oplismenus spp.	Beard grass	EM	TDA	Oplismen
		Paspalidium caespitosum C.E.Hubb.	Brigalow Grass	EM	TDR	Pspa.caes
		Paspalum dilatatum Poir*	Paspalum	EM	TDR	Pasp.dila
		Paspalum distichum L.	Water Couch	EM	ATE	Pasp.dist
		Pennisetum purpureum Schumach.*	Elephant Grass	EM	ATE	Penniset
		Setaria sphacelata (Schumach.) Stapf. & C.E.Hubb.	Sth African Pigeon Grass	EM	TDR	Seta. sphac
		Sorghum halepense (L.) Pers*	Johnson Grass	EM	TDR	Sorg.hale
		Urochloa mutica (Forssk.) Nguyen*	Para Grass	EM	ATE	Uroc.muti
	Philydraceae	*Philydrum lanuginosum* (Banks & Sol.) Gaertn.	Frogmouth	EM	ATE	Phil.lanu
	Polygonaceae	*Persicaria attenuata* (R.Br.) Sojak		EM	ATE	Pers.atte
		Persicaria decipiens (R.Br.) K.L.Wilson	Slender Knotweed	EM	ATE	Pers.deci
		Persicaria hydropiper (L.) Delarbre	Water Pepper	EM	ATE	Pers.hydr
		Persicaria lapathifolia (L.) Delarbre	Pale Knotweed	EM	ATE	Pers.lapa
		Persicaria strigosa (R.Br.) H.Gross	Spotted Knotweed	EM	ATE	Pers.stri
		Rumex brownii Campd.	Swamp Dock	EM	TDA	Rumex
	Potamogetonaceae	*Potamogeton crispus* (L.)	Curly Pondweed	SUB	SUB	Pota.cris
		Potamogeton octandrus Poir. (= *P. javanicus*)	Java pondweed	FA	ARP	Pota.java
		Potamogeton ochreatus Raoul	Blunt Pondweed	SUB	SUB	Peta.ochr
		Potamogeton perfoliatus (L.)	Clasped pondweed	SUB	SUB	Pota.perf
	Scrophulariaceae	*Bacopa monniera* (L.) Pennell	Bacopa	SUB, EM	ATE	Baco. monn

8. Fish

8.1 Introduction

Flow–ecology relationships have been the focus of a multitude of studies on freshwater fish (Arthington et al. 2003; Poff and Zimmerman 2010). In Australia this theme has received significant attention as the states and territories develop management regimes to provide for the environmental flow requirements of riverine and estuarine fish species and communities (Arthington and Pusey 2003). Equally important is to understand and quantify the ecological outcomes of altered flow regimes, to assist water managers in setting rules for water abstraction, and to guide restoration of particular flow characteristics to stressed and over-allocated rivers for ecological benefit.

To support river flow management, the ELOHA framework (Poff et al. 2010) proposes the development of quantitative flow alteration – ecological response relationships determined by empirical measurement along gradients of flow regime alteration, rather than ranking the ecological condition of regulated/supplemented river sites into categories aligned to the severity of their alteration from the natural ('Reference') ecological condition, as in the Benchmarking Methodology developed in Queensland (Brizga et al. 2002).

The ELOHA method assumes that flow is a key determinant of the aquatic ecosystem and its ecological communities and therefore ecological characteristics will be more similar within hydrological classes than between classes (Arthington et al. 2006). Evidence for the influence of flow and various facets of the flow regime (timing, magnitude, duration, rate of change, predictability and variability) on fish assemblage patterns and species is prevalent within the international literature (Poff et al. 1997; Lytle and Poff 2004; Kennard et al. 2007). Four principles linking flow and aquatic biodiversity have widespread support (Bunn and Arthington 2002).

In Australian rivers, flow has been shown to influence fish assemblage diversity and composition, movement, life history processes, recruitment, and productivity (e.g. Arthington et al. 2010; King et al. 2010). The timing of key life history events such as migration and spawning may coincide with specific hydrological events, and flow is the major determinant of patterns of longitudinal and lateral connectivity. Complex relationships with flow may also include the mediation by flow of activities of alien species of fish and plants (Bunn and Arthington 2002).

Although flow is an important determinant of the health of rivers, it cannot be considered in isolation. Fish assemblage patterns are likely to reflect a complex mix of underlying local and landscape drivers and their interactions. Published analyses of fish assemblage and species distribution patterns highlight the importance of broad-scale predictors such as climate, geology, channel structure, habitat and water quality in providing controls on the distribution of fish at a broader landscape scale (e.g. Kennard et al. 2007; Stewart-Koster et al. 2007).

This chapter documents the fish component of the ELOHA field trial in SEQ. It presents the objectives of the study and the major hypotheses tested during the field trial, followed by field, laboratory and statistical methods. The results of statistical analyses are presented and interpreted in relation to the major concepts of the ELOHA framework and the hypotheses tested during the field trial.

Four main themes are presented and interpreted:

- relationships between catchment and in-stream environmental variables and fish assemblage structure
- the importance of flow as an influence on fish assemblages and species
- differences in fish response variables ('indicators') between regulated/supplemented and unregulated sites
- fish responses to flow variability and flow alteration gradients.

The utility and relevance of the field results as guides to water management in SEQ are discussed, and the implications for future research to test, strengthen and refine the ELOHA framework and the flow–ecology relationships are outlined.

8.1.1 Objectives and hypotheses

The central objective of the ELOHA framework is to quantify flow alteration – ecological response relationships for different types of river system classified according to their natural hydrological characteristics (magnitude, timing, frequency, duration and variability).

Specifically, the aims of this chapter are to:

1. identify how existing flow regime alterations in the study area have impacted on habitat structure/heterogeneity, and the structure, dynamics and productivity of riparian vegetation assemblages
2. identify thresholds (if present) or relationships of habitat and ecological response to flow regime alteration
3. identify a limiting suite of flow variables that together governs the condition or 'health' of each river system (or river zone, or set of rivers in a bioregion) and threshold levels of ecological response to flow regime alteration for the whole suite of flow variables
4. assess the relative influence of flow regime alteration versus other pressures on habitat condition and ecological condition.

The field study was designed to address hypotheses built around the ELOHA framework and flow–ecology principles that articulate the influence of flow regimes on aquatic biodiversity (Bunn and Arthington 2002), as discussed in the literature review '*Freshwater Fish-Flow Relationships and Responses to Flow Regime Alteration: a Review of Evidence from south-east Queensland Streams*' (Arthington and Sternberg, 2011, Appendix 3). The following hypotheses are tested in this report:

Hypothesis 1: *The structure and composition of fish assemblages in the SEQ region will be influenced by interactions between flow history, natural catchment characteristics, in-stream habitat factors and anthropogenic disturbances.*

Whilst the antecedent history of stream flows is recognised as one of the principal influences on stream ecology (Poff et al. 1997; Bunn and Arthington 2002), many other catchment characteristics are also important drivers of stream ecological processes and biotic assemblages. Understanding the influences of flow on ecological responses, given the underlying variability in environmental and anthropogenic activities across the study region, presents the first significant challenge of the ELOHA field trial in SEQ.

Previous research in this region has demonstrated the influence of catchment characteristics, in-stream habitat and flow regime history (and their interactions) on fish assemblage structure in the Mary and Albert rivers (Kennard et al. 2007; Stewart-Koster et al. 2007). A test of Hypothesis 1 across a wider range of catchments will either validate previous findings or present differences that relate to the wider geographic scope and greater environmental variability of the catchments included in the present study.

This component of the fish study sets the scene for all subsequent data analysis by quantifying the relative influence of short and long-term flow history on fish distributions (measured by presence–absence data) and fish assemblage composition (measured by CPUE – catch per unit effort).

Following Kennard et al. (2007), it is expected that the distribution of fish (i.e. presence–absence patterns) will be driven largely by landscape variables and long-term patterns of river flow regimes, whereas patterns in fish assemblage composition (relative abundance of species) will be more strongly influenced by in-stream habitat and short-term flow history. The unique explanatory power of flow variables will be revealed by tests of Hypothesis 1, and outcomes will form the basis for subsequent tests of flow-alteration – ecological response relationships.

Hypothesis 2: *The structure and composition of fish assemblages will differ in a range of biological metrics across pre-development (unregulated) and historic (regulated/supplemented) flow regime classes, and between regulated/supplemented and unregulated sites over time.*

This hypothesis tests several basic tenets of the ELOHA framework – that effects of flow regulation on fish assemblages will be apparent but may vary within and among both the RFCs (unregulated) and HFCs (regulated/supplemented). If selections of Reference sites (based on modelled pre-development flow data) for comparison with regulated/ supplemented sites within each pre-development (IQQM) flow class are sound, then differences can be expected between regulated/ supplemented and unregulated fish assemblages within each flow class that are due to flow characteristics alone, not other environmental factors. Furthermore, the fish response among flow classes may vary due to the particular characteristics of the modelled Reference flow regime and the type and degree of flow regime change.

In contrast, if flow alone is the major driver of fish assemblage structure, differential responses to flow regime alteration can be expected across the gauged (Historic) flow classes but similar responses are expected within these flow classes. According to the ELOHA concept, streams that are regulated/supplemented in similar ways should show similar ecological responses, and they should also be more similar ecologically to unregulated streams that fall into the same HFCs. This hypothesis will test these concepts

Assemblage structure and composition can be represented by many metrics reflecting interest in native versus alien species richness, the richness of trait groups, patterns of abundance, and so on. Testing a number of these metrics will serve to highlight the most suitable fish ‘indicators’ for differentiation among flow regime classes and between regulated/supplemented and unregulated sites.

Based on prior knowledge of patterns in fish assemblage structure associated with flow regime change, it was expected that differences in population densities, assemblage-level metrics and assemblage composition between regulated/supplemented and unregulated rivers would depend on the nature of the natural flow regime and the degree and type of flow alteration.

Fish metrics that show significant differences between regulated/ supplemented and unregulated sites will be considered as possible candidates for testing relationships with gradients of flow and flow regime change.

Hypothesis 3: *Fish population and assemblage indicators will vary predictably along gradients of flow variability and change in flow regime characteristics (flow regulation).*

Clear flow – ecological response relationships are needed to form the basis for the ELOHA environmental flow decision-making framework (and every other environmental flow method). These relationships can be applied to predicting and communicating the ecological response to a change in flow regime, both in terms of restoring ecologically important elements of the natural flow regime in regulated/supplemented rivers, and also in predicting the ecological consequences of future changes (arising from new water resource developments, increasing water abstraction/diversion and climate change).

This part of the project set out to visualise and test the strength of the relationships between key flow variables identified as being associated with patterns in fish assemblage composition (Hypothesis 1) and indicators of fish population and assemblage condition across flow regime classes (Hypothesis 2) in SEQ.

Following on from the tests of Hypotheses 1 and 2, it is predicted that some indicators of fish assemblage structure (e.g. species richness, relative abundance of alien species, abundance of fish in particular guilds) will show responses to alterations in particular flow regime characteristics. It is expected that influential flow regime characteristics will be revealed by the multivariate analyses designed to test Hypothesis 1, whereas analyses associated with Hypothesis 2 will suggest the most responsive fish metrics (indicators).

Tests of Hypothesis 3 should reveal the characteristics of flow–fish response relationships, which may be linear, threshold or take some other form, or there may be no clear patterns of response to flow and flow alteration in the study area.

Collectively, test of the three hypotheses will enable the potential of the ELOHA framework to be discussed as it relates to the ecology of fish in SEQ. Furthermore, any constraints and limitations of the framework can be explored in terms of its utility to support water and river health management, and its potential to inform adaptive responses to climate change.

8.2 Methods

8.2.1 Study sites

Forty study sites were selected for sampling of fish assemblages along 20 rivers that reflected the major flow regime gradients in SEQ (Chapter 3). Field sites were defined as a stream length of 100 m (maximum) as this length usually incorporated multiple in-stream habitats (riffles, runs and pools) whilst minimising variation due to changes in stream morphology, geology and neighbouring land use. Sites were preferably located close to a flow gauge or IQQM modelling node (upstream or downstream) to minimise differences between flow measured (or modelling) at the gauge and flow at the actual field sites (Chapter 4).

Two field sites were surveyed within each stream reach. Sites within each reach were located a minimum of 2 km apart to ensure site independence as far as possible, whilst maintaining a reasonable proximity to the flow gauge. Wherever possible the sites selected were not currently grazed, had not been cleared in the last 20–30 years and were not subject to regular burning. These criteria were stipulated in order to reduce land use influences on flow–ecology relationships, particularly influences on riparian vegetation which is likely to be strongly influenced by such factors. It was also important to minimize any direct and indirect influences of land use on in-stream biota and process that are influenced, in turn, by riparian vegetation structure and processes (Pusey and Arthington 2003).

8.2.2 Landscape-scale environmental variables

Landscape variables used in the analysis of spatial patterns in fish assemblages included climate, catchment, reach and site topography, and geological characteristics (Chapter 5).

Climate variables

Reach mean annual temperature, reach hottest month mean temperature and reach coldest month mean temperature variables were acquired from Stein et al. (2009) for individual reaches in which sites were situated. Mean annual rainfall was supplied by the Bureau for Meteorology (2009). Rainfall values were based on a rainfall grid generated using the ANU 3-D Spline. The nearest grid point to each individual study site was used in the analysis.

Topography and morphology

Catchment boundaries for each site were delineated in a GIS using detailed stream networks based on 1:25 000 and 1:100 000 scale maps and a 30 m DEM (digital elevation model) (NASA DTED2 2007). Catchment area was computed using geometry functions of the GIS software. Drainage basin shape was represented by the elongation ratio (Re; the diameter of a circle with the same area as that of the basin divided by the length of the basin). Reach morphology variables, stream gradient (elevation difference for each reach divided by its length) and valley confinement (percentage of stream grid cells and their immediate neighbours that are not defined as valley bottoms) were acquired from Stein et al. (2009).

Site topography variables, distance to source, distance to mouth and elevation were determined for each site using GIS analysis of detailed stream networks based on 1:25 000 and 1:100 000 scale drainage maps and the 30 m DEM. Cross sectional surveys were undertaken at a riffle within each site to determine channel morphology and used in the calculation of various stream hydraulic parameters. Surveys were conducted with a dumpy and staff. This cross section information was used to determine an average bank slope for the channel side surveyed for riparian vegetation. Aspect was measured with a compass in the field or from the 30 m DEM.

Geology

Substrate geological characteristics were derived for the field sites from the SEQ Region Geoscience dataset (Queensland DNRM 2002) and digital 1:100 000 scale geology maps for the region. Geological groupings were based on broad composition characteristics and followed the classes of Stein et al. (2009).

Land use and land management variables

Landscape-scale land use and disturbance for field study catchments were assessed using the Queensland Land Use Mapping Program (QLUMP) and dataset (Witte et al. 2006) generated from baseline surveys conducted in 1999, and draft updates available from 2006 surveys for the Maroochy and Logan-Albert. The primary land use classes were based on the Australian Land use and Management Classification version 6 (BRS 2002) as these represent broad land use categories differentiating conservation and relatively natural land uses from intensive land uses. Percentages of the upstream land use were recorded as production from relatively natural environments (forestry, grazing natural vegetation), dryland agriculture and plantations (e.g. cropping, horticulture, grazing pasture), irrigated agriculture (e.g. irrigated cropping, horticulture), conservation and natural environments (e.g. national park) and intensive uses (e.g. residential and industrial uses). The percentage of each primary land use class within each catchment was calculated as lumped metrics which are non-spatially explicit and treat the whole catchment area with a similar weighting. Land use variables were also calculated using inverse distance weighted metrics (following Petersen et al. 2010), a process that weights each land use according to its proximity to the stream.

Land use based on the QLUMP data provides information at a relatively coarse scale (smallest mapped feature is 1 hectare and minimum width for linear features is 50 m). However, land management practices at a local scale can have a strong impact upon riparian communities and stream ecosystems through extremely localised activities (e.g. selective weed control, riparian replanting, localised riparian grazing and burning) that are unlikely to be reflected in the broader-scale land use datasets available.

Landholder surveys were undertaken in order to ascertain the extent of local management activities specific to riparian and stream study sites and reaches. For sites from which surveys were not returned, the metrics were generated through personal observations and knowledge of the area. Catchment disturbance data were treated as consistent throughout the sampling program as they were only recorded from surveys conducted over a single time period.

8.2.3 Flow regime data

Daily flow data collected at stream gauges was used to characterise the Historic flow regime of each river reach based on two antecedent time periods that reflected differences in the mean longevity of fish species in SEQ. Fish in the study region can be grouped into species that are small-bodied and short-lived (≤~4 years) (e.g. Australian smelt, *Retropinna semoni*) and larger bodied longer lived (>4 years) (e.g. Australian bass, *Macquaria novemaculeata*). Therefore, for each of the three sample times (see below), flow metrics representing conditions during two Historic flow regime periods (4 and 15 years) were calculated to test for the influence of flow on patterns in fish assemblage composition.

The suite of flow metrics calculated for fish–flow analyses has been shown to be correlated with patterns in fish assemblage composition in the study area (Kennard et al. 2007; Stewart-Koster et al. 2007). These flow metrics also distinguish geographic patterns in river flow regimes throughout SEQ (Chapter 3) and broadly reflect patterns in the magnitude, timing, frequency, variability and rate of change in river flows (sensu Poff et al. 1997).

In addition, the flow metrics that distinguish RFCs (Chapter 3) were included to explore their particular influences on fish assemblages. Daily flow data were sourced from the Queensland DNRM and SEQWater (Chapter 3). Flow metrics were calculated predominantly using the RAP (Marsh et al. 2003).

However, flow records for two river reaches included periods of missing data, and therefore flow metrics were estimated using the IHA program (Richter et al.1997; The Nature Conservancy 2009), a statistical package that can handle missing data values. Flow metrics tested in fish–flow analyses are listed in Table 8.1.

Table 8.1: Flow metrics included in multivariate analysis of fish assemblage structure in relation to environmental factors in the study area

Flow metric	Abbreviation	Definition	4 yr	15 yr
Mean daily flow	MDF	Mean of all daily flows	Y	Y
Median daily flow	Med	Median of all daily flows	Y	Y
Minimum daily flow	Min	Minimum of all daily flows	Y	Y
Maximum daily flow	Max	Maximum of all daily flows	Y	Y
Q90	P 10	Discharge exceeded 90% of the time	Y	Y
Q10	P 90	Discharge exceeded 10% of the time	Y	Y
Coefficient of variation of daily flow	CV	Standard deviation divided by the mean for daily flows	Y	Y
High spell number	HSNum	Number of floods greater than median daily flow	Y	Y
Mean rate of rise	MMagRise	Mean difference in daily flow during rising flow events	Y	Y
Mean rate of fall	MRateRise	Mean difference in daily flow during falling flow events	Y	Y
Mean magnitude of rise	MMagFall	Mean magnitude of rise for all flow events	Y	Y
Mean magnitude of fall	MRateFall	Mean magnitude of fall for all flow events	Y	Y
Number of zero flow days per year	Under0.1	Number of zero flow days per year	Y	Y
Baseflow index	BFI	Baseflow Index (ratio of baseflow to total flow)	Y	Y
Mean dailybaseflow	MDBF	Total baseflow component divided by number of days of record	Y	Y
Mean daily flow in January	MDFJanuary	Indicators that distinguish the IQQM classes	Y	Y
Mean daily flow in September	MDFSeptember		Y	Y
Mean daily flow in November	MDFNovember		Y	Y
ARI 1 year	PS1YrARI			Y
ARI 2 years	PS2YrARI			Y
ARI 10 years	PS10YrARI			Y
1 day maximum flow	MA 1daysMaxMean		Y	Y
7 day maximum flow	MA 7daysMaxMean		Y	Y
Mean annual 1 day minimum	MA 1daysMinMean		Y	Y
Mean annual 3 day minimum	MA 3daysMinMean		Y	Y
Mean annual 7 day minimum	MA 7daysMinMean		Y	Y
Predictability of mean monthly flow	P_MDFM			Y
Constancy of mean monthly flow	C_MDFM			Y

8.2.4 Fish sampling and data

Fish were sampled three times at each of 40 sample sites during July–August 2009, October–November 2009 and April–May 2010. These sample times reflect differences in climatic conditions (and therefore river flow regime) throughout SEQ, with June–August typically having cooler air temperatures and lowest rainfall, while September–November is usually a warmer period with increasing rainfall, December–February are the hottest months with greatest rainfall, and March–May are mild months with decreasing rainfall (Bureau of Meteorology 2009). No sampling was undertaken during the hottest and wettest period (December–February) as higher river flows prevented effective fish sampling.

Fish sampling sites were delineated by block netting sites into separate pool, riffle and run habitats (where available), and fish were sampled using multiple pass backpack electrofishing with a Smith–Root LR-24 backpack electrofisher (Figure 8.1; see also Pusey et al. 2004). Electrofisher settings varied according to physical water characteristics, and sampling was done with settings to induce minimal harm to fish.

At sites with areas of deeper open water habitat, seine netting was also undertaken to capture pelagic species that are often under-represented by backpack electrofishing. Fish were placed in a water-filled bucket until all fish were captured from each habitat unit, then identified and standard length measured. All alien species were euthanased according to sampling and ethics permits (Griffith University Animal Ethics Approval No. ENV/21/08/AEC, Queensland DPI permit number PRM00157K). Most native fish were returned to the water alive (apart from some specimens retained to verify identification). Fish sample data was standardised to 450 m^2 reach surface area (combining all habitat units sampled) to allow for meaningful comparisons of richness and relative abundance among sites.

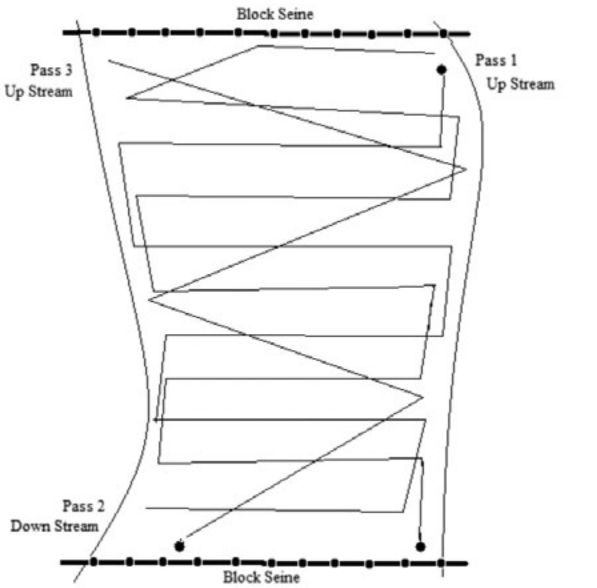
8.2.5 In-stream habitat data

Following fish sampling, in-stream habitat variables describing flow velocity, depth, width, substrate, vegetation and bank condition were recorded from random points in each study reach (Figure 8.2).

The number of random habitat points varied according to stream length sampled for fish, although a minimum of 10 sample points was generally assessed. At each habitat sample point, reach width (m), depth (m), flow velocity (ms^{-1}), substrate composition (estimated as a percentage of mud, sand, fine gravel, coarse gravel, cobble and bedrock) and presence of macrophytes, leaf litter, overhanging/submerged/emergent vegetation, root mass, undercut bank, large woody debris (>15 cm diameter), small woody debris (<15 cm diameter) and filamentous algae were recorded (following Pusey et al. 2004).

For each sample time, in-stream habitat data from each sample point was averaged to derive a single mean value for each habitat variable for each sampling site.

Figure 8.1: Fish sampling design based on multiple pass backpack electrofishing (top diagram) within separate pool, riffle and run habitats (bottom diagram). Diagrams adapted from Pusey et al. (2004).



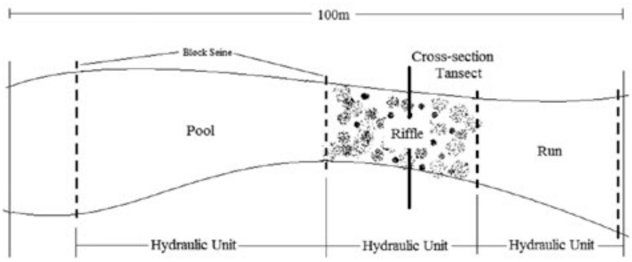
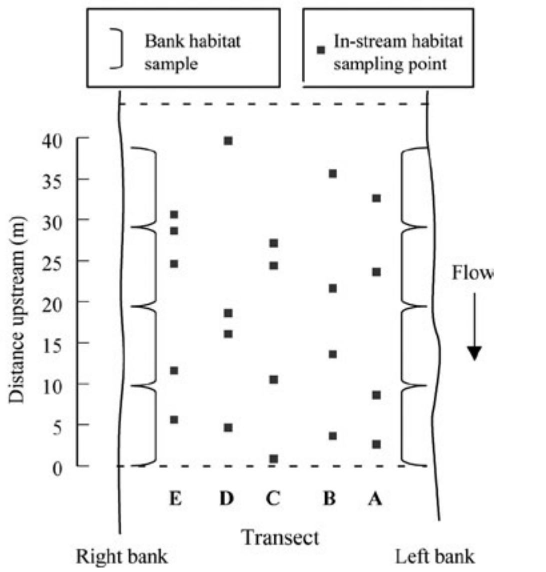


Figure 8.2: Design used to survey in-stream and bank habitat characteristics within each sampling site. Diagram adapted from Pusey et al. (2004).



8.2.6 Data analysis

A number of analyses were used to test for fish responses to environmental and flow gradients in the study area, RFC, flow variability across the study area, and flow regime change associated with dams and regulated/supplemented flows. These analyses are described below.

Predictor variable reduction

Reduction of the number of environmental predictor variables was necessary prior to statistical analysis. Therefore, each of the environmental predictor datasets (catchment land use disturbance, climate, reach scale geomorphology and in-stream habitat) was separately subjected to PCA to derive a reduced set of independent, composite variables that summarise the major patterns of environmental variation in the study area. In each dataset, all variables were normalised to produce a common variance.

Draftsman's plots were used to determine if data required transformation following normalisation due to skewness. In the case of right-skewness, a log (X + 1) transformation was performed to reduce severe skewness and to balance the influence of outliers in each predictor variable dataset.

Results of each PCA were interpreted according to gradients of original input variables that had the two highest loadings for each PCA axis. PCA axes that had eigenvalues >1 were retained for use in the main fish–environmental factor analysis. This resulted in a total of two climate variables, two geography/geology variables, three in-stream habitat variables, three medium term (4-year) and three long-term (15-year) flow variables. One final continuous predictor variable was also used, the composite Gower metric (Gower 1971, Chapter 3) describing alterations in the flow regime of each river reach based on differences between modelled Reference and gauged (Historic) flow data.

The purpose of this inclusion was to seek evidence of fish responses to the gradient of flow regime change caused by dams in the study area.This process resulted in a total of 14 predictor variables; six of which remained constant for each sampling date, whereas the remaining eight (in-stream habitat, medium and long-term flow regime) varied with each sampling time.

Fish–environmental factor analysis

Temporal patterns in fish assemblage composition were tested using a single-factor PERMANOVA (Anderson 2001) to assess whether the entire suite of samples varied significantly through time and to validate the use of separate analyses for each sampling period. To assess the strength of the relationship between environmental variables (i.e. composite environmental PCA axes) and patterns in the composition of fish assemblages (Hypothesis 1), distance-based linear modelling (DISTLM) (Legendre and Anderson 1999) was used in Primer V6 with the PERMANOVA+ add-on package (Anderson et al. 2008). DISTLM is a multiple multivariate regression analysis that uses permutation to test for the strength and significance of relationship between predictor (i.e. environmental) and ecological response variables.

In this study, two questions about the ecological responses of fish were of particular interest: firstly, how the presence or absence of fish species, and secondly, how patterns in the composition of the fish were related to environmental gradients in the study area.

Comparing these two tests can therefore indicate whether patterns in the distribution of fish are driven by the same (or different) environmental variables that influence changes in the composition of fish assemblages, and furthermore, how much of the observed spatial and temporal variation in fish assemblage patterns was related to short and long-term flow history (Hypothesis 1). Patterns in the presence–absence fish data were compared using the Sørensen similarity measure to indicate how the distributions of individual taxa were associated with environmental predictor variables. DISTLM analysis was then undertaken using an Akaike's Information Criterion selection procedure run with 9999 permutations. The BEST selection procedure was employed to model all possible combinations of variables.

The AICc (Akaikes Information Criterion adjusted for small sample size) was used as this first analysis was primarily exploratory in nature. Patterns in the composition of fish assemblages (i.e. including both presence–absence and abundance information) were compared using the modified Gower (log base 2) similarity measure. The modified Gower measure has the advantage over the Bray–Curtis metric for abundance data as the relative weight of change in the abundance of taxa can be specified according to the patterns of interest in a study (Anderson et al. 2006). A log base 2 function values the doubling of a species abundance to be weighted the same as a change in species composition (i.e. based on presence–absence data) (Anderson et al. 2006).

These analyses should reveal how much of the observed spatial and temporal variation in fish assemblage patterns was related to short and long-term flow history versus other environmental factors that are also known to influence fish assemblages in SEQ (e.g. Kennard et al. 2007).

Fish indicator analysis

Comparisons of indicator sensitivity to disturbance are valuable for identifying appropriate indicators and metrics to assess response and recovery to impacts (e.g. Benejam et al. 2010) and the time scales of each response. This study examined indicators that can be determined with little basic ecological information to allow for comparisons with other local studies, and to inform similar studies in the many regions worldwide where the fish fauna has been poorly studied and easily measured indicators such as species richness and relative abundance would be useful.

Indicators were grouped into metrics of univariate population abundances and both univariate and multivariate measures of fish assemblage structure (Table 8.2). For population level analyses the fauna was grouped into species present in greater than 75, 50 and 25 % of samples in the entire dataset. Assemblage level indicators were grouped into univariate measures indicating patterns of richness (e.g. total species, native species and alien species richness), assemblage abundance (e.g. total fish abundance) and composition of fish assemblages (similarity in presence–absence patterns).

Patterns in each indicator (response variable) were compared across categorical factors (flow regime class: groups 1–5, Chapter 3), regulated/supplemented versus unregulated reaches and three sample times, creating a three-factor statistical design. Patterns in fish assemblage metrics were compared across both RFCs (modelled natural) and HFCs (gauged)to test if fish assemblage indicators better reflected long-term Historic or recent Historic flow regimes.

According to the ELOHA framework, fish population and assemblage indicators will reflect differences associated with flow regime classes, and these indicators will show differences between regulated/supplemented and unregulated reaches dependent on the particular flow class. Flow regime class (Class) and flow regulation (Regulation) were designated as fixed factors, whereas the factor time (Time) was considered a random factor. Replicate sample sites (n=2) in each river reach were considered random. In this analysis, the interaction between flow class and regulation is of prime interest. Differences in fish indicators among rivers within each flow class were not tested, as some flow classes were only represented by a single river.

Table 8.2: Indicators and metrics used to test fish species and assemblage level differences between flow regime classes and between regulated/supplemented and unregulated sites

Fish density = number per 450m^{-2}, multivariate assemblage composition = modified Gower Log base 2.

Type	Definition	Code
Species >75% FOC	Density of long-finned eel	AngRei
	Density of Duboulays rainbowfish	MelDub
	Density of Australian smelt	RetSem
Species >50% FOC	Density of freshwater catfish	TanTan
	Density of western carp gudgeon	HypKlu
	Density of firetail gudgeon	HypGal
Species >25% FOC	Density of Pacific blue-eye	PseSig
	Density of gambusia	GamHol
	Density of Midgleys carp gudgeon	HypSp1
	Density of striped gudgeon	GobAus
Assemblage richness	Number of alien species at each sampling site	AlienRichness
	Number of species at each sampling site	SPR
	Number of native species at each sampling site	NativeRich
	Number of non-migratory species at each sampling site	NonMigRich
	Proportion of all species that are native	PropNatRich
Assemblage density	Density of all alien species at each sampling site	AlienDensity
	Density of all species at each sampling site	TotDensity
	Density of all native species at each sampling site	NatDensity
	Density of non-migratory species at each sampling site	NonMigDensity
	Proportion of individuals that are native	PropNatDens
Assemblage composition	Ratio of the number of species: total individuals in a sample	SPDensity
	Fish assemblage using presence–absence data, Sorensen index	Comp-PA
	Fish assemblage including abundance data	Comp-CPUE
	Fish assemblage excluding taxa present in <25% of samples	Comp-widespread
	Fish assemblage excluding taxa accounting for <1% total abundance	Comp-abundant
	Composition of non-migratory species	Comp-nonmig
	Composition of native species	Comp-native
	Composition of fish species that live for <4 years	Comp-shortliv

In the context of the ELOHA framework, the process for selecting potentially suitable indicators was that a particular indicator would firstly show a significant difference among RFCs (modelled natural) and between regulated/supplemented and unregulated sites within each RFC. Secondly, the indicator would show significant differences among classes reflecting Historic flow regimes, yet would not differ between regulated/supplemented and unregulated sites within each class (Hypothesis 2, above).

Regulated/supplemented and unregulated sites were sampled in RFCs 1, 2 and 5, and HFCs 1, 2, 3 and 4 out of a total of five flow classes of each type (Reference and Historic flows). Univariate data were subjected to Permutational (Multivariate) Analysis of Variance (Anderson 2001) and compared using Euclidean distance, whereas multivariate data were compared using the Sorensen index (presence–absence data) or modified Gower (log base 2) for abundance data (Anderson et al. 2006). Differences were considered significant at a=0.05. Variance components were calculated for all factors and interaction terms to assist in the interpretation of differences in the context of effect sizes. All data were analysed using the PERMANOVA+ package in PRIMER v6 (Anderson et al. 2008).

Fish response to flow regime gradients

This part of the project was designed to visualise and test the strength of the relationships between key flow variables identified as being associated with patterns in fish assemblage composition (Hypothesis 1), and indicators of fish population and assemblage structure across flow regime classes (Hypothesis 2) in the study area. These analyses addressed fish responses to **flow variability** across the study area (including all sites, all times, and both regulated and unregulated flows), and responses to **flow regime change** (based on comparison of Reference and Historic flow data).

Fish response to flow variability

Tests of Hypothesis 2 identified that Duboulays rainbowfish (*Melanotaenia duboulayi*) was the most likely indicator species to reflect changes in flow regime in the study area, as densities varied significantly between regulated and unregulated sites within RFCs, yet did not differ significantly between regulated/supplemented and unregulated sites within HFCs. No other species showed this pattern of response to flow regulation (supplementation). However, because of their widespread distribution in the dataset, densities of long-finned eel (*Anguilla reinhardtii*) were also compared against selected flow metrics. Despite no univariate assemblage metrics showing significant potential as indicators of flow regime change according to the criteria set out above (see Fish Indicator Analysis), the number of native species (native species richness) was also tested against selected flow metrics.

Mean densities of Duboulays rainbowfish (*Melanotaenia duboulayi*), long-finned eel (*Anguilla reinhardtii*) and native species richness were plotted against the raw hydrological data for each site for the preceding 4-year period. Because fish indicators showed little temporal variation, samples were averaged across the three sample times and the two sample sites on each river reach. To test the significance of these relationships, GLS regression was applied using nlme in the R using package (Pinheiro et al. 2011).

A second set of analyses was run to further explore flow–ecology relationships using GLS regression, but this time the fish metrics were not averaged across the three sample times and the two sample sites on each river reach. A GLS regression was fitted for each relationship and if the model was significant (p<0.05) the model fit was plotted. Residuals were examined to look for outliers and violations of model concepts, especially heteroscedastic or non-normal errors. Where significant models were identified, log-likelihood ratio tests were used to ensure that the model was an improvement over the null model. The log-likelihood ratio statistic *D* was calculated from the equation 2x(log likelihood null model-log likelihood alternate model), where the alternate model is the univariate GLS model. The significance of *D* was compared against a X^2 distribution with one degree of freedom (Bolker 2008). GLS and GNLS models were fitted using the *gls* and *gnls* functions in the nlme package for R (Pinheiro 2011).

This set of analyses modelled several of the most abundance and common fish species as well as univariate measures of fish assemblage structure (Table 8.2), such as species richness, species density and alien species density against flow metrics revealed to influence fish in the multivariate tests of Hypothesis 1. Two sets of flow metrics were tested – those describing 4-year antecedent flow periods (number of zero flow days, mean daily flow and CV of daily flow), and those describing 15-year antecedent flow periods (1-year ARI, high spell number and the constancy and predictability of mean monthly flows (see Table 8.1 for details of these metrics).

Fish response to flow regime change

These analyses tested the significance of relationships between changes in flow regime (i.e. the Gower values describing the composite change in flow between Reference and Historic flow metrics) and mean densities per site and sampling time of long-finned eels (*Anguilla reinhardtii*), Duboulays rainbowfish (*Melanotaenia duboulayi*) and native species richness. Relationships were tested using the gls function in the nlme package in R (Pinheiro et al. 2011).

Any or all of these regression relationships can be influenced by environmental factors unrelated to flow. To examine the direct influences of flow on fish species and assemblages, PLS modelling was used to quantify the effect of flow regulation (if any) on fish metrics, using the methods of Englund et al. (1997a,b) and Zhang et al. (1998). PLS modelling is similar to principal components analysis (PCA) in that many predictor variables can be summarised into a reduced set of latent components (Eriksson et al. 1995). An advantage of PLS modelling is that it can handle correlated predictor variables and also situations where the number of predictor variables greatly exceeds the number of cases (Eriksson et al. 1995).

Reference PLS models were first developed for fish metrics using environmental parameters not directly influenced by flow regulation as predictors. These models were based on unregulated sites only, with the environmental predictors based on variable importance from the multivariate regression analysis DISTLM (described above). Environmental variables were standardised (zero mean, unit variance) prior to analysis (Jansson et al. 2000). Models were cross-validated using 'leave one out' cross-validation (Wehrens and Mevik 2009) and the appropriate number of components for individual models was determined from a plot of the root mean squared error of prediction versus the number of components.

The Reference model was then used to predict values for fish metrics at regulated sites. The 'effect' of flow regulation was calculated as [(observed value–predicted value)/predicted value] × 100 (Zhang et al. 1998). The effect of flow regulation was considered to be significant if the mean of the effects did not include zero, corresponding to $p<0.05$ (Zhang et al. 1998). PLS models were fitted using the pls package for R (Wehrens and Mevik 2009) and the orthogonal scores algorithm.

Finally, to test the hypothesis that increasing overall flow regime change causes increased divergence of fish assemblage structure from Reference condition (Hypothesis 3), the 'effects' from the PLS models were correlated with the Gower metric values obtained for the comparison of Reference and Historic flow regimes for individual stream gauges (Chapter 3). If increasing flow regime change is associated with increasing divergence of biota from Reference condition, then the flow regulation effect for individual sites should be correlated with the Gower metric. A further set of correlations was run using individual flow metrics in place of the overall Gower gradient of flow regime change.

8.3 Results

8.3.1 Fish fauna of south-east Queensland

Totals of 14,261 individuals comprising 35 species were recorded across the three fish surveys (Table 6.3). Long-finned eel (*Anguilla reinhardtii*), Duboulays rainbowfish (*Melanotaenia duboulayi*) and Australian smelt (*Retropinna semoni*) were collected in >75% of all samples, while freshwater catfish (*Tandanus tandanus*), western carp gudgeon (*Hypseleotris klunzingeri*) and firetailed gudgeon (*Hypseleotris galii*) were collected in >50% of all samples, and Pacific blue-eye (*Pseudomugil signifier*), gambusia (*Gambusia holbrooki*), Midgleys carp gudgeon (*Hypseleotris sp.*) and striped gudgeon (*Gobiomorphus australis*) were collected in >25% of all samples (Table 6.3).

These ten species comprised 84.9% of the total fish in electrofishing collections. The remaining 25 species were infrequently sampled (<25% of all samples), thereby contributing to their low relative abundances in the dataset. Species recorded at high densities per 10 m of stream length were ornate rainbowfish (*Rhadinocentrus ornatus*), smelt, striped gudgeon, the *Hypseleotris* complex, Duboulays rainbowfish, Pacific blue-eye, hardyheads (*Craterocephalus stercusmuscarum, Craterocephalus marjoriae*) and long-finned eels (*Anguilla reinhardtii*).

Five alien species were sampled: common carp (*Cyprinus carpio*), gambusia, tilapia (*Orechromis mossambicus*), swordtail (*Xiphophrus helleri*) and platy (*Xiphophrus maculatus*). However, these species only contributed 3.71% of the total number of fish collected.

Fish assemblages across all sites differed significantly among the three sample times (PERMANOVA F=1.710, P=0.0213), hence their relationships with environmental variables were analysed separately for each time period. By doing so it was possible to observe variation in the factors associated with seasonal variations in fish assemblages. The relative importance of flow variables versus other environmental factors was of particular interest.

8.3.2 Environmental gradients

Reduction of the number of environmental predictor variables was necessary prior to statistical analysis. Therefore, each of the environmental predictor datasets (catchment land use disturbance, climate, reach scale geomorphology and in-stream habitat) was separately subjected to PCA to derive a reduced set of independent, composite variables that summarise the major patterns of environmental variation in the study area. In each dataset, all variables were normalised to produce a common variance. Results of the PCA for streams of SEQ are shown in Table 8.4.

Table 8.3: Species, native/alien status (*alien), age, density per 10 m, frequency of occurrence and proportion of total catch in streams of SEQ over three sampling periods 2009–2010.

Families and common names are given in Appendix 3 to this Scientific Report.

Scientific name	Maximum age (years)	Density Per 10m	Frequency of occurrence	Proportion of total
Ambassis agassizii	3-4	0.16	23.5	1.1
Amniataba percoides	4	0.09	1.7	0.1
Anguilla australis	32	0.04	3.5	<0.1
Anguilla reinhardtii	45	0.32	79.1	10.6
Arrhamphus sclerolepis	7	0.04	0.9	<0.1
Craterocephalus marjoriae	2	0.39	15.7	2.3
Craterocephalus stercusmuscarum	2	0.06	15.7	0.4
Cyprinus carpio*	>15?	0.05	6.1	0.1
Gambusia holbrooki*	2	0.25	30.4	2.4
Glossamia aprion	5	0.10	2.6	0.1
Gobiomorphus australis	5	0.83	25.2	5.7
Gobiomorphus coxii	5	0.18	5.2	0.4
Hypseleotris compressa	3-5	0.38	13.9	1.4
Hypseleotris galii	2-3	0.43	57.4	7.0
Hypseleotris klunzingeri	2-3	0.42	60.0	10.9
Hypseleotris sp.	2-3	0.26	29.6	2.5
Leiopotherapon unicolor	3-5	0.37	16.5	1.3
Maccullochella peelii mariensis	>10	0.01	0.9	<0.1
Macquaria novemaculeata	22	0.05	7.8	0.1
Melanotaenia duboulayi	4	0.51	76.5	11.6
Mogurnda adspera	3	0.24	23.5	1.6
Mugil cephalus	9	0.11	3.5	0.1
Myxus petardi	8?	0.05	1.7	<0.1
Nematolosa erebi	5	0.03	2.6	<0.1
Neosilurus hyrtlii	5	0.05	2.6	<0.1
Notesthes robusta	4?	0.08	5.2	0.1
Orechromis mossambicus*	11	0.12	0.9	<0.1
Philypnodon grandiceps	<3 ?	0.13	24.3	0.9
Philypnodon macrostomas	<3 ?	0.08	20.9	0.6
Pseudomugil signifer	2	0.41	49.6	8.7
Retropinna semoni	2-3	0.76	75.7	25.2
Rhadinocentrus ornatus	3-4	0.83	11.3	1.2
Tandanus tandanus	8-12	0.13	69.6	3.3
Xiphophrus helleri*	2	0.12	1.7	0.1
Xiphorus maculatus*	2	0.07	6.1	0.1
			Total	14 261

Table 8.4: Principal Components from the PCA of catchment landscape, climate, reach, habitat and flow variables for fish sampling sites, showing factors and percent of variation explained

Principal components	Variation explained	Increasing	Decreasing
1. Catchment PC1	33.3%	Production from natural environments	Intensive land use
2. Catchment PC2	22.6%	Natural land	Dryland agriculture
3. Climate PC1	52.3%	Hottest month mean temp.	Mean annual rainfall
4. Climate PC2	44.7%	Mean annual temp, coldest month mean temp.	
5. Reach PC1	20.7%	Catchment area, distance to source	Valley slope and relief ratio
6. Reach PC2	15.6%	Igneous rock, distance to mouth	Sedimentary and unconsolidated rock
7. 4yr flow PC1	65.1-66.3%	Mean daily flow, mean daily baseflow, magnitude of 10th percentile flow	
8. 4yr flow PC2	17.0-18.1%	Baseflow index	CV mean daily flow
9. 4yr flow PC3	5.5-7.0%	Number of zero flow days	Minimum daily flow, number of floods greater than median daily flow
10. 15yr flow PC1	62.0-62.1%	Mean daily flow, magnitude of 1yr ARI	
11. 15yr flow PC2	17.7-18.5%	Baseflow index	CV mean daily flow
12. 15yr flow PC3	7.3-8.6%	Constancy, predictability of mean monthly flow	Number of floods greater than median daily flow
Temporally variable principal components			
Time 1			
1. Habitat PC1	25.20%	Sand, large woody debris	Coarse gravel, cobble
2. Habitat PC2	14%	Fine gravel, stream width	Submerged vegetation, Rock substrate
3. Habitat PC3	10.80%	Filamentous algae, macrophyte cover	Root mass, undercut bank
Time 2			
1. Habitat PC1	23%	Sand, large woody debris	Overhanging vegetation, cobble
2. Habitat PC2	15.90%	Leaf litter, filamentous algae	Site depth, submerged vegetation
3. Habitat PC3	14.40%	Rock, mud	Fine and coarse gravel
Time 3			
1. Habitat PC1	26.60%	Sand, small and large woody debris	Submerged vegetation
2. Habitat PC2	14.80%	Fine gravel, flow velocity	Rock
3. Habitat PC3	10.00%	Stream width	Mud, filamentous algae

Catchment land use gradients

Catchment PC1 accounted for 33.3% of variation in catchment land use (i.e. disturbance variables), describing a gradient of decreasing production from relatively natural environments and increasing intensive land use surrounding fish sampling sites (Table 8.4). Catchment PC2 (22.6% of variation) described a gradient of decreasing land use as conservation and natural environments to increasing dryland agricultural production (PDA). In other words, the study area is characterised by a gradient ranging from areas of natural and conservation land use to areas of intensive land use for agriculture.

Climate gradients

Climate PC1 (52.3% of climatic variation) described a gradient of decreasing hottest mean monthly temperature and increasing mean annual rainfall, while climate PC2 (44.7%) describes a gradient of decreasing mean annual temperature and decreasing coldest monthly mean temperature (Table 8.4). This analysis clarifies the main seasonal patterns of temperature peaks and declines and rainfall variation that may affect fish in the study area.

Reach scale gradients

Reach PC1 accounted for 20.7% of variation in river reach characteristics in the study area, describing a gradient of decreasing catchment area and distance to source (i.e. lowland to upland catchments) and increasing valley slope and relief ratio (Table 8.4). Reach PC2 (15.6%) describes a gradient of decreasing igneous rock, distance to mouth and increasing sedimentary and unconsolidated rock, reflecting position of fish sampling sites along the upstream–downstream gradient of geology and substrate characteristics of study streams.

Temporally variable predictor variables

In-stream habitat

In-stream habitat variables differed significantly over the three sampling occasions and were analysed separately to permit meaningful interpretation of temporal fish responses.

Sample time 1 (July–August 2009):

Habitat PC1 accounted for 25.2% of variation in stream habitat characteristics, describing a gradient of decreasing sand and large woody debris and increasing coarse gravel and cobble substrates. Habitat PC2 (14%) describes a gradient of decreasing fine gravel and stream width and increasing submerged vegetation and rock substrate (Table 8.4). Habitat PC3 (10.8%) describes a gradient of decreasing filamentous algae and macrophyte (aquatic vegetation) cover and increasing root mass and undercut banks.

Sample time 2 (October–December 2009):

Habitat PC1 accounted for 23.0% of variation in stream habitat characteristics, describing a gradient of decreasing sand and large woody debris and increasing overhanging vegetation and cobble substrates (Table 8.4). Habitat PC2 (15.9%) describes a gradient of decreasing leaf-litter and filamentous algae and increasing site depth and submerged vegetation. Habitat PC3 (14.4%) describes a gradient of decreasing rock and mud and increasing fine and coarse gravel.

Sample time 3 (April–May 2010):

Habitat PC1 accounted for 26.6% of variation in stream habitat characteristics, describing a gradient of decreasing sand, small woody debris and large woody debris and increasing submerged vegetation (Table 8.4). Habitat PC2 (14.8%) describes a gradient of decreasing fine gravel, flow velocity and increasing rock. Habitat PC3 (10.0%) describes a gradient of decreasing wetted width, flow velocity and increasing mud and filamentous algae.

These temporal differences in the nature of in-stream habitat gradients are interesting in that flow velocity appears to exert a significant influence on habitat structure only during the third (late summer) sampling period, when slower velocities and narrowing stream widths were associated with muddier substrates and increased growth of filamentous algae.

Short-term flow regime (4 years prior to sampling)

Sample time 1 (July–August 2009):

Short-term flow PC1 accounted for 65.5% of variation in recent flow regime characteristics, describing a gradient of decreasing mean daily flow and mean daily baseflow (Table 8.4). Short-term flow PC2 (17.0%) describes a gradient of decreasing baseflow index and increasing CV of daily flows. Short-term flow PC3 (7.0%) describes a gradient of decreasing number of zero flows and increasing minimum flow magnitude and number of high flows (i.e. floods greater than the median daily flow).

Sample time 2 (October–December 2009):

Short-term flow PC1 accounted for 66.3% of variation in recent flow regime characteristics, describing a gradient of decreasing mean daily flow, mean daily baseflow and magnitude of 10th percentile flows (Table 8.4). Short-term flow PC2 (17.5%) describes a gradient of decreasing baseflow index and increasing CV daily flows. Short-term flow PC3 (5.5%) describes a gradient of decreasing number of zero flow days and increasing minimum flow magnitude and number of high flow spells.

Sample time 3 (April–May 2010):

Short-term flow PC1 accounted for 65.1% of variation in recent flow regime characteristics, describing a gradient of decreasing mean daily flow and mean daily baseflow (Table 8.4). Short-term flow PC2 (18.1%) describes a gradient of decreasing baseflow index and increasing CV daily flows. Short-term flow PC3 (5.6%) describes a gradient of decreasing number of zero flow days and increasing minimum flow magnitude and number of high flow spells.

Across the three sampling periods, the flow variables with the most pronounced effects on recent (4-year) flow history are very similar, differing only in their proportional explanatory power. From this it can be stated that the most important flow variables estimated 4 years prior to each sampling period are essentially low flow factors, such as mean daily flow and mean daily baseflow, as well as the variability of daily flows and the number of zero flow days. All of these flow variables are likely to have effects on fish assemblages (Bunn and Arthington 2002; Kennard et al. 2007).

Long-term flow regime (15 years prior to sampling)

Sample time 1 (July–August 2009):

Long-term flow PC1 accounted for 62.1% of variation in recent flow regime characteristics, describing a gradient of decreasing mean daily flow and magnitude of 1-year annual return interval flood. Long-term flow PC2 (17.7%) describes a gradient of decreasing baseflow index and increasing CV of daily flow magnitude. Long-term PC3 (8.6%) describes a gradient of decreasing constancy and predictability of mean monthly flow and increasing frequency of flow pulses (Table 8.4).

Sample time 2 (October–December 2009):

Long-term flow PC1 accounted for 62.0% of variation in recent flow regime characteristics, describing a gradient of decreasing mean daily flow and magnitude of 1-year annual return interval flood. Long-term flow PC2 (17.9%) describes a gradient of decreasing baseflow index and increasing CV of daily flow magnitude. Long-term PC3 (7.7%) describes a gradient of decreasing constancy and predictability of mean monthly flow and increasing frequency of flow pulses (Table 8.4).

Sample time 3 (April–May 2010):

Long-term flow PC1 accounted for 62.0% of variation in recent flow regime characteristics, describing a gradient of decreasing mean daily flow and magnitude of 1-year annual return interval flood. Long-term flow PC2 (18.5%) describes a gradient of decreasing baseflow index and increasing CV of daily flow magnitude. Long-term PC3 (7.3%) describes a gradient of decreasing constancy and predictability of mean monthly flow and increasing frequency of flow pulses (Table 8.4).

Across the three sampling periods, the flow variables with the most pronounced effects on long-term (15-year) flow history are very similar, differing only slightly in their proportional explanatory power. From this it can be stated that the most important flow variables estimated 15 years prior to each sampling period are essentially mean daily flows and their variability (as for short-term flow history), but also the magnitude of the 1-year annual return interval flood and the increasing frequency of high flow pulses.

In addition, the temporal patterns of flows are important over this longer period of time before fish sampling, such that monthly flow patterns vary across a gradient of predictability and constancy. Most of these flow variables are likely to have effects on fish assemblages (Bunn and Arthington 2002; Kennard et al. 2007).

8.3.3 Relationships between gradients of environmental variables and fish assemblage structure

The purpose of this analysis was to understand the unique and indirect explanatory power of flow variables in the study area over the three sampling occasions relative to other environmental factors that can influence fish in the region.

This preliminary analysis serves primarily to demonstrate how fish assemblages are arrayed against environmental gradients in the study area and also in relation to flow regulation (supplementation). Results from DISTLM analysis are presented in Table 8.5.

Table 8.5: Summary of results from DISTLM analysis of relationships between gradients of environmental variables and fish assemblage structure (presence–absence and composition) for three sampling times

Environmental variables expressed as Principal Components from the PCA of catchment landscape, climate, reach, habitat and flow variables (Table 8.4).

Assemblage metrics	Variable	Pearson correlations of environmental variables with each of the dbRDA axes								Percentage of variation explained by individual axes				
		dbRDA 1	dbRDA 2	dbRDA 3	dbRDA 4	dbRDA 5	dbRDA 6	dbRDA 7	dbRDA 8	Axis	Individual	Cumulative	Individual	Cumulative
Time 1 Pres-Abs	Climate_PC1	-0.44	-0.19	0.43	-0.10	0.64	0.41			1	46.47	46.47	24.59	24.59
	Climate_PC2	-0.12	0.52	0.14	-0.67	-0.36	0.36			2	25.17	71.64	13.32	37.91
	Geol_PC2	-0.83	-0.06	-0.02	-0.03	-0.27	-0.48			3	14.98	86.62	7.93	45.84
	CatchDist_PC1	-0.30	-0.05	-0.51	0.40	-0.23	0.66			4	8.23	94.85	4.35	50.19
	Flow4yr_PC1	0.02	-0.44	-0.61	-0.62	0.22	-0.04			5	4.49	99.34	2.37	52.57
	Flow15yr_PC3	-0.11	0.71	-0.41	0.10	0.53	-0.18			6	0.66	100.00	0.35	52.92
Time 1 Composition	Climate_PC1	0.58	0.11	-0.28	-0.75					1	47.26	47.26	14.43	14.43
	Geol_PC2	0.59	-0.04	-0.49	0.64					2	22.97	70.23	7.01	21.45
	Flow4yr_PC1	-0.55	0.13	-0.82	-0.11					3	20.34	90.57	6.21	27.66
	Flow15yr_PC3	-0.03	-0.99	-0.12	-0.12					4	9.43	100.00	2.88	**30.53**
Time 2 Pres-Abs	Climate_PC1	0.54	0.34	-0.11	0.67	-0.26	-0.27			1	50.13	50.13	27.11	27.11
	Climate_PC2	0.12	0.32	-0.71	-0.35	-0.32	0.40			2	21.16	71.29	11.44	38.55
	Geol_PC2	0.77	-0.41	0.04	-0.16	0.35	0.30			3	15.64	86.92	8.46	47.01
	CatchDist_PC1	0.17	-0.44	0.11	-0.33	-0.73	-0.36			4	7.19	94.12	3.89	50.90
	Flow4yr_PC1	-0.25	-0.55	-0.17	0.55	-0.24	0.50			5	4.26	98.38	2.30	53.20
	Flow4yr_PC3	-0.09	-0.36	-0.66	0.05	0.34	-0.55			6	1.62	100.00	0.88	**54.08**
Time 2 Composition	Climate_PC2	-0.19	-0.85	0.23	0.44					1	50.03	50.03	19.06	19.06
	Geol_PC2	-0.92	0.12	0.22	-0.30					2	29.01	79.04	11.05	30.11
	CatchDist_PC1	-0.29	0.39	-0.33	0.81					3	11.99	91.03	4.57	34.68
	Flow4yr_PC1	0.17	0.34	0.89	0.26					4	8.97	100.00	3.42	**38.10**
Time 3 Pres-Abs	Climate_PC1	0.67	0.19	0.09	0.10	0.34	0.11	-0.02	0.61	1	38.83	38.83	22.31	22.31
	Geol_PC1	-0.23	-0.57	0.06	0.08	0.73	0.15	0.24	-0.02	2	25.73	64.56	14.78	37.09
	Geol_PC2	0.66	-0.36	0.22	-0.08	-0.10	0.13	-0.09	-0.60	3	15.01	79.57	8.62	45.71
	CatchDist_Pc2	-0.09	0.17	0.58	0.18	0.28	-0.62	-0.35	-0.13	4	8.02	87.59	4.61	50.32
	Habit_PC1	0.19	-0.17	-0.73	-0.10	0.20	-0.52	-0.28	-0.05	5	6.10	93.69	3.50	53.82
	Flow4yr_PC1	0.15	0.09	-0.14	0.72	-0.09	-0.25	0.58	-0.16	6	3.56	97.25	2.05	55.87
	Flow4yr_PC3	-0.07	-0.56	0.05	0.49	-0.37	0.05	-0.43	0.35	7	1.51	98.76	0.87	56.74
	Flow15yr_PC2	-0.08	0.37	-0.23	0.43	0.28	0.48	-0.46	-0.32	8	1.24	100.00	0.71	**57.45**
Time 3 Composition	Climate_PC1	0.81	-0.36	0.47						1	53.82	53.82	13.86	13.86
	Climate_PC2	-0.45	0.14	0.88						2	28.38	82.20	7.31	21.17
	Habit_PC1	-0.38	-0.92	-0.05						3	17.80	100.00	4.58	**25.75**

Sample time 1: July-August 2009

Presence–Absence fish assemblage composition

The multivariate DISTLM determined that 52.9% of the variation in fish assemblages in winter 2009 was associated with 6 of 15 predictor variables extracted by PCA (Table 8.5). Both climate 1 and climate 2 Principal Components (air temperature gradients, Table 8.4), reach PC2 (distance to river mouth), 4-year flow PC1 (mean daily flow, mean daily baseflow) and 15-year flow PC3 (constancy, predictability of monthly flows and number of high flow spells) were associated with presence–absence patterns of fish assemblages.

Distance-based redundancy analysis (dbRDA) indicated that these presence–absence patterns were structured by reach scale geology and distance to mouth (dbRDA axis 1) and reach scale air temperature, 4-year flow PC1 (mean daily flow) and 15-year PC3 (constancy, predictability of monthly flows and number of high spells) (dbRDA axis 2) (Fig 8.3a).

Striped gudgeon (*Gobiomorphus australis*), ornate rainbowfish (*Rhadinocentrus ornatus*), gambusia (*Gambusia holbrooki*), Midgleys carp gudgeon (*Hypseleotris* sp.) and purple spotted gudgeon (*Mogurnda adspera*) were the species most strongly associated with patterns of variation in fish assemblage structure based on presence–absence data.

Midgleys carp gudgeon, glassfish (*Ambassis agassizii*), purple spotted gudgeon and gambusia were significantly negatively associated with dbRDA axis 2 (ρ=-0.56, -0.58, -0.64 and -0.66, respectively) representing most of the significant flow predictors determined by the DISTLM (i.e. temperature gradients, short term daily flow and 15-year constancy and predictability of monthly flows and number of high spells).

Modified Gower fish assemblage composition

Patterns in fish assemblage composition (based on CPUE abundances compared using the modified Gower metric) among sites were significantly associated with climatic, reach scale geology and both short and long-term flow history variables (Figure 8.3b). Compared with patterns in fish presence–absence, the model explained much less variation in the fish data (30.5%). The dbRDA axis 1 separated fish assemblages according to a gradient in climate (PC1), reach geology and distance to mouth (PC2) and short-term flow history (PC1), whereas dbRDA axis 2 separated fish assemblages along a gradient of long-term flow history (PC3).

Duboulays rainbowfish (*Melanotaenia duboulayi*) and ornate rainbowfish (*Rhadinocentrus ornatus*) were strongly associated with dbRDA axis 1 (-0.55 and 0.52 respectively), whereas glassfish (*Ambassis agassizii*), gambusia (*Gambusia holbrooki*), Midgleys carp gudgeon (*Hypseleotris* sp.) and purple-spotted gudgeon (*Mogurnda adspera*) were strongly associated with a gradient of long-term flow history, that is dbRDA axis 2 (0.60, 0.62, 0.52 and 0.59 respectively).

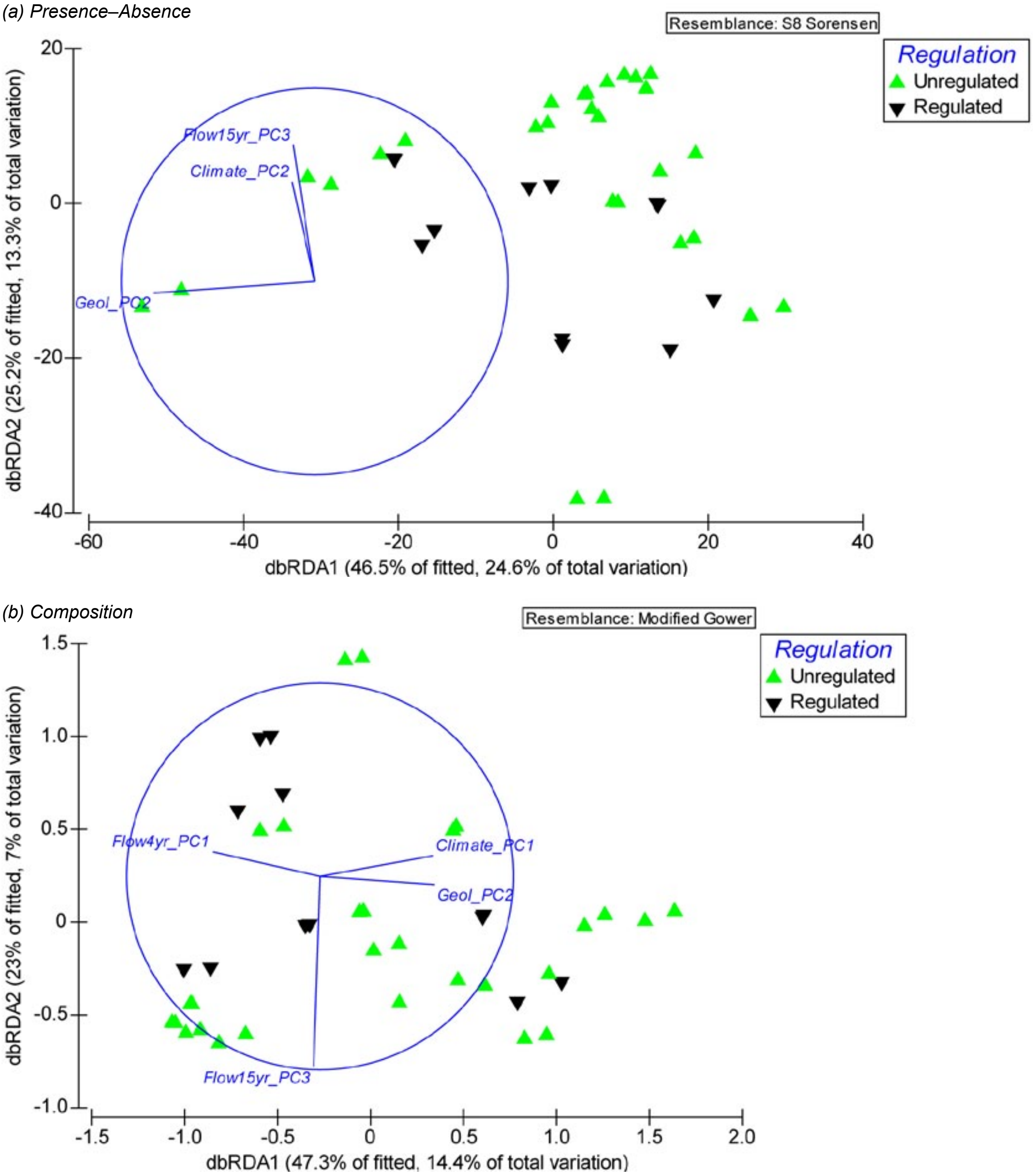
Figure 8.3: Relationship between gradients of environmental variation:

(a) presence–absence patterns

(b) composition of fish assemblage structure at sample time 1 (July–August 2009).

(a) Presence–Absence

(b) Composition



Sample time 2: November–December 2009

Presence–Absence fish assemblage composition

A total of six climate, reach geology, catchment disturbance and short-term flow regime predictor variables were significantly associated with patterns in presence–absence composition of fish assemblages sampled in late spring 2009, and together explained 54.1% of total variation (Table 8.5). The distance-based redundancy analysis (dbRDA) axis 1 explained a gradient of increasing climate (PC1) and reach scale geology (PC2) scores (>0.5), whereas dbRDA axis 2 described a gradient of short-term flow (PC1 and 3) and catchment disturbance (PC2) scores (Figure 8.4a).

In the fish assemblage at this time of year, long-finned eel (*Anguilla reinhardtii*), striped gudgeon (*Gobiomorphus australis*) and empire gudgeon (*Hypseleotris compressa*) were strongly associated with dbRDA axis 1 (0.56, 0.69 and 0.62, respectively), representing climatic and reach geological factors. Australian smelt (*Retropinna semoni*) and common carp (*Cyprinus carpio*) were strongly associated with dbRDA axis 2 (0.42 and -0.48, respectively), representing short-term flow and catchment disturbance gradients.

Modified Gower assemblage composition

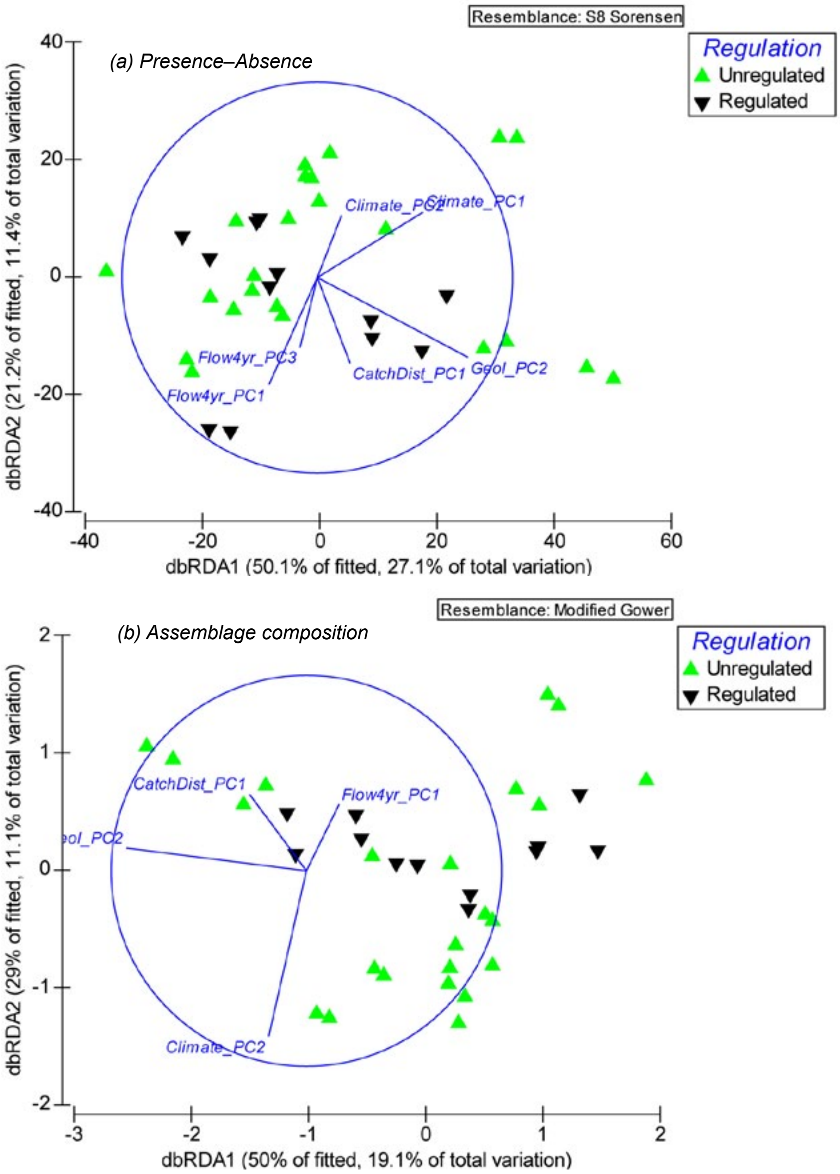
Patterns in fish assemblages (CPUE abundances) in late spring 2009 were significantly associated with climatic (PC2), reach geology (PC2), catchment disturbance (PC1) and short-term flow (PC1) variables. The best model explained a total of 38.1% variation in fish assemblage patterns based on CPUE data (Table 8.5).

The distance-based redundancy analysis (dbRDA) axis 1 was associated with a gradient of reach scale geology PC2 (-0.92), whereas axis 2 described a gradient in climate (PC2) and short-term flow regime (PC1) (-0.848 and 0.341) (Figure 8.4b). Striped gudgeon (*Gobiomorphus australis*) and empire gudgeon (*Hypseleotris compressa*) abundances were strongly associated with dbRDA axis 1 (-0.68 and -0.62, respectively), whereas Pacific blue-eye (*Pseudomugil signifier*) and Australian smelt (*Retropinna semoni*) were strongly associated with dbRDA axis 2 (0.37 and -0.36 respectively).

Figure 8.4: Relationship between gradients of environmental variation:

(a) presence–absence patterns

(b) composition of fish assemblage structure at sample time 2 (October–December 2009).



Sample time 3: April–May 2010

Presence–Absence fish assemblage composition

A total of eight variables, including climate (PC1), reach geology (PC1 and 2), catchment disturbance (PC2), in-stream habitat (PC1), short-term flow history (PC1 and 3) and long-term flow history (PC2) explained 57.4% of the total variation in fish assemblages sampled in autumn 2010 (April–May) (Table 8.5). Climate (PC1) and Geology (PC2) were strongly associated with dbRDA axis 1 (0.667 and 0.656, respectively), whereas long-term flow history (PC2) and reach scale geology (PC2) were associated with dbRDA axis 2 (0.558 and -0.567, respectively) (Figure 8.5a).

Long-finned eel (*Anguilla reinhardtii*), striped gudgeon (Gobiomorphus australis), empire gudgeon (*Hypseleotris compressa*) and ornate rainbowfish (*Rhadinocentrus ornatus*) were strongly associated with a composite climatic and geological gradient - dbRDA axis 1 (0.49, 0.69, 0.50 and 0.53, respectively). Midgleys carp gudgeon (*Hypseleotris* sp.) and Duboulays rainbowfish (*Melanotaenia duboulayi*) were strongly associated with a gradient of long-term flow history and reach-scale geology axis 2 (0.52 and -0.47, respectively).

Modified Gower assemblage composition

Patterns in fish assemblages (CPUE abundances) in autumn 2010 were significantly associated with climatic variables (PC1 and 2) and in-stream habitat (PC1) variables. The best overall model explained 25.7% of total variation in fish assemblages (Table 8.5). The distance-based redundancy analysis (dbRDA) axis 1 was associated with a gradient of climate PC1 (0.81), while dbRDA axis 2 described a gradient of habitat PC1 (-0.92) (Figure 8.5b).

Striped gudgeon (*Gobiomorphus australis*), Duboulays rainbowfish (*Melanotaenia duboulayi*), Australian smelt (Retropinna semoni) and ornate rainbowfish (*Rhadinocentrus ornatus*) were strongly associated with dbRDA axis 1 (0.51, -0.54, -0.58 and 0.45, respectively), while long-finned eels (*Anguilla reinhardtii*) were strongly associated with a gradient of habitat factors (dbRDA axis 2, -0.59).

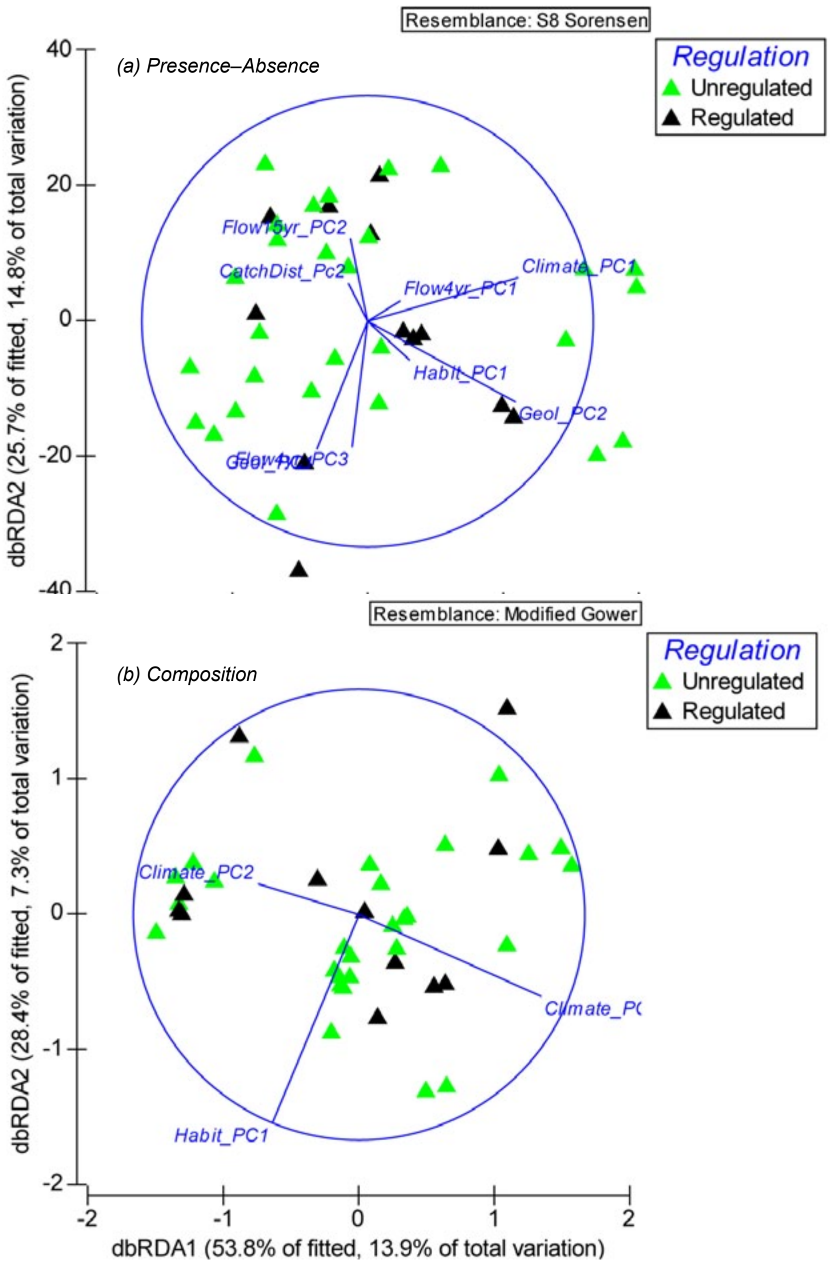


Figure 8.5: Relationship between gradients of environmental variation:

(a) presence–absence patterns

(b) composition of fish assemblage structure at sample time 3 (April–May 2010).

8.3.3 Interpretation of fish assemblage patterns

Relationships between gradients of environmental variables and fish assemblage structure

Environmental variables considered during this study included climatic factors, landscape land use and disturbance regimes, reach scale geomorphology, in-stream habitat structure, flow history (4 and 15-year) and the Gower gradient of flow regime change. These variables displayed significant gradients of variability across the study area (Chapter 5), and these gradients clearly influenced the structure of fish assemblages.

One of the most prominent findings was the consistency of explanatory power associated with models of fish presence–absence patterns. Across all three sampling periods, 52.9–57.4% of spatial variation in fish presence–absence patterns could be associated with gradients in climatic factors, geology, catchment disturbance, channel morphology and flow, although not necessarily in that sequence.

After accounting for the influence of climatic factors, catchment land use, geology and in-stream habitat structure, flow variables alone typically explained only an additional 5–6.5% of variation in fish presence–absence assemblage patterns. In contrast, flow variables had more influence on numerical fish assemblage patterns in the study area. Specifically, short-term flow variables explained 8.97–20.34% of variation in fish assemblage structure, and were most important in the first sampling period (July–August 2009). Long-term flow variables explained 1.24–9.43% and were also most important in the July–August 2009 sampling period.

These findings suggest that gradients of zero and low flows and a gradient of increasing variability (CV) of daily flows, and possibly alterations to these gradients associated with dams and other influences on stream flow regimes, may govern some of the observed fish assemblage differences across the study area, after accounting for other environmental gradients. Other flow gradients of importance to fish are those associated with the magnitude and duration of spells of higher flows (e.g. 10^{th} percentile exceedance flow and 1-year ARI) and the predictability of monthly flow patterns (i.e. some facet of flow seasonality may have influenced fish assemblages).

The composite measurement of altered flow regimes at selected study sites below dams and weirs – the Gower metric (Gower 1971) – did not emerge as having any influence on patterns of variation in fish assemblage structure. In other words, the gradient of flow regime change expressed by the Gower metric (Chapter 3) is not reflected in any strong patterns of fish assemblage structure across the study area. Assemblages at regulated sites appear to lie within the range of variability of those at unregulated sites.

This preliminary result may reflect the relatively coarse nature of this composite metric, which combines all individual flow changes into one composite metric. It does not necessarily mean that flow alterations have had no effect in the study area. What is more likely is that individual flow metrics and combinations of flow metrics (as revealed by PCA) are more likely to have an effect on fish assemblages, and on the individual species that drive patterns of assemblage structure.

Conclusion

Hypothesis 1 is supported by the results of this study. The structure and composition of fish assemblages in the SEQ region are influenced by interactions between flow history, natural catchment characteristics, in-stream habitat factors and anthropogenic disturbances.

8.3.5 Fish indicator analysis

Differences in population level indicators between regulated/supplemented and unregulated sites

Densities of individual species showed few and inconsistent differences between regulated/supplemented and unregulated sites across flow regime classes (Table 8.6). For RFCs, densities of Duboulays rainbowfish (*Melanotaenia duboulayi*) were significantly higher (18.9%) in regulated/supplemented sites when compared with unregulated sites in RFC 2. In contrast, densities of Pacific blue-eye (*Pseudomugil signifier*) were significantly lower (77.6%) in regulated sites when compared with unregulated sites in RFC 1. No other species showed significant differences between regulated/supplemented and unregulated sites within any of modelled flow classes.

Fish species population densities did, however, reflect significant differences between regulated/supplemented and unregulated sites among HFCs. Densities of long-finned eels (*Anguilla reinhardtii*), fire-tailed gudgeon and *Hypseleotric* sp. were significantly lower in regulated/supplemented sites when compared with unregulated sites in gauged HFC 2. Patterns in the densities of Pacific blue-eye (*Pseudomugil signifier*) were inconsistent among current flow classes.

Densities were significantly lower (30%) in regulated/supplemented sites when compared with unregulated sites in HFC 3, yet were significantly greater (82%) in regulated/supplemented sites when compared with unregulated sites in HFC 4. Densities of all other species did not differ significantly between regulated and unregulated sites in any of the current flow regime classes.

Table 8.6: Significance levels (p) and proportion of variance explained by the RFC x regulation interaction in both RFCs and HFCs

Pairwise tests are shown to indicate where significant differences between regulated and unregulated sites in each flow regime class were detected. ND indicates no significant difference (p>0.05). For univariate response variables, the mean percentage difference between unregulated and regulated/supplemented sites is given to indicate level of change. VC = variance component.

Indicators	RFC x regulation pairwise tests					HFC x regulation pairwise tests					
	P value	*VC*	*1*	*2*	*5*	*P value*	*VC*	*1*	*2*	*3*	*4*
Angrei	0.094	7.1	ND	ND	ND	0.010	0.2	ND	0.016 (-25.18%)	ND	ND
Meldub	0.002	4.7	ND	0.0036 (18.9%)	ND	0.059	0.3	ND	ND	ND	ND
Retsem	0.298	2.1	ND	ND	ND	0.108	0.2	ND	ND	ND	ND
Tan tan	0.394	1.3	ND	ND	ND	0.583	0.0	ND	ND	ND	ND
Hyp klu	0.873	0.0	ND	ND	ND	0.978	0.0	ND	ND	ND	ND
Hyp gal	0.828	0.0	ND	ND	ND	0.004	0.3	ND	0.0163 (-31.33%)	ND	ND
Pse sig	0.034	8.5	0.001 (-77.6%)	ND	ND	0.000	0.4	ND	ND	0.036 (-30.07%)	0.004 (82.6%)
Gam hol	0.152	0.1	ND	ND	ND	0.074	0.2	ND	ND	ND	ND
Hypsp1	0.443	0.3	ND	ND	ND	0.017	0.2	ND	0.081 (-28.9%)	ND	ND
Gobaus	0.612	0.0	ND	ND	ND	0.010	0.1	ND	ND	ND	ND
ALR	0.178	3.4	ND	ND	ND	0.791	0.0	ND	ND	ND	ND
SPR	0.286	3.3	ND	ND	ND	0.014	0.3	ND	ND	0.0123 (21.48%)	ND
NSPR	0.256	4.9	ND	ND	ND	0.019	0.3	ND	ND	0.0144 (29.57%)	ND
NMR	0.587	0.0	ND	ND	ND	0.021	0.2	ND	ND	ND	0.047 (98.27%)
PNR	0.215	5.7	ND	ND	ND	0.900	0.0	ND	ND	ND	ND
ALD	0.129	0.2	ND	ND	ND	0.066	0.2	ND	ND	ND	ND
TABUN	0.509	0.0	ND	ND	ND	0.195	0.1	ND	ND	ND	ND
NAD	0.497	0.0	ND	ND	ND	0.191	0.1	ND	ND	ND	ND
NMD	0.652	0.0	ND	ND	ND	0.301	0.1	ND	ND	ND	ND
PND	0.121	1.3	ND	ND	ND	0.062	0.2	ND	ND	ND	ND
SPDen	0.310	1.9	ND	ND	ND	0.105	0.2	ND	ND	ND	ND
Pres-abs. comp	0.005	9.0	0.004	ND	ND	0.026	0.1	0.0788	ND	ND	ND
Gower comp	0.011	9.2	0.0104	ND	0.0021 (-31.18%)	0.000	0.2	0.0027	ND	0.0012	0.0344
25% FOC	0.017	0.1	0.0171	ND	0.0035 (-40.74%)	0.000	0.3	0.0013	ND	0.0001	ND
<1% tabun	0.010	0.1	0.0177	ND	0.0011 (-38.19%)	0.000	0.3	0.0008	ND	0.0001	ND
NonMig comp	0.014	0.1	0.0301	ND	0.0036 (-25.74%)	0.000	0.2	0.0028	ND	0.0182	0.0362
Native comp	0.012	0.1	0.0087	ND	0.0021 (-32.09%)	0.000	0.3	0.0024	ND	0.0011	0.0428
Shortlived taxa	0.030	0.1	0.0412	ND	0.0062 (-25.03%)	0.000	0.3	0.0058	ND	0.0129	0.0379

Differences in univariate assemblage level indicators between regulated/supplemented and unregulated sites

There were significant differences in species richness, non-migratory species richness, total fish density, native density, alien density, non-migratory density and species density among RFCs (results not shown). None of these univariate metrics of fish assemblage structure show significant differences between regulated/supplemented and unregulated sites within RFCs (Table 8.6).

However, 3 of the 11 composite assemblage level metrics showed significant differences between regulated/supplemented and unregulated sites within HFC. Total species richness and native species richness were significantly greater (21% and 29%, respectively) in regulated/supplemented sites compared with unregulated sites in HFC 3. Non-migratory species richness was almost double (98% higher) in regulated/supplemented sites when compared with unregulated sites in HFC 4 (Table 8.6).

Differences in multivariate fish assemblage composition between regulated/supplemented and unregulated sites

Fish assemblage composition differed inconsistently between regulated/supplemented and unregulated sites among RFCs. For the presence–absence composition of assemblages, differences between regulated/supplemented and unregulated sites were only detected in RFC 1, but not RFC 2 or 5 (Table 8.6).

All metrics of fish assemblage composition (total assemblage composition, native species composition, widespread species composition, abundant species composition, non-migratory assemblage composition, and short-lived species composition) reflected inconsistent patterns among regulated/supplemented and unregulated sites. Specifically, these six metrics were significantly different between regulated/supplemented and unregulated sites in RFCs 1 and 5, but not RFC 2 (Table 8.6).

All metrics of fish assemblage composition differed significantly between regulated/supplemented and unregulated sites in HFC 1, whereas all metrics of assemblage composition (excluding presence–absence composition) showed significant differences between regulated/supplemented and unregulated sites in HFC 3 (Table 8.6).

Total assemblage composition, non-migratory assemblage composition, native assemblage composition and short-lived assemblage composition differed significantly between regulated/supplemented and unregulated sites in HFC 4. No metrics of assemblage composition differed significantly between regulated/supplemented and unregulated sites in HFC 2.

8.3.6. Interpretation of fish indicator responses

If individual species populations and/or composite fish assemblage metrics are reflective of flow regime change, significant differences between regulated/supplemented and unregulated sites in some or all of the modelled RFCs would be expected. This expectation arises from ELOHA and the nature of the study design, which deliberately compared regulated/supplemented with unregulated sites within and among flow regime classes.

Only 2 of the 10 individual species population densities and none of the univariate assemblage metrics showed significant differences between regulated/supplemented and unregulated sites in any of the RFCs. Densities of the widespread Duboulays rainbowfish (*Melanotaenia duboulayi*) were 18% greater in regulated/supplemented sites when compared with unregulated sites in RFC 2, whereas densities of Pacific blue-eye (*Pseudomugil signifier*) were 77% lower in regulated/supplemented sites compared with unregulated sites in RFC 1.

In contrast, many more species and composite assemblage level indicators differed significantly between regulated and unregulated sites grouped within gauged (Historic) flow classes. Strict criteria for selecting fish indicators of flow regime change were that a metric should differ significantly between Reference (natural) regulated/supplemented and unregulated sites, yet show no significant differences between regulated/supplemented and unregulated sites under Historic flow regimes. Duboulays rainbowfish (*Melanotaenia duboulayi*) was the only species ('indicator') to meet both criteria.

Densities of Duboulays rainbowfish were significantly greater in regulated/supplemented sites when compared with unregulated sites within RFC 2, yet did not differ significantly between regulated/supplemented and unregulated sites in any of the HFCs.

Duboulays rainbowfish (*Melanotaenia duboulayi*) were collected in >75% of all samples during this study. Its frequency of occurrence and response to flow conditions and alterations adds strength to the suggestion that this species may be a useful indicator in tests of the ELOHA framework in SEQ. Long-finned eels (*Anguilla reinhardtii*) and Australian smelt (*Retropinna semoni*) were also common but did not show equivalent responses to flow change below dams.

Pacific blue-eye, a less common species, did show some marked responses to flow regime change, but responses were not fully concordant with the criteria used to identify useful indicator species or metrics. If these criteria were to be relaxed, Pacific blue-eye (*Pseudomugil signifier*) could be a useful indicator species and this possibility was explored in tests of Hypothesis 3.

8.3.6 Spatial variability and natural geographic variation in assemblage composition and population density

All multivariate measures of fish assemblage composition indicated significant differences between regulated/supplemented and unregulated sites in some or all RFCs and HFCs. For RFCs, there were significant differences in all metrics of fish assemblage composition in RFC 1 and 5, but not RFC 2 (data not shown). In gauged flow classes, there were significant differences in all metrics between regulated/supplemented and unregulated sites in HFC 1, all composition metrics (excluding presence–absence composition) differed in current HFC 3 and most metrics differed in HFC 4.

If fish assemblage composition was significantly influenced by flow variability alone (and also sensitive to flow regulation), then differences in composition between regulated/supplemented and unregulated sites in RFCs (not in HFCs) would be expected. Two of the multivariate assemblage metrics (Sorensen presence–absence similarity and the Gower metric) were significantly different between regulated/supplemented and unregulated sites in some HFCs, but less different than in the Reference flow classification.

These results suggest some adjustment in the composition of fish assemblages at certain regulated/supplemented sites over time, as predicted by ELOHA if flow is a major driver of aquatic biodiversity patterns and ecosystem structure. Again, the length of time since flow regulation may be a factor in the relatively slight adjustment of fish to Historic flow conditions at some regulated/supplemented sites, whereas at other regulated/supplemented sites in HFCs 3 and 4, more adjustment seems to have occurred.

Conclusion

Hypothesis 2 is supported by some of the results of this study. The structure and composition of fish assemblages differ in a range of biological metrics across pre-development (Reference) and Historic (regulated/supplemented) flow regime classes, and between regulated and unregulated sites.

8.3.7 Fish response to flow regime gradients

This part of the project was designed to visualise and test the strength of the relationships between key flow variables identified as being associated with patterns in fish assemblage composition (Hypothesis 1) and indicators of fish population and assemblage structure across flow regime classes (Hypothesis 2) in the study area.

These analyses addressed fish responses to **flow variability** across the study area (including all sites, all times, and both regulated and unregulated flows), and responses to **flow regime change** (based on comparison of Reference and Historic flow data).

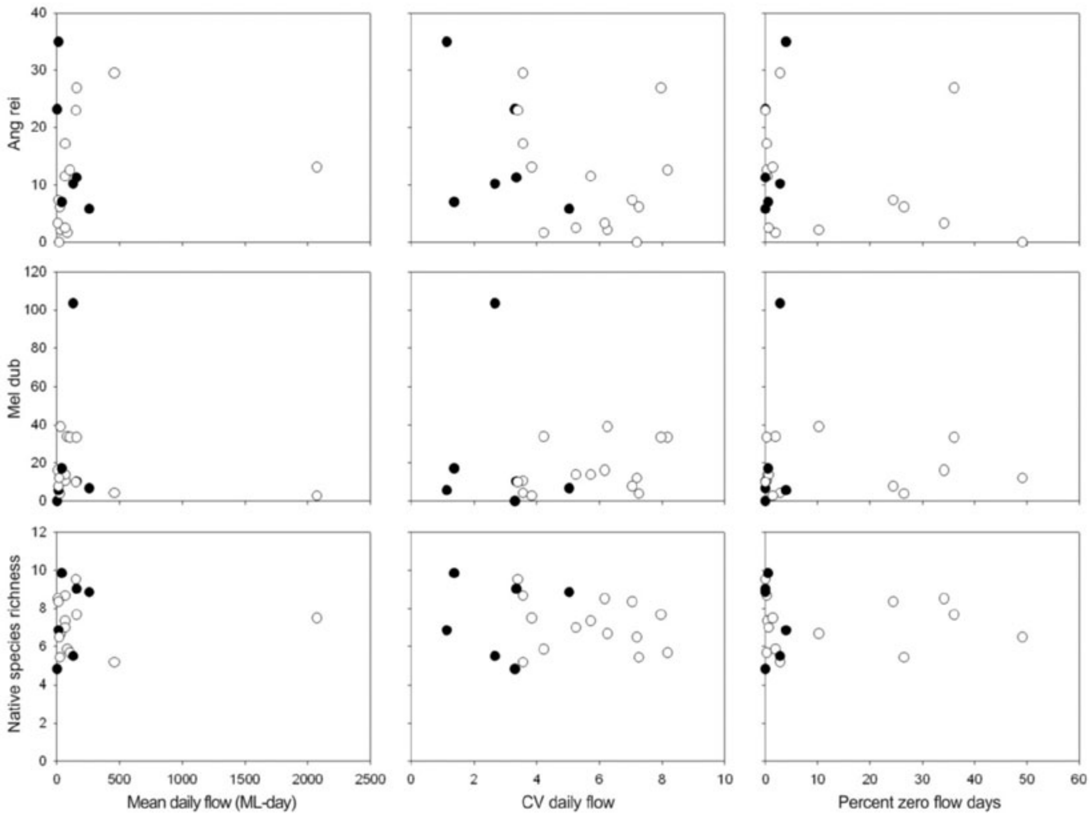
Fish response to flow variability

Three response variables derived from Hypothesis 2 were tested: mean densities per site and sampling time of long-finned eels (*Anguilla reinhardtii*), Duboulays rainbowfish (*Melanotaenia duboulayi*) and native species richness. None of the three response variables showed significant relationships with variation in flow magnitude (mean daily flow), flow variability (CV of daily flow) and flow regime harshness (percentage of zero flow days).

All relationships were not significantly different from zero (P>0.1), indicating that patterns in fish assemblages were highly variable and this variance exceeded any potential variation explained by attributes of the flow regime (Figure 8.6).

Figure 8.6: Plots indicating patterns in the mean densities per site and sampling time of long-finned eel (Ang rei), Dubolays rainbowfish (Mel dub) and native species richness in relation to mean daily flow, CV of daily flow and percentage of zero flow days in the 4 years prior to sampling at each sample point

Clear circles indicate unregulated sites, whereas solid circles indicate regulated sites.



The second set of analyses modelled several of the common fish species as well as univariate measures of fish assemblage structure (species richness, species density and alien species density) against flow metrics revealed to influence fish in the multivariate tests of Hypothesis 1. Two sets of flow metrics were tested – those describing 4-year antecedent flow periods (number of zero flow days, mean daily flow and CV of daily flow), and those describing 15-year antecedent flow periods (1-year ARI, high spell number (the number of floods greater than the median flow) and the constancy and predictability of mean monthly flows (see Table 8.1 for details of these metrics).

Significant relationshps are presented in Table 8.7 with plot of these relationships presented in Figure 8.7.

From these analyses it appears that species richness at each sampling site is significantly related to the variability (CV) of mean daily flow during the 4 years before sampling, to the number of floods greater than the median flow (HSNum), and to the constancy and predictability of monthly flows (estimated for the 15 years before sampling). These flow metrics were associated with gradients of fish assemblage structure in multivariate analyses, but here the result show their individual links to fish assemblage richness.

There is an indication that the number of alien fish individuals per site increased with increase in the number of zero flow days over the 4-year antecedent flow period. However, this positive relationship is driven by a small number of extreme values (Fig.8.7). The ratio of the number of species to total individuals in a sample appears to decline with increasing mean daily flow.

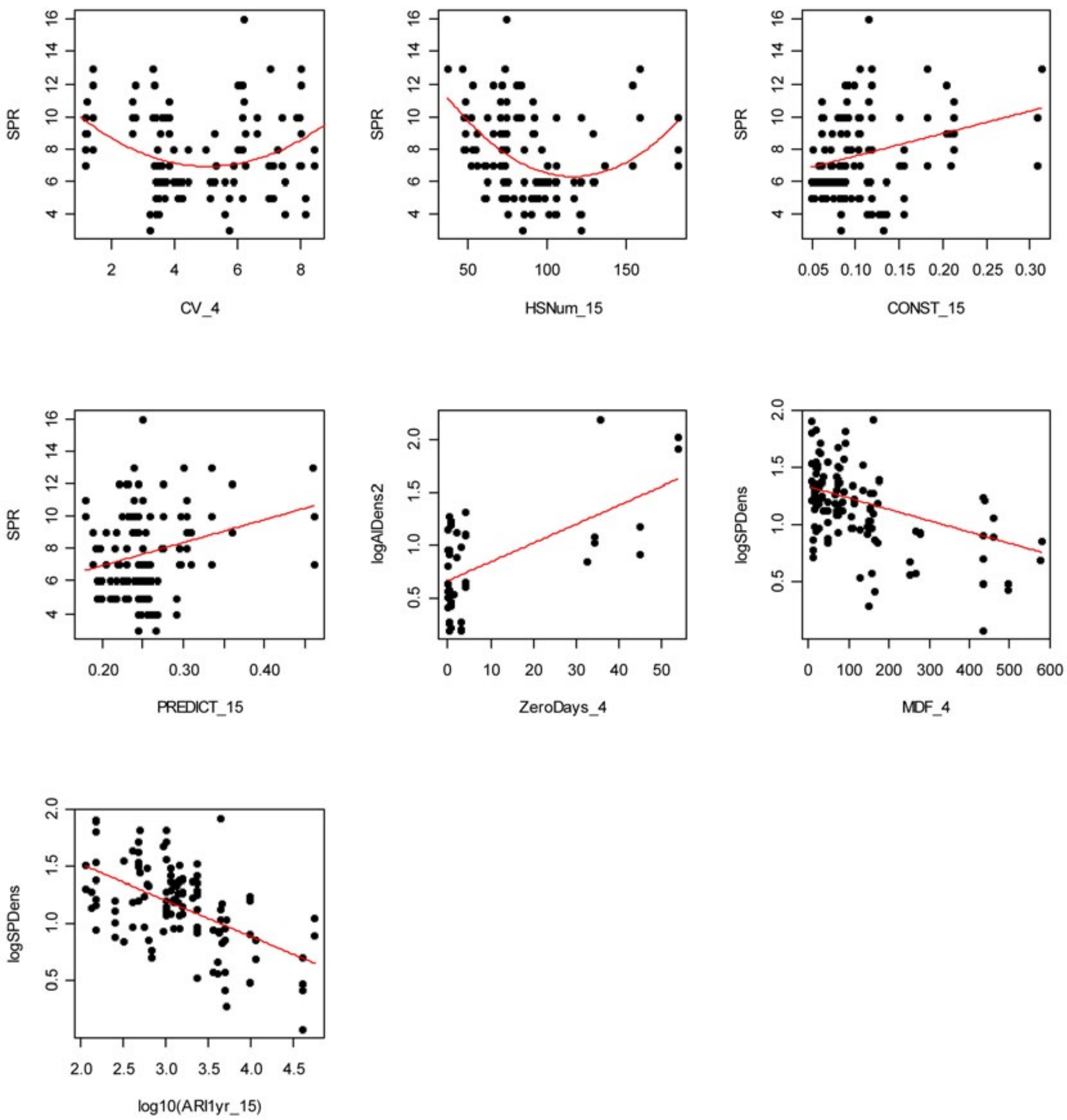
Table 8.7: Summary of GLS regression models describing relationships between fish metrics and selected flow metrics

D *is the value of the log-likelihood ratio test comparing model fits (i.e. the null model compared to the alternate model). Significance for* D *is determined by comparison with* X^2 *with one degree of freedom (3.841). Significance: * 0.01 ≥* p *> 0.001; ***p *≤ 0.001.*

Fish metric	Flow metric	Equation	Log-likelihood (flow model)	Log-likelihood (null model)	D
Species Richness (SPR)	CV_4	$SPR = 11.694 + -1.883x + 0.186x^2$	-270.71	-276.70	11.99
SPR	HSNum_15	$SPR = 16.807 + -0.181x + 0.0007x^2$	-262.50	-276.70	28.41
SPR	CONST_15	$SPR = 6.192 + 13.891x$	-269.23	-276.70	14.95
SPR	PREDICT_15	$SPR = 4.113 + 14.168x$	-269.40	-276.70	14.61
Alien Density	ZeroDays_4	$\log_{10}(AlienDensity) = 0.657 + 0.018x$	-25.15	-30.50	10.70
Ratio of the number of species: total individuals in a sample (c)	MDF_4	$\log_{10}(SPDens) = 1.333 + -0.001x$	-34.71	-46.06	22.70
SPDensity	ARI_1yr_15	$\log_{10}SPDens = 2.158 + -0.317 \log_{10}(ARI\_1yr\_15)$	-28.51	-46.06	35.11

Figure 8.7: Plots of model fits for significant GLS models describing relationships between fish metrics and selected flow metrics

See Table 8.6 for GLS model summaries.

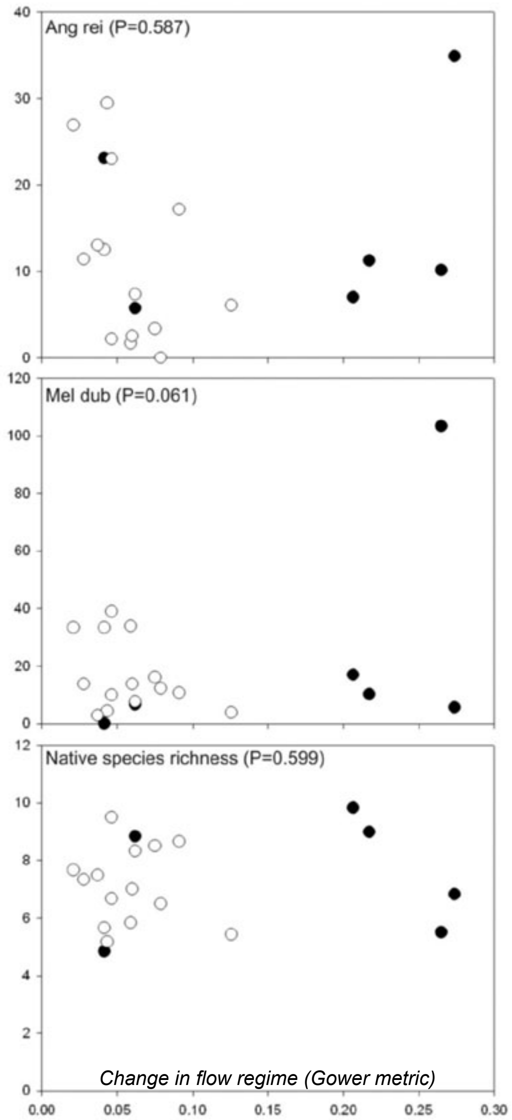


Fish response to flow regime change

Another set of analyses tested the significance of relationships between change in the overall flow regime (expressed as the Gower metric of dissimilarity) and densities of long-finned eels (*Anguilla reinhardtii*), Duboulays rainbowfish (*Melanotaenia duboulayi*) and native species richness. Densities of these two species and native species richness were not significantly related to change in flow regime across the Gower gradient (Figure 8.8).

Figure 8.8: Scatterplots testing the relationship between change in flow regime (Gower metric) and densities of long-finned eel (Ang rei), Duboulays rainbowfish (Mel dub) and native fish species richness in SEQ rivers

Clear circles indicate unregulated sites, whereas solid circles indicate regulated/supplemented sites. Probability values (P) indicate the significance of the relationship differing from zero (i.e. a flat line).



Finally, the effect of flow regulation on fish metrics was determined using PLS modelling. Reference models for each fish metric were constructed from predictor variables unaffected by flow regulation. The Reference model was then used to predict fish metric values at regulated/supplemented sites.

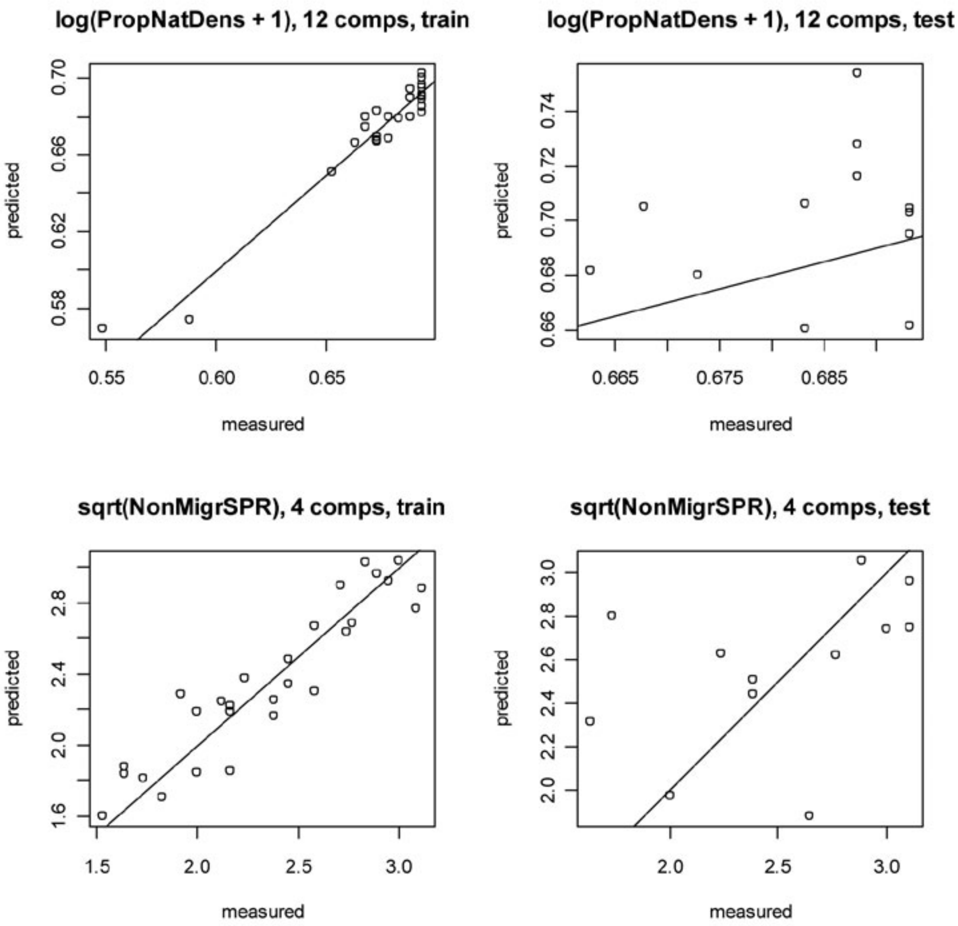
Only two fish metrics (PropNatDens and NonMigrSPR) had sufficiently high R^2 values to predict metric values at regulated (test) sites (Table 8.8). Figure 8.9 shows that PropNatDen was lower than expected at 2 regulated sites and higher than expected at 10 sites, while non-migratory species richness was lower than expected at 6 sites and higher than expected at 6 sites.

Table 8.8: Summary of PLS model fits for selected fish metrics

The effect of flow regulation ('effect') is calculated as (Observed–Predicted/Predicted ×100) and is shown as the mean effect across all sites ± 95% confidence interval. If the 95% confidence interval does not include zero then an effect from flow regulation exists.

Metric	R^2	Components	Effect
log(PropNatDens + 1)	0.709	12	-56.1 ± 0.4
√NonMigrSPR	0.635	4	3.5 ± 13.2

Figure 8.9: Plots of predicted versus measured values of fish metrics for PLS Reference models (left hand column) and PLS test models (right hand side).

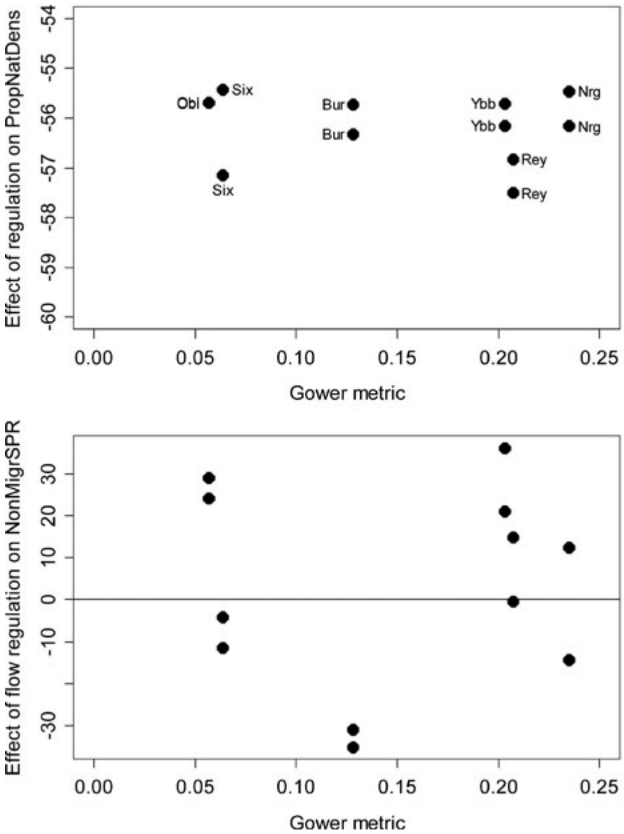


The effect of flow regulation for the two fish metrics was plotted against the Gower metric (Figure 8.10). This tested the ELOHA concept that increasing degree of flow regulation is associated with increasing biotic change.

If this concept is valid then the effect of flow regulation should increase with increasing Gower metric values. While there are only six dams represented in Figure 8.10, it is evident that there is no increasing effect of flow regulation on these fish metrics.

Figure 8.10: Scatterplots showing relationships between the effect of flow regulation versus the Gower metric (an indicator of flow regime change) for sites downstream of dams in the study area

Two sites were surveyed downstream of each dam and each dam is represented by a pair of points (the average of flow regulation effects for all samples). Site codes: Obi – Obi Obi Creek; Six – Six Mile Creek; Bur – Burnett Creek; Ybb – Yabba Creek; Rey – Reynolds Creek; Nrg – Nerang River.

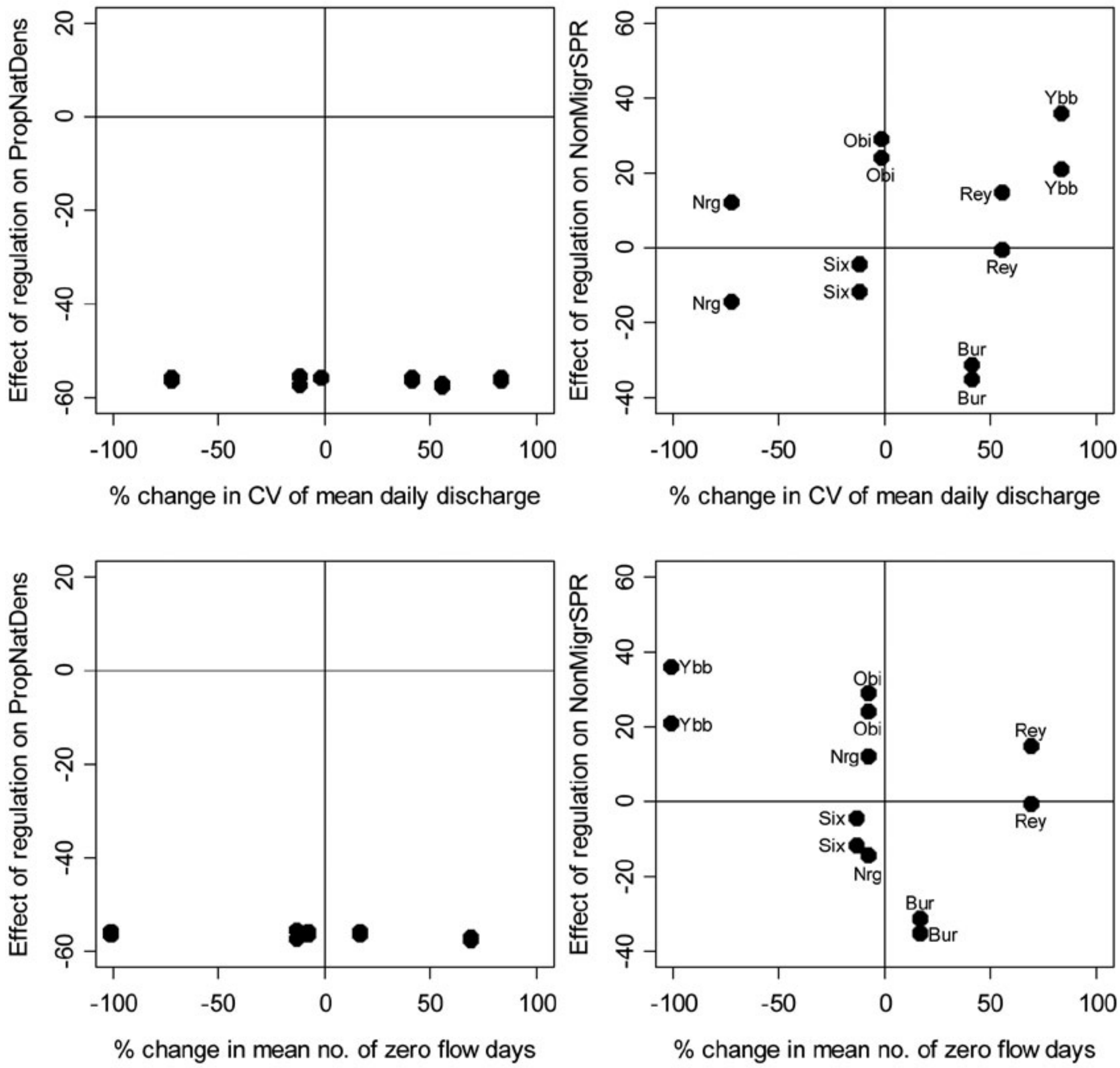


The final set of plots presents changes in the two fish metrics as predicted by the PLS models, versus percentage change in the CV of mean daily discharge and the mean number of days without flow (Figure 8.11).

This form of plot derives from Poff and Zimmerman (2010). It is clear that the proportion of native fish density is depressed by both negative and positive change in daily flow variability. However, there is no predictable gradient of fish response to change in this flow metric, irrespective of the finding that CV of daily flow influenced the structure of fish assemblages.

A different set of responses is apparent in Figure 8.11 for non-migratory species richness, with a hint of gradients of response to both flow metrics, but the patterns are not statistically significant.

Figure 8.11: Scatterplots of change in total cover as predicted by PLS models (effect of flow regulation) *versus* % change in individual flow metrics, calculated as (Historic–Reference)/Reference x 100).



Conclusion

Hypothesis 3 is supported by findings for the influence of flow on fish species richness. Fish population and assemblage indicators do vary predictably along gradients of flow variability.

This fish metric was significantly related to the variability (CV) of mean daily flow during the 4 years before sampling, to the number of floods greater than the median flow (HSNum), and to the constancy and predictability of monthly flows (estimated for the 15 years before sampling). There is also an indication that the number of alien fish individuals per site increased with increase in the number of zero flow days over the 4-year antecendent flow period.

The ratio of the number of species to total individuals in a sample declined with increasing mean daily flow. A similar trend is evident between the ratio of the number of species to total individuals in a sample and the 1-year annual return interval flood (ARI_1yr). As the number of such floods increased over the 15-year antecendent flow period, fewer species but many individuals were present.

However, the hypothesis that fish population and assemblage metrics will vary predictably along gradients of flow regime change is not supported by the findings of this study. The proportion of native fish density was found to be depressed by both negative and positive change in daily flow variability (CV of mean daily flow). A different set of responses was apparent for non-migratory species richness, with a hint of gradients of response to both flow metrics, but the patterns were not statistically significant.

8.4 Discussion

8.4.1 The Fauna

The fish fauna sampled during this ELOHA trial is typical of that to be expected in streams and rivers of SEQ (Appendix 3) and the numerical dominance patterns of common species were also familiar from previous studies (Pusey et al. 1993; Pusey et al. 2004; Kennard et al. 2007). These species were long-finned eel, western carp gudgeon, Duboulays rainbowfish (*Melanotaenia duboulayi*), Pacific blue-eye (*Pseudomugil signifier*) and Australian smelt (*Retropinna semoni*).

Species in this group were the most commonly encountered during sampling, with the addition of the freshwater catfish (*Tandanus tandanus*). However, a different suite of species occurred in highest densities. These species included striped gudgeon, ornate rainbowfish, Duboulays rainbowfish, Australian smelt (*Retropinna semoni*) and the firetailed gudgeon (*Hypseleotris galii*).

Five alien species were sampled – common carp (*Cyprinus carpio*), gambusia or mosquitofish (*Gambusia holbrooki*), tilapia (*Oreochromis mossambicus*), swordtail (*Xiphophorus helleri*) and platy (*Xiphophorus manculatus*) – yet contributed to only 3.71% of the total number of fish collected. This latter finding is significant as it has reduced the potential for alien species to serve as indicators of flow regulation (supplementation) by dams, as discussed below.

Hypothesis testing

The ELOHA framework is underpinned by several concepts. This study tested several hypotheses related to these concepts as a means of validating the utility of the framework for fish in SEQ. Three major hypotheses were tested, each providing input to the final evaluation of the ELOHA framework in SEQ.

Hypothesis 1

Hypothesis 1 stated that the *structure and composition of fish assemblages in the SEQ region will be influenced by interactions between flow history, natural catchment characteristics, in-stream habitat factors and anthropogenic disturbances.* This hypothesis is supported by the results of the study.

Environmental variables considered during this study included climatic factors, catchment characteristics, land use disturbance, reach scale geomorphology, in-stream habitat structure and flow history. These variables displayed significant gradients of variability across the study area (Chapter 5), and these gradients clearly influenced the distribution patterns of fish and the composition of fish assemblages.

Across all three sampling periods, 52.9–57.4% of spatial variation in fish presence–absence patterns could be associated with gradients in climatic factors, catchment characteristics and land use disturbance, geology, channel morphology and flow history. This is not an unexpected result, given the fact that species distributions are often constrained by climatic factors such as thermal regime and other climatic factors, and by historical patterns of landscape evolution that shape the nature of stream channels and the structure of aquatic habitats and their accessibility (e.g. Esselman and Allan 2010; Snelder and Lamouroux 2010).

In the general literature on stream fish assemblages, these environmental factors are visualised as acting like a series of filters, progressively removing species from the broader regional ‘pool’ of species, until a series of subsets of the total regional fauna are sifted out and persist in particular parts of the landscape and under certain habitat conditions (Poff 1997; Kennard et al. 2007; Stewart-Koster et al. 2007).

Flow variables typically explained only 5–6.5% of variation in fish presence–absence (distribution) patterns in the study area, other environmental gradients (climate, geomorphology) having more influence on fish distribution patterns. Flow variables had more influence on numerical fish assemblage patterns (based on CPUE).

Short-term (4 years before fish sampling) flow variables explained 8.97–20.34% of variation in fish assemblage composition, and were most important in the first sampling period (July–August 2009). Long-term (15 years before fish sampling) flow variables explained 1.24–9.43% and were also most important in the July–August 2009 sampling period.

As reported above, the full suite of flow variables considered during this study was reduced down to several sets of independent, composite variables that summarise the major patterns of flow variation across the study area. From this analysis there are many similarities in the short and long-term flow variables that distinguish major flow regime gradients in the study area. This similarity arises because the two antecedent flow periods overlap to some extent.

Results of multivariate analyses suggest that zero flow and low flow gradients and a gradient of increasing variability (CV) of daily flows, and possibly alterations to these gradients associated with dams and other influences on stream flow regimes, may govern some of the observed fish assemblage differences across the study area, after accounting for other environmental gradients.

Other flow gradients of importance to fish are those associated with higher flows (e.g. 1-year ARI flood), and the predictability and constancy of monthly flow patterns, implying that some facet of flow seasonality over 15 years before sampling influenced fish assemblage structure.

The composite measurement of altered flow regimes at selected study sites below dams and weirs – the Gower metric (Gower 1971) – did not emerge as having any influence on patterns of variation in fish assemblage structure. In other words, the gradient of flow regime change expressed by the Gower metric (Chapter 3) is not reflected in any single strong gradient of fish assemblage structure across the study area.

This result may reflect the relatively coarse nature of this composite metric. It does not necessarily mean that flow alterations have had no effect in the study area. What is more likely is that individual flow metrics and combinations of flow metrics (as revealed by PCA) are more likely to have an effect on fish assemblages, and on the individual species that drive patterns of assemblage structure.

Tests of Hypotheses 2 and 3 build on the results of multivariate analyses. Tests of Hypothesis 2 explored the effects of flow regime change by comparing regulated/supplemented and unregulated sites across the flow regime classes. Tests of Hypothesis 3 are designed to search for gradients of fish response to flow regime change expressed as the Gower metric and in relation to individual flow metrics, with emphasis on the flow variables that demonstrably have an association with fish assemblage structure and the dominance of individual species across sites and times. The species of most prominence in the multivariate tests of Hypothesis 1 included the various species of gudgeons, both species of rainbowfish, Australian smelt (*Retropinna semoni*), long-finned eels (*Anguilla reinhardtii*), southern blue-eye (*Pseudomugil signifier*), glass fish (*Ambassis agasizii*), and alien carp (*Cyprinus carpio*) and Gambusia – the mosquitofish (*Gambusia holbrooki*). The jroe common and abundant of these species were tested for their responses to flow variability and flow regime change, as discussed below.

Hypothesis 2

Hypothesis 2 stated that the structure and composition of fish assemblages will differ in a range of biological metrics across pre-development (unregulated) and Historic (regulated/supplemented) flow regime classes, and between regulated/supplemented and unregulated sites. This hypothesis is supported by results of the study.

There were significant differences in seven univariate metrics of fish assemblage structure across the RFCs (species richness, non-migratory species richness, total fish density, native density, alien density, non-migratory density and species density). This result confirms the ELOHA prediction that distinctive flow regime classes will have different ecological characteristics.

However, none of these univariate metrics of fish assemblage structure showed significant differences between regulated/supplemented and unregulated sites within any of the RFCs. This suggests that flow regime alteration by dams and other factor has not significantly affected these indicators of assemblage structure in the study area, or at least not during the time elapsed since dams were constructed or modified (10–50 years ago).

Strict criteria for selecting fish indicators of flow regime change were that a metric should differ significantly between Reference (natural) regulated/supplemented and unregulated sites, yet show no significant differences between regulated/supplemented and unregulated sites under Historic flow regimes. Duboulays rainbowfish (*Melanotaenia duboulayi*) was the only species ('indicator') to meet both criteria. Densities of this rainbowfish were significantly greater in regulated/supplemented sites when compared with unregulated sites within RFC 2, yet did not differ significantly between regulated/supplemented and unregulated sites in any of the HFCs.

No univariate assemblage level indicators showed significant differences between regulated/supplemented and unregulated sites in any of the RFCs, indicating that there have been no substantial changes in these attributes of fish assemblages attributable to flow regulation. Previously, the proportion of native species has been found to be greater in unregulated sites when compared with regulated/supplemented sites in both coastal and inland regions of New South Wales (Gehrke et al 1995; 1999; Gehrke and Harris 2001). However, in the present study, the proportion of both native species density and richness did not differ significantly between regulated/supplemented and unregulated sites.

Three interpretations of these findings are possible.The first is that the effects of flow regime change by dams and water extraction on measures such as species richness take time to become detectable, or to occur at all. Rivers in NSW have been influenced by dams and water extraction for a much longer period (up to 100 years) than the rivers studied in SEQ (10–50 years since dam construction or modification by raising the dam wall, e.g. Hinze Dam).

This far shorter history of flow regime change may explain the lack of significant impacts on fish species richness. A decrease in richness at regulated/supplemented sites would imply failure to survive, or recruit or immigrate, and so persist in a river reach. The impairment of these processes to the point of species absence from a river reach would not be immediate.

A low level of recruitment and/or immigration could be sufficient to allow a species to persist for some time after the flow regime is altered. In contrast, rivers experiencing a longer history of flow regime change can be expected to show greater change in the processes that sustain fish assemblages as well as ecosystem structure and function (Poff et al. 1997; Bunn and Arthington 2002).

Secondly, the nature of flow regime change in rivers of SEQ has been relatively slight. Of the 35 flow metrics tested, 57% showed a change of 20% or less between modelled (Reference) and Historic flow conditions across the region. For the sites sampled, the equivalent figure was 31% and 21% showed a change of 10% or less (Chapter 3). Many relatively low level differences amount to a relatively subtle flow alteration gradient across the study area. The flow alteration gradient reflected by the Gower metric indicates that on a scale of 0–1, the maximum overall flow regime change is around 0.25.

Finding distinctive patterns of change in composite fish assemblage indicators along such a slight gradient of flow alteration may be difificult. However, some individual flow metrics changed positively or negatively by 50–125% at some of the monitored sites, and therefore it was reasonable to anticipate significant responses to flow regime change in some fish indicators.

Thirdly, the number of samples required to detect a significant difference in composite assemblage level indicators would need to be much larger than was possible in the current study of alterations in flow conditions below dams. This issue of statistical power raises challenges for tests of the ELOHA framework in regions where there are few dams or management practices that alter flow regimes, and few opportunities to sample numerous sites along a distinctive flow alteration gradient, especially a stronger gradient than the one studied here.

Two of the multivariate assemblage metrics (Sorensen presence–absence similarity and the modified Gower assemblage similarity metric) were significantly different between regulated/supplemented and unregulated sites in some HFCs, but were less different than in the Reference flow classification. These results for assemblage level indicators suggest some adjustment in the composition of fish assemblages at certain regulated/supplemented sites over time, as predicted by ELOHA if flow is a major driver of aquatic biodiversity patterns.

Three of 11 composite fish assemblage level metrics showed significant differences between regulated/supplemented and unregulated sites within HFCs. Total species richness and native species richness were significantly greater (21% and 29%, respectively) in regulated/supplemented sites compared with unregulated sites in HFC 3. This class consisted of 19 gauges located mostly in the Mary and Logan–Albert River catchments, with sites on the Logan and Nerang Rivers influenced by dams immediately upstream.

Higher species richness at regulated/supplemented sites was associated with Gower metric values between 0.05 (Logan River) and 0.24 (Nerang River). Total species richness includes both native and alien species, suggesting that alien species may have inflated the values of total species richness. However native species richness was even more elevated than total species richness, indicating that more native species were associated with sites below these dams.

Aquatic vegetation composition was also affected by flow regulation at sites downstream of Hinze Dam on the Nerang River. However, it is suggested above that these vegetation differences could be due in part to the far coarser substrates of the Nerang River compared to other Reference sites in RFC 5, which have sandy substrates. Riparian vegetation below Hinze Dam has also undergone a shift in assemblage structure but only one species (*Streblus brunonianus*, Whalebone tree) distinguished between the regulated/supplemented and unregulated sites.

Given that sites on the Nerang River have undergone changes in both riparian and aquatic vegetation, it is perhaps not surprising to find that fish assemblages also differed from the Reference condition for RFC 5. Environmental factors driving differences in vegetation composition below Hinze Dam (e.g. substrate composition) may also have affected fish assemblages, as would the actual riparian and aquatic vegetation structure at these regulated/supplemented sites.

Non-migratory species richness was almost double (98% higher) in one regulated/supplemented site (Six Mile Creek regulated/ supplemented by Six Mile Creek Dam) when compared with unregulated sites in HFC 4. Again it is notable that one of the greatest differences in flow regulation effect on aquatic vegetation also occurred at the site on Six Mile Creek where overall flow regime change has been relatively slight (0.052).

Previous studies have found that different fish species have inconsistent associations with anthropogenic flow regime change across large spatial scales (e.g. Gehrke et al 1995; Gehrke and Harris 2001), which suggests that patterns in fish population densities may only ever be weakly influenced by flow variability when placed in the context of all of the factors that fish experience throughout many generations.

Almost all of the fish species sampled in this study have wide geographic distributions to the north and south of the study area, and therefore have persisted in rivers that differ very broadly in their flow regimes (Pusey et al. 2004; Kennard et al. 2010). It is possible that the flow regime changes measured in SEQ fall within the full range of hydrologic conditions experienced over historical time. A capacity to tolerate or recover from such changes may be embedded in the genome of a species across its broader geographic range.

Hypothesis 3

Hypothesis 3 stated that fish population and assemblage indicators will vary predictably along gradients of flow variability and change in flow regime characteristics (flow regulation). This hypothesis is only partially supported by the results of the study.

Species richness at each sampling site was significantly related to the variability (CV) of mean daily flow during the 4 years before sampling, to HSNum, and to the constancy and predictability of monthly flows (estimated for the 15 years before sampling). These flow metrics were associated with gradients of fish assemblage structure in multivariate analyses, but further analysis showed their individual links to fish assemblage richness.

The relationship of species richness to the variability (CV) of daily flows provides tentative support for the idea that fish species richness may be maximised at both low and high levels of disturbance (i.e. daily variability of flow), with many sites supporting lower species richness at moderate levels of disturbance. In the lower range of disturbance values (low CVdaily flow), more species may be able to persist because available aquatic habitat is relatively stable, whereas at high levels of disturbance, variable water levels may create a wider range of opportunities for members of the fish assemblage in SEQ streams. Some of the species included in this analysis were alien, but the numbers of alien species per sample were typically nil or very low, and therefore unlikely to have inflated estimates of species richness.

From similar analyses there is an indication that the number of alien fish individuals per site increased with increase in the number of zero flow days over the 4-year antecendent flow period. However this positive relationship is driven by a small number of extreme values. One interpretation is that the alien species present are more tolerant of zero flow spells than native species and so persist at the expense of native individuals.

The ratio of the number of species to total individuals in a sample appears to decline with increasing mean daily flow, suggesting that fish assemblages were stressed when sites had been exposed to higher daily flows over the 4-year antecendent flow period. High abundance of a few species is an indicator of stress in pollution studies and may be a useful indicator of stress when daily flows have been elevated during the length of life of short-lived fish species.

A similar trend is evident between the ratio of the number of species to total individuals in a sample and the 1-year annual return interval flood (ARI_1yr). As the number of such floods increased over the 15-year antecendent flow period, fewer species but many individuals were present, suggesting that this fish metric may be a useful indicator of stress associated with more frequent floods.

The overall gradient of flow regime change across the study area, as expressed by the Gower metric (Chapter 3), was not reflected in any strong patterns of fish assemblage responses to flow regulation (supplementation). This result may reflect the relatively coarse nature of this composite metric, and does not mean that flow alterations have had no effect in the study area. It was expected that individual flow metrics and combinations of flow metrics might show effects on fish assemblages, and on the individual species that drive patterns of assemblage structure.

Although Duboulays rainbowfish (*Melanotaenia duboulayi*) did respond significantly to flow regime change, patterns in abundance were highly variable and there was no predictable response to flow regime change. Pacific blue-eye (*Pseudomugil signifier*) was too patchily distributed to make a similar statistical test worthwhile. Long-finned eels (Anguilla reinhardtii) were widespread, but again there was no predictable relationship between eel abundances and gradients of change in flow magnitude (mean daily flow), flow variability (CV of daily flow) and flow regime harshness (percentage of zero flow days) caused by dams in the study area.

As noted above, some of the greatest differences in flow regulation effect on fish occurred at sites downstream of dams where overall flow regime changes have been relatively minor on the Gower scale, such as Six Mile Creek. However, some individual metrics have changed markedly at this regulated/supplemented site.

The mean annual 1-day, 3-day, 7-day and 30-day minima are in the range 50–100% lower than at Reference sites, and the low spell duration is 100% higher below Six Mile Creek Dam. Such large reductions in low flow levels and the huge increase in duration of low flows could bring about the 77% reduction in densities of Pacific blue-eye (*Pseudomugil signifier*) at this site.

It is notable that the two fish species most affected by flow regulation (rainbowfish and Pacific blue-eye) are those inhabiting the upper levels of the water column and using aquatic vegetation for shelter and as spawning sites (Pusey et al. 2004). Further work is warranted on the influence of flow and substrate characteristics, disturbance by high flows, the composition and levels of cover by aquatic vegetation, and fish responses.

An effect of flow regulation was found for the density of native fish relative to alien species; this metric was lower than expected at 2 regulated/supplemented sites and higher than expected at 10 regulated sites. This implies that a high relative density of alien individuals was rarely associated with flow regime change in the study area, in contrast to the common observation that the converse occurs (Bunn and Arthington 2002). Furthermore, when deviations from expected were plotted against the Gower metric, there was no obvious pattern of response of this metric to the overall degree of flow regime change.

An effect of flow regulation was also found for non-migratory species richness; this metric was lower than expected at 6 regulated/ supplemented sites and higher than expected at 6 regulated/ supplemented sites. When deviations were plotted against the Gower metric, there was no obvious pattern of response to the overall degree of flow regime change. Deviations of these two fish metrics (density of native fish relative to alien species, non-migratory species richness) from expected did not show any consistent relationships with individual flow metrics, for example the CV of daily flows and the mean number of days without flow.

Collectively, these results do not support the hypothesis that increasing change in individual flow variables will result in increasing biotic change. However, they show clearly that each dam has had a different effect on fish species richness and assemblage structure in terms of density of native fish relative to alien species and non-migratory species richness. Some of the patterns are consistent with expectations from the literature, but others are not. The common observation that alien fish species can flourish at regulated/ supplemented sites was upheld at only 2 sites of 12 studied.

It is possible that the range of flow extremes caused by dams in SEQ falls within the full range of hydrologic conditions experienced by various species of fish over historical time and wider geographic range, and during drought and flood periods. Thus many of the common species in the study area may have a high level of resistance to the types and degrees of flow regime change brought about by dams in the study area. They may also have well-developed resilience traits that enable populations to recover from hydrologic disturbances, such as low flows, increased number of zero flow days and flooding (Pusey et al. 1993).

The difficulty in developing predictive relationships between flow metrics and ecological responses may be due in part to uncertainty in the estimation of hydrologic metrics used as predictor variables in flow–ecology response models (Pusey et al. 2009). If there are inherent uncertainties in the estimate of important flow metrics, such as very low and high flows, the effort to identify flow–ecology relationships may be compromised. Consequently, steps to minimise uncertainty in flow data could result in better detection of ecological responses to different aspects of the flow regime, and to flow regime change by dams.

8.4.3 Implications

Hypothesis 1 that the structure and composition of fish assemblages in the SEQ region will be influenced by interactions between flow history, natural catchment characteristics, in-stream habitat factors and anthropogenic disturbances is supported by the results of this study.

Across all three sampling periods, flow variables explained at most 20.34% of variation in fish assemblage composition, while around 30% of spatial variation in assemblage patterns was associated with gradients in climatic factors, catchment land use, reach scale geomorphology and habitat structure. This finding (echoed in results for riparian and aquatic vegetation) implies that the ecological roles of flow are contingent upon climate, landscape setting and land use disturbance regimes. Therefore environmental flows cannot be expected to achieve improvements in fish assemblages (and stream ecosystem health) when managed in isolation from catchment and channel processes and disturbances, such as agricultural land use patterns and intensity. The condition of the catchment surrounding each stream reach must also be managed to achieve healthy stream/ river ecosystems.

The strong association between climatic variables (temperature and rainfall) and fish assemblage structure suggests that fish in streams of the region could be vulnerable to climate change (e.g. rising temperatures, low flows and longer dry spells). If land use and water abstraction patterns also change over time, it can be expected that stream ecosystem health will deteriorate as a consequence of the interactions and synergies between climate change, land use and flow regime change. Ongoing monitoring and further research will be needed to track trajectories of change and to determine the best mix of adaptation strategies to achieve improvements in stream ecosystem health.

Hypothesis 2 stated that the structure and composition of fish assemblages will differ in a range of biological metrics across pre-development (unregulated) and Historic (regulated/supplemented) flow regime classes, and between regulated/supplemented and unregulated sites. This hypothesis is supported by some of the results of the study.

Seven univariate metrics of fish assemblage structure (species richness, non-migratory species richness, total fish density, native density, alien density, non-migratory density and species density) differed significantly across the RFCs. This result means that there is merit in considering these flow classes as distinctive management units with different fish assemblage characteristics that should be maintained as a component of stream ecosystem health in SEQ.

The SEQ Ecosystem Health Monitoring program (EHMP) places a strong emphasis on protection of fish assemblages and biodiversity (Kennard et al. 2006; Bunn et al. 2010). This ELOHA study provides support for evaluating monitoring results in relation to the different flow classes of streams identified here for the SEQ region.

None of the univariate metrics of fish assemblage structure (species richness, non-migratory species richness, total fish density, native density, alien density, non-migratory density and species density) show statistically significant differences between regulated/supplemented and unregulated sites within any of the RFCs.

However, other findings indicate that fish assemblages could be on a trajectory of adjustment to flow regime change brought about by the presence of dams. Two metrics of fish assemblage composition (Sorensen presence–absence similarity and the modified Gower assemblage similarity metric) were significantly different between regulated/supplemented and unregulated sites in some RFCs. These were also different between regulated/supplemented and unregulated sites within some HFCs, but less so than between regulated/supplemented and unregulated sites within the RFCs.

These results suggest there has been some adjustment in the composition of fish assemblages at certain regulated/supplemented sites since dams were constructed and operated in ways that alter downstream flow regimes. The patterns observed conform to predictions of the ELOHA framework, in suggesting that impacts of flow regime change may be major drivers of aquatic biodiversity patterns.

Strict criteria for selecting fish indicators of flow regime change were that a metric should differ significantly between Reference (natural) regulated/supplemented and unregulated sites, yet show no significant differences between regulated/supplemented and unregulated sites under Historic flow regimes. Duboulays rainbowfish (*Melanotaenia duboulayi*) was the only species ('indicator') to meet both criteria.

Densities of Duboulays rainbowfish (*Melanotaenia duboulayi*) were significantly greater in regulated/supplemented sites when compared with unregulated sites within RFC 2. The frequency of occurrence (in >75% of all samples) and response to flow conditions and alterations adds strength to the suggestion that this species may be a useful indicator of dam impacts in SEQ.

Long-finned eels (*Anguilla reinhardtii*) and Australian smelt (*Retropinna semoni*) were also common but did not show equivalent responses to flow regime change. Pacific blue-eye (*Pseudomugil signifier*), a less common species, did show some marked responses to flow regime change at some sites, but responses were not fully concordant with the criteria used to identify useful indicator species or metrics. If these criteria were to be relaxed, Pacific blue-eye could be a useful indicator species

Hypothesis 3 stated that fish population and assemblage indicators will vary predictably along gradients of flow variability and change in flow regime characteristics (flow regulation). This hypothesis is partially supported by the results of fish analyses.

Several analyses indicate that species richness, and probably also assemblage composition, was influenced by the variability (CV) of daily flows. As this flow metric was increased below some dams and decreased below other dams across the study area, it may prove to be an important flow variable for future management. Both low and high levels of daily flow variability appear to affect species richness.

This study has found depressed species richness and inflated abundance of some fish at sites with elevated daily flows. This type of pattern is an indicator of stress in pollution studies, and may be a useful indicator of stress in rivers regulated by dams. A similar trend was evident between the ratio of the number of species to total individuals in a sample and the 1-year annual return interval flood (ARI_1yr). As the number of such floods increased, fewer species but many individuals were present, suggesting that this fish metric may also be a useful indicator of stress associated with more frequent floods.

The number of alien fish individuals per site increased with increase in the number of zero flow days over the 4-year antecendent flow period. This finding suggests that the alien species present were more tolerant of zero flow spells than native species and so persisted at the expense of native individuals.

In summary, a few fish response variables that are easy to estimate from fish surveys – species richness, native fish abundance and alien fish abundance – appear to have merit as indicators of daily flow variability, floods and flow regime change in streams of SEQ.

The lack of any gradients of biotic response to overall flow regime change (Gower metric) or to any individual flow metrics may be a function of the relatively low level of overall flow change, the relative recency of dams in the region (constructed or altered 10–50 years ago), and the low number of more strongly regulated/supplemented sites available for testing this relationship (six). It was not feasible to test for gradient relationships within each flow class for the same reason – lack of replication of regulated/supplemented sites.

However, this study has demonstrated that dams have had impacts on the fish fauna of streams in SEQ. Given the evidence of widespread alterations to flow regimes by dams (and other water-related disturbances) in SEQ (Chapter 3), and the likelihood of further flow regime change by water resource development, increasing demands for water and climate change, it is recommended that flow variability and flow regulation impacts should be incorporated into the ongoing monitoring of stream ecosystem health.

The present study assessed patterns in fish abundance and broad assemblage level indicators such as species richness as a first step in this ELOHA trial, because such indicators are readily compared with other studies of river condition and ecosystem health (e.g.

Kennard et al. 2005, 2006) and information from other regions where knowledge of fundamental flow-related species biology is poor.

In a process similar to this ELOHA study, Kennard et al. (2005, 2006) trialled a range of fish indicators and found that three (observed versus expected native species richness, percentage of native species expected and percentage alien individuals) strongly reflected a disturbance gradient based on in-stream habitat condition in the SEQ SEQ region. These fish indicators have been incorporated into the SEQ EHMP.

This ELOHA study provides support for the sensitivity of species richness and proportion of alien species abundance to flow variability, and potentially to flow regime change. The habitat disturbance gradient studied by Kennard et al. (2006) was more pronounced than the flow regime gradient studied here, and it is possible that a more severe gradient of flow regime change would substantiate the utility of the EHMP fish indicators or the similar ones studied here.

Consideration of the utility of incorporating fish (and riparian and aquatic vegetation) metrics into the SEQ EHMP as indicators of flow regime change is strongly recommended. The monitoring of stream ecosystem health must take the landscape setting into account, on the premise that the whole catchment and river system represents the ecological unit to be monitored and managed for ecological health.

Stream health monitoring programs in SEQ should seek to collect biotic response data along gradients of flow regime change, environmental flow allocations and land use (following a common geographically referenced framework derived from this ELOHA study and the SEQ EHMP), to enable description of the relationship between catchment attributes and disturbances, flow characteristics and ecological outcomes.

Streams in the study area experienced several high flow events during the study period, and some were flooded again in early 2011. The effects of the 2011 floods on the biotic assemblages of SEQ streams should be studied to determine if flooding has implications for the health of the fish fauna of the region.

A comparison between the observed effects of flow regime change brought about by dams and the effects of recent floods could be instructive. It could guide stream rehabilitation following flooding and provide input into the management of environmental flow allocations over the longer term, taking the beneficial and destructive effects of flooding into account.

9. Synthesis

This project (*Hydro-ecological relationships and thresholds to inform environmental flow management and river restoration*) represents the first attempt in Australia to explore the scientific implications of using the framework ELOHA (Poff et al. 2010) as a means to understand how flow regime alterations affect rivers and their riparian and aquatic biota at a regional scale, and to interpret findings in terms of environmental flows and water management.

The following extract from the original submission to the NWC conveys the intent of the project:

> 'This project will provide a synthesis of hydro-ecological relationships in unregulated rivers of coastal and inland Queensland, and undertake field studies comparing unregulated and regulated rivers to identify thresholds of habitat and ecological response to flow regime alteration that will inform environmental flow management in rivers with contrasting flow regime characteristics and particular human 'footprints' (such as extent/type of rural and urban; riparian degradation, water quality impairment, alien species). The outcomes of the project will inform environmental flow management in rivers that may be regulated in the future, as well as the restoration of rivers that have been/are still regulated.'

Clear flow–ecological response relationships are needed to form the basis for the ELOHA environmental flow decision-making framework (and every other environmental flow method). These relationships can be applied to predicting and communicating the ecological response to a change in flow regime, both in terms of restoring ecologically important elements of the natural flow regime in regulated/supplemented rivers, and also predicting the ecological consequences of future changes (arising from new water resource developments, increasing water abstraction/diversion and climate change).

The findings of this ELOHA trial in SEQ are synthesized in terms of project objectives in two categories – environmental and educational. These sections are followed by a summary of key project outcomes and recommendations.

9.1 Environmental objectives

9.1.1 Hydrological regimes of unregulated river basins in south-east Queensland

Objective 1: *To provide an analysis of the hydrological regimes of unregulated (Reference) river basins in Queensland that are relevant to the study areas selected for this project.*

Flow regime classification represents the first step in the ELOHA framework – the building of the 'hydrological foundation' to underpin analysis of hydro-ecological relationships in rivers with 'natural' flow regimes and in rivers with flow regimes altered by dams and weirs.

Classification of the hydrological regimes of unregulated river basins (termed Reference flow regimes) in SEQ was based on modelled pre-development flow data derived from an ''IQQM (Simons et al. 1996). The IQQM (Reference) classification allowed the characteristics and variability of 'natural' flow regimes within the study area to be described, while the second classification (Objective 2, below) revealed changes in the flow regimes of localities affected by dams and weirs, abstraction and land use change. Flow classification was undertaken according to existing protocols for calculation and selection of flow metrics (Olden and Poff 2003; Kennard et al. 2010a,b).

Classification of Reference flow metrics identified six RFCs, which were separated along a gradient of discharge magnitude (primarily *high* discharge magnitude) and discharge variability (represented by the coefficient of variation of mean daily discharge). RFCs with high discharge variability had low magnitude discharge per unit of catchment area. A second discharge gradient, represented by low discharge magnitude and spell durations, was also present. Thus the main driver of spatial variability in Reference flow regime patterns in SEQ is discharge magnitude.

Each RFC contained IQQM nodes from at least three catchments in the study area. Nonetheless, a geographic element was present in the Reference classification, for example RFC 4 included IQQM nodes in the west and north-west of the study area. This region has lower rainfall than the eastern part of the study area and consequently RFC 4 is characterised by long periods of low flow and low discharge magnitude per unit of catchment area. In comparison, RFC 5 included IQQM nodes in the eastern (coastal) part of the study area. This part of the study area is characterised by high rainfall and hence RFC 5 is characterised by relatively high discharge magnitude per unit of catchment area.

The Reference flow classification had some concordance with the continental flow classification of Kennard et al. (2010a). The continental flow classification identified four flow classes in SEQ – two perennial flow classes and two intermittent flow classes. The majority of IQQM nodes which had analogues in the continental flow classification belonged to continental flow class 7 (Intermittent–Unpredictable).

Direct comparison of these classifications is difficult as only 30.5% of the IQQM nodes in the Reference flow classification had an equivalent site in the continental flow classification. In attempting to align the Reference classification with the continental classification two flow classes were defined as 'Perennial', three flow classes as 'Intermittent' and one flow class as 'Intermittent–Unpredictable'. Implications for extending the geographic scope of ELOHA study results are discussed further below.

9.1.2 Hydrological regimes of regulated river basins in south-east Queensland

Objective 2: *To provide a quantitative assessment of how the flow regimes of regulated rivers in these study areas have been altered by water infrastructure and the array of types/degrees of flow regulation. Flow metrics of relevance to ecological responses to flow alteration will be included in these analyses.*

A second flow regime classification based on gauged flow data (termed Historic flow data) identified five HFCs. Thus the most obvious flow regime change (from Reference to Historic) is the loss of a flow class under the Historic flow regime.

Six flow metrics discriminated between the HFCs. These flow metrics described discharge magnitude, the timing of high and low spells and discharge variability (discharge constancy). The principal discharge gradient was high discharge magnitude and the secondary gradient was related to spell duration and low discharge magnitude. Thus the main drivers of spatial patterns in the Reference and Historic flow classifications were similar.

Comparison of the Reference and Historic flow classifications showed several similarities in classification structure despite differences in the number of nodes/gauges used in each classification and the number of flow regime classes identified. Two HFCs had direct analogues in the Reference flow classification.

These were HFC 2 (equivalent to RFC 4, gauges located in the drier parts of the study area) and HFC 5 (equivalent to RFC 5, small coastal catchments in high rainfall areas). HFC 1 consisted almost entirely of gauges with flow regimes influenced directly or indirectly by flow regulation and hence had no analogue in the Reference flow classification. The principal flow class changes associated with the Historic flow regime can be summarised as:

- loss of two RFCs (RFCs 3 and 6, although none of the IQQM nodes in RFC 6 had a corresponding gauge in the Historic flow classification)
- re-distribution of RFC 1 nodes into two HFCs (mainly HFCs 3 and 4)
- creation of a perennial HFC comprised of gauges influenced by flow regime alteration and one unregulated creek (Teewah Creek) with a relatively high groundwater component to discharge.

All HFCs contained at least one gauge with a flow regime influenced by a dam.

The geographic extent of flow regime alteration by dams and weirs (and possibly land use) in the study area is considerable. However, the degree of overall flow regime alteration from Reference conditions is relatively minor, with a maximum Gower dissimilarity of 0.25 compared to 1.0 for perfect dissimilarity (total change in every flow metric). Yet all streams and rivers in the study area have been subjected to some level of flow regime change. In general the greatest flow regime changes in the study area have occurred in streams/rivers downstream of dams (i.e. Nerang River, Reynolds Creek, Yabba Creek, Lockyer Creek, Brisbane River and Burnett Creek).

However, three of the 11 gauges with the greatest flow regime change are not downstream of dams. These are gauges on Running Creek, Mudgeeraba Creek and the South Pine River. Land use changes in these systems are extensive. The primary land use change in the Running Creek catchment is agriculture, while urbanisation is the dominant land use for catchments of Mudgeeraba Creek and the South Pine River. Furthermore, the presence of dams does not necessarily imply extensive flow regime change (e.g. Six Mile Creek has a Gower metric of 0.052, indicating relatively minor change from the Reference flow regime).

Across the region, 57% of comparisons between individual Reference and Historic flow metrics showed a change in metric value of 20% or less, and 37% of comparisons showed a change in metric value of 10% or less. Magnitude metrics have undergone the greatest change across the study area, in particular low spell duration (LSDur), with most gauges showing an increase in LSDur under the Historic flow regime. This marked change in LSDur could represent the effects of dams on downstream flows, or levels of water extraction from supplemented flow, or effects of increasingly dry conditions over the study period, or all three processes.

Mean rates of rise and fall have also increased substantially when compared to the Reference value. In contrast, mean monthly discharge metrics and MA3-90dayMin have decreased in value relative to Reference flow values. Metrics associated with frequency, duration, variability and timing have undergone relatively minor change from Reference condition.

The results of flow classifications and analysis of flow regime change indicate that every dam in the study area has altered the flow regime in a different way. Thus there is no replication of each type and degree of change in flow regime, as recommended in the design of ELOHA studies (Arthington et al. 2006). Instead, a range of different changes has occurred according to the characteristics of each dam's location, the storage capacity of the dam, water release strategies and downstream water abstraction practices.

Therefore, drawing out generalisations about management strategies is difficult since each dam appears to have generated a unique flow regime downstream. If every dam has different ecological effects, then it follows that ecological restoration by re-regulation of the flow regime (i.e. by providing environmental flows) will be likely to take a different form. The environmental flows for each system will also depend on the particular ecological changes downstream and the overall goals associated with environmental water management.

In summary, comparisons of Reference and Historic flow regimes in SEQ suggest that flow regime changes brought about by dams and other factors within the study area have been relatively minor. This is not to say that changes of a relatively low magnitude will have no ecological effects. It is possible that a 10–20% change in some metrics could have a significant impact on certain ecological indicators, and this is precisely what the ELOHA method seeks to determine.

9.1.3 Literature reviews

Objective 3: *To provide a synthesis of knowledge of ecological responses to flow regime alteration in selected rivers within the study areas selected for the project. This will be based on past research supported by LWRRDC, Queensland DNRM, Queensland Fisheries, CRCFE, monitoring studies, Ph D theses, benchmarking studies for WRPs and consultancies.*

Appendices 1–3 to this Scientific Report present three literature reviews. These reviews summarise literature on the ecology of riparian and aquatic vegetation and fish in SEQ streams and rivers, with an emphasis on the roles of hydrology and the impacts of flow regime change.

From these reviews and knowledge derived from personal research, hypotheses have been developed as the main focus for field research during the trial of the ELOHA framework. Material contained in these review is incorporated into the final Scientific Report, as appropriate, and will be used in several future publications. The literature reviews for SEQ are:

- Appendix 1: Riparian vegetation–flow relationships and responses to flow regime alteration: a review of evidence from south-east Queensland streams. C. S. James and A. Barnes.
- Appendix 2: Aquatic vegetation–flow relationships and responses to flow regime alteration: a review of evidence from south-east Queensland streams. S.J. Mackay.
- Appendix 3: Freshwater fish—flow relationships and responses to flow regime alteration: a review of evidence from south-east Queensland streams. A.H. Arthington and D. Sternberg.

An additional component (Appendix 4) of the ELOHA study has involved preparing a synthesis of past research on the ecology of fish in relation to hydrology in Cooper Creek. The intent of this review was to inform DERM's review of the Cooper Creek Water Resource Plan, and to support the proposal to designate Cooper Creek as a Wild River, as follows:

- Appendix 4: Extreme hydrologic variability and the boom and bust ecology of fish in arid-zone floodplain rivers: a case study with implications for environmental flows, conservation and management. *Ecohydrology* 4: 708–720. A.H. Arthington and S.R. Balcombe (2011).

9.1.4 Field research program

Objective 4: *To design a field research program that will identify how existing flow regime alterations in the study areas have impacted on habitat structure/heterogeneity, and the structure, dynamics and productivity of biological assemblages, life history strategies (aquatic plants, invertebrates, fish) and food web structure.*

Study area

The ELOHA field study was conducted in the South Coast, Logan–Albert, Brisbane, Pine–Caboolture, Maroochy, Noosa and Mary River catchments of coastal SEQ, Australia. This region was chosen for several reasons. The ecology of streams and rivers in the region has been investigated by staff of the Australian Rivers Institute, as it has a relatively dense stream gauging network and a variety of flow regime types.

The water resource needs of the region have been investigated (Moreton Basin WRP, Logan–Albert WRP and Mary Basin WRP), thus the results of the project will be of relevance to the management of these catchments and to the monitoring and review of WRPs over the next 5–10 years.

There is considerable natural environmental and geological variation across the study region (Bridges et al. 1990; Ellis 1968; Murphy et al. 1976; Whitaker et al. 1980). Distinct topographic regions are identifiable within the region with coastal plains, river floodplains and estuaries in the east, and foothills and mountains with plateaux over 300 m above mean sea level to the west, north and south of the study region.

The climate is subtropical and dominated by summer rainfall with warm summers and mild winters, but sits adjacent to the temperate/subtropical transitional zone. The area also exhibits a strong rainfall gradient with a decrease in rainfall in an east to west direction across the study area (Bridges 1990).

The Brisbane and Mary River catchments comprise approximately 72% of the total study area of 32 000 km^2. However, the Noosa and Maroochy catchments have higher mean annual runoff per unit of catchment area (560.8 and 782.8 $Ml.year^{-1}.km^{-2}$) than the Brisbane (82 $Ml.year^{-1}.km^{-2}$) and Mary (213 $Ml.year^{-1}.km^{-2}$) catchments.

This reflects rainfall gradients across the region. The volume of water held in storage varies considerably among catchments. The greatest volume of water is held in the Brisbane River catchment due to the presence of Wivenhoe and Somerset Dams (Chapter 3). The storage capacity of dams and weirs in the study area is approximately 38% of the mean annual runoff.

Study design

The types and degree of ecological impact on rivers brought about by flow regime alteration can be examined in at least five different ways:

- before–after comparisons of control (Reference) and impact (regulated) site (BACI designs)
- referential approach (comparison of impacted sites with unimpacted Reference sites)
- comparison of Observed vs modelled Expected ecological attributes (Kennard et al. 2006a,b)
- experiments on the mechanisms underlying patterns identified by other methods
- multiple lines and levels of evidence (Downes et al. 2002).

This project has focussed on the referential approach to establish any potential gradients of response to flow regime alteration within and across hydrological classes. A series of principles and criteria for the selection of field sampling sites was developed (Chapter 4). Site selection was based primarily upon the Reference flow classification (Chapter 3).

The flow conditions preceding biotic sampling were expected to be key predictors in statistical analyses/models and it was therefore essential that discharge data were available for each study site for the duration of the field program. Thus all study sites had to be an appropriate distance to *currently* operating stream gauges. The section of stream or river upstream and downstream of an individual stream gauge, for which the discharge recorded at the gauge could be considered representative, is defined as a *study reach*. Study reaches were defined by local topography and the presence of major inflows upstream or downstream of individual stream gauges.

Following delineation of study reaches, a subset of reaches was selected for which suitable study sites were expected to be found. This preliminary selection was based on ease of access, the presence of suitable riparian vegetation (see below), the location of tributaries near gauges, and workplace health and safety issues. Study reaches in the main channel of the Brisbane River were not considered since the Brisbane River was the focus of a recent environmental flow study (Arthington et al. 2000), the riparian vegetation of the main channel is highly modified as a result of land management activities (McCosker 2000), and the river downstream of Wivenhoe Dam is difficult to survey due to the high volume of water released constantly from the dam.

However, study reaches were selected downstream of the remaining major dams in the study area. Site selection was also limited by depth requirements for sampling aquatic vegetation and for sampling fish by backpack electrofisher (maximum suitable depth 1–1.5 m). The selection of specific field sampling locations and sites within reaches was made using aerial photos, satellite imagery, discussion with landholders, previous field experience and field visits.

Field sites were defined as a stream length of 100 m (maximum) as this length allowed multiple vegetation transects to be conducted within a site, and usually incorporated multiple in-stream habitats (riffles, runs and pools) whilst minimising variation due to changes in stream morphology, geology and neighbouring land use. Sites were preferably located close to the gauge or IQQM node (upstream or downstream) to minimise differences between flow at the gauge and the actual field site, as described above.

Two sites were surveyed within each reach. Sites within a study reach were a minimum of 2 km apart to ensure site independence as far as possible, whilst maintaining a reasonable proximity to the gauge. A further important consideration, particularly for the riparian vegetation element, was current and historical land uses. Wherever possible the sites selected were not currently grazed, had not been cleared in the last 20–30 years and were not subject to regular burning.

These criteria were stipulated to reduce other influences on flow–ecology relationships, particularly for the riparian vegetation component which was likely to be strongly influenced by such factors, but also for in-stream biota and process that are influenced, in turn, by riparian vegetation structure and processes. Riparian impacts at potential sites were inferred from observation and by discussion with landowners. Grazed sites were unavoidable in some reaches (particularly given other restrictions on site selection) due to the widespread extent of grazing in some catchments and lack of livestock exclusion from the riparian zone (e.g. Burnett Creek and Teviot Brook).

Following hydrologic classification, regulated /supplemented reaches (i.e. stream reaches close to a stream gauge or IQQM node located downstream of a dam or weir) within SEQ were identified and their RFC and HFC membership determined.

Regulated reaches were selected where hydrological analyses (Chapter 3) indicated that flow regimes deviated from natural and for which appropriate non-regulated Reference sites could be found. Limitations on data availability (both gauged and modelled) precluded the inclusion of some regulated reaches in these analyses. Once the flow class membership for each regulated reach had been established for both Reference and Historic flow classifications, Reference reaches were selected. Each Reference site was located in a stream section proximal to a gauge or IQQM node not subject to significant flow regime alteration.

Reference sites drawn from the sites falling within the Reference classification enabled the effects of flow regime alteration on ecological systems common to that class to be assessed. In the ELOHA framework, if flow alteration has an impact Reference and regulated sites within the same hydrologic class should differ ecologically.

Reference sites drawn from the sites falling within the Historic classification enable the effects of flow regime alteration on ecological systems to be assessed in a different way. If a regulated site falls within an HFC that also includes unregulated sites, then in theory, these regulated sites should not differ ecologically, because they are hydrologically similar and their ecological condition is assumed to be dependent upon the characteristics of the flow regime.

Both types of comparisions are needed to test the concepts Implicit in the ELOHA framework. There are caveats associated with these concepts, as explained within each chapter of the report and below.

9.1.5 Field research program and hypotheses tested

Objective 5: *To implement the field research program in selected rivers of contrasting flow regime type and range/degree of hydrological alteration by studying habitat and ecological responses to gradients of change in each flow characteristic.*

The field study was comprised of three main components, riparian vegetation, aquatic vegetation and fish response to flow variability and flow regime change in SEQ. These ecological 'components' were selected for several reasons, a major one being that each is known to respond to flow variability and flow regime change (Poff et al. 1997; Bunn and Arthington 2002; see also literature reviews, Appendices 1–3). In addition, each component performs vital ecological roles and can be regarded as an ecological 'asset'.

Furthermore, riparian and aquatic vegetation and fish all contribute to the overall ecological 'health' of streams/rivers. Their inclusion in this study provided an opportunity to consider their roles as indicators of ecosystem health with particular reference to stresses related to flow regime change, and as indicators of other environmental factors, particularly land use and climate change.

Forty-four individual study sites associated with stream gauges were selected for sampling along 20 streams/rivers that reflected the major flow regime gradients in SEQ (Chapter 3). Riparian vegetation surveys were conducted at all forty-four sites whereas aquatic vegetation and fish surveys were conducted at 40 sites. Field and laboratory methods for each study component differed and are described in Chapters 6–8.

The following sections present 13 hypotheses tested during the field study.

Riparian vegetation

The ELOHA field study was designed to address hypotheses extracted from the literature review '*Riparian vegetation–flow relationships and responses to flow regime alteration: a review of evidence from South East Queensland streams* (James and Barnes, 2011, Appendix 1) and from the ELOHA framework. The following hypotheses were tested:

Hypothesis 1: *The structure and composition of riparian assemblages in the SEQ region will be influenced by stream flow.*

Stream flows are a major control on the distribution, abundance and diversity of plants on stream and river banks (Merritt et al. 2010). Here it is suggested that riparian plant distributions, abundances, diversity and variability in streams of SEQ will be governed largely by stream flow regimes. Whilst there is significant evidence to suggest links between stream flows and riparian vegetation for other regions of Australia and internationally, this link has not been made for riparian vegetation of SEQ. This hypothesis investigated the relative contributions of flow as a physical disturbance and as a resource provider in structuring riparian vegetation. A number of sub-hypotheses were tested:

- Flood and high flow disturbance are major controls on the composition and structure of riparian vegetation.
- Baseflow and low flows are major controls on the composition, structure and lateral distribution of riparian vegetation.
- Variability in stream flows will drive variability in riparian vegetation.

Hypothesis 2: *The structure and composition of riparian assemblages in the SEQ region will be influenced by interactions between flow variables, and other natural factors and anthropogenic disturbances.*

This hypothesis recognizes that riparian vegetation is likely to be influenced by interactions between flow variables and other natural environmental factors and anthropogenic disturbances. It was predicted that assemblages at a site scale (overall abundances, species richness, diversity and community composition) would be strongly controlled by landscape variables and that the influence of specific flow variables on local riparian plant distributions would be conditional on regional settings.

Under this hypothesis the focus was on landscape variables that are likely to influence the importance of the physical effects of flood disturbances (i.e. topography, stream gradients) and moisture availability (climate) in driving local scale riparian vegetation patterns.

Hypothesis 3: *Riparian assemblage structure in streams of SEQ will differ across the Reference and Historic flow classes.*

This hypothesis addresses the basic premise of the ELOHA framework that different hydrological classes will have different riparian assemblages because stream flows are the major control on the distribution, abundance and diversity of plants on stream and river banks. Tests of this hypothesis were based on the Historic (gauged) and Reference flow regime classifications as riparian vegetation community composition and structure are expected to respond to long-term flow regime characteristics.

Following on from this, it is suggested that if flow is a major driver of riparian vegetation patterns, then regulated sites should be DISSIMILAR to unregulated sites for a given RFC, Conversely, for a given HFC regulated sites should be SIMILAR to unregulated sites if flow is important.

Hypothesis 4: *Changes in stream flow regimes will alter the distribution, abundance and diversity of plants on stream and river banks.*

This hypothesis focused on responses of riparian vegetation to flow regime change. A review of evidence for the impacts of flow regime change on riparian vegetation suggested a number of potential effects. The manner in which vegetation is likely to respond depends on the nature of the flow regime alteration. This study focused on the following predictions:

- Changes to near-stream vegetation density where flow regimes are altered.
- Reduction in the regeneration of native species in the bank full channel where high flows and flood disturbance are reduced.
- A reduction in the proportion of species characteristic of early (and intermediate) successional stages where flood disturbance is reduced.
- Reductions in species diversity where flow variability is reduced.
- Increased proportion of exotic species with flow regime change.

Aquatic vegetation

The ELOHA field study was designed to address hypotheses extracted from the literature review '*Aquatic vegetation–flow relationships and responses to flow regime alteration: a review of evidence from South East Queensland streams*' (Mackay, 2011, Appendix 2) and from the ELOHA framework. The following hypotheses were tested:

Hypothesis 1: *Streams with similar flow regime characteristics should be more similar in terms of aquatic vegetation assemblage metrics (abundance, species richness, and species traits) and assemblage composition than streams with different flow regime characteristics*.

This hypothesis is a statement of a key premise of the ELOHA framework. If this premise holds true in SEQ, then differences in aquatic vegetation (species composition, abundance, functional attributes) should be evident between flow classes identified by flow regime classification (Chapter 3). Tests of this hypothesis were based on the Historic flow regime classification (i.e. the classification based on stream gauge data) as aquatic vegetation is expected to respond to recent short-term antecedent flow regime characteristics and short-term flow events.

Hypothesis 2: *Aquatic vegetation abundance will vary inversely with discharge magnitude*.

Hypothesis 1 assumes that the flow regime will be an important driver of vegetation assemblage structure. Hypotheses 2–4 specify individual components of the flow regime that may drive vegetation assemblage patterns. Hypothesis 2 is related to Hypothesis 1 in that discharge magnitude (standardised by catchment area) is the primary difference among HFCs. Evidence to support this hypothesis is presented in Appendix 2.

Hypothesis 3: *Macrophyte abundance will vary inversely with flood frequency*.

In New Zealand catchments aquatic macrophytes are limited to rivers with 13 or less high flow disturbances per year (Riis and Biggs 2003). Flood frequency, like discharge magnitude, is a convenient hydrologic metric for examining aquatic vegetation patterns in relation to individual flow metrics. Flood frequency is a relatively easy metric to calculate when compared with hydraulic metrics, which may require channel surveys. However, flood frequency is not easily transferred to other river catchments since the effects of flooding are related to geomorphology and channel form, which may vary between catchments.

Hypothesis 4: *Aquatic vegetation abundance will be positively correlated with discharge variability*.

The CV of mean daily discharge is a component of the disturbance axis of the aquatic vegetation conceptual model (Appendix 2) and there is evidence to suggest that the relationship between daily discharge variability and aquatic vegetation cover is positive. This is due to the fact that streams with high mean daily discharge variability tend to have conditions amenable to aquatic vegetation growth – long periods of low flow, often relatively low turbidity and relatively fine substrates (see also Riis et al. 2008).

Hypothesis 5: *Aquatic vegetation abundance (as biomass or cover) will be higher in regulated sites than unregulated sites, if flow regulation results in increased discharge stability or reduced frequency of substrate mobilisation*.

Changes in substrate stability may result from flow regulation since flood frequency and magnitude will be reduced downstream of dams. This may result in reduced frequency of substrate mobilisation and hence increased aquatic vegetation abundance. Previous work in SEQ has shown that flow regulation may result in increased aquatic vegetation cover downstream of dams, although this response may depend upon the extent of shading by riparian canopy cover.

Hypothesis 6: *Increasing degree of flow alteration from baseline condition will produce increasing degree of change in aquatic vegetation assemblages*.

The ELOHA framework assumes that increasing degree of flow regime change (from Reference condition) is associated with increasing degree of ecological change (Poff et al. 2010). Flow regimes downstream of dams in SEQ have undergone varying degrees of change from Reference condition (Chapter 3). Hence there is scope to examine changes in aquatic vegetation assemblages over a gradient of flow regime change.

Fish

The ELOHA field study was designed to address hypotheses extracted from the literature review '*Freshwater Fish–Flow Relationships and Responses to Flow Regime Alteration: a Review of Evidence from south-east Queensland Streams*' (Arthington and Sternberg, 2011, Appendix 3) and from the ELOHA framework. The following hypotheses were tested:

Hypothesis 1: *The structure and composition of fish assemblages in the SEQ region will be influenced by interactions between flow history, natural catchment characteristics, in-stream habitat factors and anthropogenic disturbances.*

Whilst the antecedent history of stream flows is recognised as one of the principal influences on stream ecology (Poff et al. 1997; Bunn and Arthington 2002), many other catchment characteristics are also important drivers of stream ecological processes and biotic assemblages. Understanding the influences of flow on ecological responses, given the underlying variability in environmental and anthropogenic activities across the study region, presents the first significant challenge of the ELOHA framework.

This component of the fish study sets the scene for all subsequent data analysis by quantifying the relative influence of short and long-term flow history on fish distributions (measured by presence–absence data) and fish assemblage composition (measured by CPUE).

Following Kennard et al. (2007), it is expected that the distribution of fish (i.e. presence–absence patterns) will be driven largely by landscape variables and long-term patterns of river flow regimes, whereas patterns in fish assemblage composition (relative abundance of species) will be more strongly influenced by in-stream habitat and short-term flow history. The unique explanatory power of flow variables should be revealed by tests of Hypothesis 1. The flow variables found to be of most significance for fish species and assemblages formed the basis for subsequent tests of flow alteration – ecological response relationships.

Hypothesis 2: *The structure and composition of fish assemblages will differ in a range of biological metrics across Reference (unregulated) and Historic (regulated/supplemented) flow regime classes, and between regulated/supplemented and unregulated sites over time.*

This hypothesis tests several basic tenets of the ELOHA framework. The first is that effects of flow regulation on fish assemblages will be apparent, but may vary within and among both the Reference (unregulated) and Historic (regulated/supplemented) flow regime classes. If selections of Reference sites (based on modelled pre-development flow data) for comparison with regulated sites within each RFC (IQQM) were sound, then differences can be expected between regulated and unregulated fish assemblages within each flow class that are due to flow characteristics alone, rather than other environmental factors. Furthermore, the fish response among flow classes may vary due to the particular characteristics of the Reference flow regime and the type and degree of flow regime change.

In contrast, if flow alone is the major driver of fish assemblage structure, differential responses to flow regime alteration can be expected across the RFCs (gauged) but similar responses are expected within these flow classes. According to the ELOHA framework, streams that are regulated in similar ways should show similar ecological responses, and they should also be more similar ecologically to unregulated streams that fall into the same HFCs. This hypothesis tested these concepts.

Assemblage structure and composition can be represented by many metrics reflecting interest in native versus alien species richness, the richness of trait groups, patterns of abundance and species composition. Testing 14 of these metrics served to highlight the most suitable fish 'indicators' for differentiation among flow regime classes and between regulated and unregulated sites.

Based on prior knowledge of patterns in fish assemblage structure associated with flow regime change (literature review, Appendix 3), it was expected that differences in population densities, assemblage-level metrics and assemblage composition between regulated and unregulated rivers would depend on the nature of the natural flow regime and the degree and type of flow alteration.

Fish metrics that showed significant differences between regulated and unregulated sites were considered as candidates for testing relationships along gradients of flow variability and flow regime change.

Hypothesis 3: *Fish population and assemblage indicators will vary predictably along gradients of flow variability and change in flow regime characteristics.*

This part of the project set out to visualise and test the strength of the relationships between flow variables identified as being associated with patterns in fish assemblage composition (from Hypothesis 1) and indicators of fish population and assemblage condition across flow regime classes (from Hypothesis 2) in SEQ.

It was expected that some indicators of fish assemblage structure (e.g. species richness, relative abundance of alien species, abundance of fish in particular guilds) would show responses to alterations in particular flow regime characteristics. Tests of Hypothesis 3 should reveal the characteristics of flow–fish response relationships, which may be linear, threshold or take some other form, or there may be no clear patterns of response to flow variability and flow regime change by dams and other flow-related factors.

Collectively, tests of the three hypotheses enable the potential of the ELOHA framework to be discussed as it relates to the ecology of fish in SEQ. Furthermore, any constraints and limitations of the framework can be explored in terms of its utility to support water and river health management, and its potential to inform adaptive responses to climate change.

9.1.6 Responses to flow variation across classes

Objective 6: *To identify thresholds (if there are any) or linear relationships of habitat and ecological response to flow regime alteration, with emphasis on responses of riparian vegetation, aquatic vegetation, and fish or other 'indicators' of ecosystem 'health' (structure, productivity, resilience)*.

This section gives a summary of ecological differences across the RFCs and HFCs, and between regulated and unregulated sites. Project objectives do not specifically address these questions, but they need to be discussed to flesh out the testing of hypotheses and the applicability of the ELOHA framework in SEQ. Relationships between ecological response and gradients of flow regime alteration are discussed in the next section.

Riparian vegetation

This study tested for differences in riparian vegetation (primarily trees and shrubs) assemblage structure among RFCs and HFCs and for effects of flow regime alteration on a range of riparian vegetation metrics based on life history stage, growth form, origin (native or exotic) and successional stage.

Analyses were conducted on all tree and shrub vegetation within the bankfull limit of each site, and near-stream vegetation only. Near-stream was defined as vegetation within 5 m of the waterline as this zone was found to contain chiefly only those tree and shrub species generally considered riparian and may therefore be influenced to a greater degree by hydrology compared with vegetation further away from the stream edge.

The results of the analysis of riparian vegetation provides mixed support for the ELOHA prediction that distinctive flow regime classes will have different ecological characteristics (Hypothesis 3).There were significant differences in 12 bankfull riparian metrics across the Reference flow classes but only one significant differences (for species density, D_SPECIES) across the Reference flow classes for the near-stream vegetation metrics.

Bankfull riparian tree and shrub assemblage structures also differed significantly across the Reference flow classes. Distinct near-stream tree and shrub assemblages however were not found across the Reference flow classes. Furthermore, both bankfull and near-stream tree and shrub assemblages were relatively poor predictors of Reference class membership.

It should be noted also that the Reference flow classes differed across a number of other non- flow related variables. In particular, all the climate variables differed significantly across the Reference flow classes and were also identified as important variables in the riparian assemblage and metric analyses.Evidence for the effects of flow regulation on riparian vegetation is also mixed.

Analyses revealed no significant differences in tree and shrub assemblages between regulated and unregulated sites for bankfull vegetation or near-stream vegetation overall (disregarding flow class). For riparian vegetation metrics, significant differences between regulated and unregulated were observed only for the riparian bankfull metrics D_SPECIES and BA_LATE when sites strongly impacted by flow regulation were compared (so excluding sites 12, 13, 15 and 16 which are only weakly impacted by regulation). No significant differences were found between any of the bankfull or near-stream vegetation metrics when all regulated sites were included in the analyses.

Support for Hypothesis 4 that sites should be DISSIMILAR to unregulated sites for a given RFC, was only found for RFC 5. Regulated sites on the Nerang River (sites 4 and 6) were significantly different in their tree and shrub assemblage structure to the unregulated sites in RFC 5. Furthermore, significant differences were not found between Nerang sites (4 and 6) and unregulated sites within HFC 3, supporting the hypothesis that, for a given HFC, regulated sites should be SIMILAR to unregulated sites if flow is important.

These results suggest that vegetation below Hinze Dam on the Nerang River has undergone a shift in assemblage structure and vegetation is now similar to that found in the lower discharge sites in the Mary and Logan catchments. However, indicator species analysis suggested that only one species (Streblus brunonianus, Whalebone tree) distinguished between the regulated and unregulated sites in RFC 5.

Other flow classes (RFCs 1 and 2 and HFCs 1, 3 and 4) were not found to differ significant in bankfull assemblage structure between regulated and unregulated sites and no significant differences in near-stream tree and shrub assemblages between regulated and non-regulated sites could be detected amongst the RFCs (1, 2 and 5) or HFCs (1, 2, 3 and 4).

An effect of flow regulation was detected for densities of reeds, rushes and sedges. Sites with flow regulation had higher densities of these plant groups than predicted from a model representing densities expected at unregulated sites. There is substantial national and international evidence to suggest that a reduction in high in-channel flows and flood flow components will result in the encroachment of vegetation into the main channel itself and bring about a subsequent reduction in active channel width.

The Mary River WRP (Brizga et al. 2004) suggests that riparian vegetation thickening may have taken place downstream of Cedar Pocket Dam on Deep Creek, while encroachment of riparian vegetation into the main channel has been reported on the Nerang River below Hinze Dam (Brizga et al. 2006b), on Reynolds Creek below Moogerah Dam (Brizga et al. 2006c) and on the Brisbane River below Wivenhoe Dam (McCosker 2000). ELOHA analyses suggest that these large herbaceous vegetation groups are denser in regulated streams. However, the Reference model was relatively poor with a low R^2 so this result should be treated with some caution.

Aquatic vegetation

Broad-scale aquatic vegetation patterns were not consistently related to differences in Historic flow regimes. In contrast, several pieces of evidence showed that aquatic vegetation patterns were influenced by flow regime changes resulting from flow regulation by dams in the study area.

Aquatic vegetation cover is often found to be higher downstream of dams, due to changes in flow regimes and/or changes in physical habitat (French and Chambers 1997). Reductions in flood frequency and magnitude that occur downstream of dams should favour aquatic vegetation growth due to reductions in scouring and substrate mobilisation (Riis and Biggs 2001; aquatic vegetation Hypothesis 5). However, aquatic vegetation cover was not significantly higher downstream of dams in the study area, compared with unregulated sites. While bankfull substrate stability was generally higher in unregulated sites, regulated sites were found to be subjected to periodic substrate mobilisation (as shown by bankfull shear stresses exceeding the critical shear stress required to mobilise the median particle size).

Simple comparison of total cover in unregulated and regulated sites is a relatively coarse comparison, since it is known that the influence of flow regulation on submerged macrophytes in the study area is highly dependent upon local-scale (within-site) habitat characteristics (Mackay 2007). A more refined method of examining the effect of flow regulation on total cover (Englund et al. 1997; Zhang et al. 1998; Mackay 2007) showed that total cover was higher in regulated sites compared with unregulated sites.

It is clear from this method that the response of aquatic vegetation to flow regime changes downstream of dams is site specific and supports the work of Mackay (2007), who found that submerged macrophyte cover in the Brisbane River downstream of Wivenhoe Dam was higher than expected, although the effect of flow regulation detected was very small. Thus Hypothesis 5 was supported by the results of this study.

Evidence for the effect of flow regulation on aquatic vegetation was shown by the divergence in assemblage structure of regulated sites from unregulated sites in RFCs 1 and 5. RFC 1 includes Obi Obi Creek downstream of Baroon Pocket Dam and RFC 5 includes the Nerang River downstream of Hinze Dam. Aquatic vegetation composition at regulated sites was not significantly dissimilar from that at unregulated sites, with the exception of sites downstream of Hinze Dam on the Nerang River (Hypothesis 5).

However these differences could be due in part to the far coarser substrates of the Nerang River compared to other Reference sites in the same RFC, which have sandy substrates. In other words, flow is not the only factor that can bring about differences in aquatic vegetation between regulated and unregulated sites within a flow class. This follows from the vegetation conceptual model (Appendix 2).

Variable discharge regimes, such as the discharge regime of HFC 2, should favour vegetation taxa with traits that enhance survival or persistence in such environments. The flow regime of HFC 2 is characterised by long duration of high spells, long periods of zero discharge and low discharge constancy (Chapter 3). Aquatic vegetation dominating this flow class may be expected to be tolerant of desiccation and exposure, either through establishment of seed banks or morphologic variation. In particular, amphibious taxa should be especially dominant.

Most Brock and Casanova (1997) functional groups occurred with insufficient frequency to draw conclusions about the association of these traits with particular HFCs. However, ARF (Amphibious Fluctuation Responders, floating-stranded) were especially dominant in HFC 2 and ARP (Amphibious Fluctuation Responders, morphologically plastic) were rare in HFC 1.

Fish

This study tested for differences in fish assemblage structure among RFCs and HFCs and the effects of flow regime alteration on 14 metrics of fish response. There were significant differences in species richness, non-migratory fish species richness, total fish density, native fish density, alien fish density, non-migratory density and species density across the RFCs. This result supports the ELOHA prediction that distinctive flow regime classes will have different ecological characteristics (Hypothesis 2).

None of these univariate metrics of fish assemblage structure showed significant differences between regulated and unregulated sites within any of the RFCs. This suggests that flow regime change by dams and other factor has not significantly affected fish assemblage structure in the study area, or at least not during the time elapsed since dams were constructed (10–50 years ago). Fish assemblages could be on a trajectory of adjustment to flow regime change that is continuing to this day.

Three of 11 composite fish assemblage level metrics showed significant differences between regulated and unregulated sites within HFCs. Total species richness and native species richness were significantly greater (21% and 29%, respectively) in regulated sites compared with unregulated sites in HFC 3. This class consisted of 19 gauges located mostly in the Mary and Logan–Albert River catchments, with sites on the Logan and Nerang Rivers influenced by dams immediately upstream.

Higher species richness at regulated sites was associated with Gower metric values between 0.05 (Logan River) and 0.24 (Nerang River). Total species richness includes both native and alien species, suggesting that alien species may have inflated the values of total species richness. However, native species richness was even more elevated than total species richness, indicating that more native species were associated with sites below these dams.

Aquatic vegetation composition was also affected by flow regulation at sites downstream of Hinze Dam on the Nerang River. However, it is suggested above that these vegetation differences could be due in part to the far coarser substrates of the Nerang River compared to other Reference sites in the RFC 5, which have sandy substrates. Riparian vegetation below Hinze Dam has also undergone a shift in assemblage structure but only one species (*Streblus brunonianus*, Whalebone tree) distinguished between the regulated and unregulated sites.

Given that sites on the Nerang River have undergone changes in both riparian and aquatic vegetation, it is perhaps not surprising to find that fish assemblages also differed from the Reference condition for RFC 5. Environmental factors driving differences in vegetation composition below Hinze Dam (e.g. substrate composition) may also have affected fish assemblages, as would the actual riparian and aquatic vegetation structure at these regulated sites.

Non-migratory species richness was almost double (98% higher) in one regulated site (Six Mile Creek regulated by Six Mile Creek Dam) when compared with unregulated sites in HFC 4. Again it is notable that one of the greatest differences in flow regulation effect on aquatic vegetation also occurred at the site on Six Mile Creek, even though flow regime change has been relatively slight (0.052 on the multivariate Gower scale).

Densities of Duboulays rainbowfish (*Melanotaenia duboulayi*) were significantly higher (18.9%) in regulated sites when compared with unregulated sites in RFC 2. In contrast, densities of Pacific blue-eye (*Pseudomugil signifier*) were significantly lower (77.6%) in regulated sites when compared with unregulated Reference sites in RFC 1, and again, Six Mile Creek showed a marked response. No other individual species showed significant differences between regulated and unregulated sites within any RFCs.

9.1.7 Limiting flow variables

Objective 7: *To identify a limiting suite of flow variables that together govern the condition or 'health' of each river system (or river zone, or set of rivers in a bioregion) and thresholds levels of ecological response to flow regime alteration for the whole suite of flow variables.*

These objectives cover similar themes and for efficiency they have been addressed in one composite summary.

Importance of flow

Across the three biotic components studied, hydrologic metrics alone explained relatively low proportions of observed variation in patterns of assemblage composition in the study area. The figures are: riparian tree and shrub assemblages 14.08%, compared to 4.1% for aquatic vegetation (based on species cover data). For fish 8.97–20.34% of variation in assemblage structure (based on CPUE) was explained by short-term flow variables (4 years of antecedent flows) and flows were most important in the first sampling period (July–August 2009), whereas long-term flow variables (15 years of antecedent flows) explained 1.24–9.43% of spatial variaton in fish assemblages and were also most important in the July–August 2009 sampling period.

These findings indicate that the importance of flow is situational, and that climatic and catchment variables set the context for the responses of biota to flow variability and flow regime changes due to dams and other factors. This has implications for the management of flows and for maintenance of stream ecosystem health and monitoring of ecological health and condition, as discussed below.

Is there a limiting suite of flow variables that together govern the condition or 'health' of each river system (or river zone, or set of rivers in a bioregion)? Certainly some of the 35 flow variables tested were more important to riparian and aquatic biota than others. For the streams studied here, flow variables important to riparian vegetation, aquatic vegetation and fish are given in Table 9.1. These variables have been identified by several types of analysis (ordination techniques, analysis of variance, ANOSIM, Kruskal–Wallis test and regression).

Table 9.1: Summary of flow variables correlated with the structure of riparian tree and shrub communities, aquatic vegetation assemblage composition and fish assemblage structure (based on CPUE)

Q = flow of sufficient volume and frequency to mobilise the mean particle size (D50) of stream substrates.

Biotic component	Important flow variables
Riparian vegetation	coefficient of variation in dry season flows (CVDry) mean bankfull shear stress (BFShear) mean bankfull discharge (BFDIS) a number of other hydrology variables were also important particularly CV, Med and BFDur.
Aquatic vegetation	Discharge required to mobilise the median particle size, Q_D50MOVE logarithm of the frequency of discharge events required to mobilise the median particle size occurred (FD50MOVE)
Fish	number of zero flow days mean daily flow baseflow index mean daily baseflow minimum daily flow magnitude coefficient of variation daily flows (CVDaily) magnitude of 10th percentile flow number of floods greater than median daily flow number of floods greater than median daily flow magnitude of 1-yr ARI constancy and predictability of mean monthly flow

Riparian vegetation

A number of significant relationships were identified between selected riparian vegetation metrics and flow metrics based on GLS regression. The CV in the dry season flows was negatively linearly associated with a number of riparian vegetation metrics – species richness (RICH), species density (D_SPECIES), the density of natives (D_NATIVES), the density of late successional species (D_LATE), the basal area of late successional species (BA_LATE) and the density of native regeneration (D_REGEN_NATIVE).

It is suggested here that variation in dry season flows may be critical, as riparian vegetation may be more reliant on stream flows during the dry season when rainfall is lower. Variation in flows during low rainfall periods may result in vegetation being subjected to intermittent moisture stress. Furthermore, variation in flows at this period of the year may result in frequent spells during which the streams cease to flow, resulting in a dropping of the local riparian groundwater table.

Bankfull discharge was also negatively linearly associated with three riparian vegetation metrics (near-stream species richness (RICH), species density (D_SPECIES) and near-stream species density) such that high values of these metrics were associated with low bankfull discharges.

Some more complex relationships between selected flow variables and riparian metrics were also indicated. Species richness (RICH) was minimized at intermediate values for the CV of daily flows. Riparian metrics (near stream species richness, species density and the density of intermediate successional stage species) appeared to be maximized at intermediate values for bankfull shear stress but only one of these quadratic relationships was significant (near-stream species richness), regression analysis for the other riparian metrics found a linear fit was better.

This result provides some very tentative support to the idea that riparian species diversity may be maximised at intermediate levels of disturbance, with low levels of disturbance resulting in dominance by only a few species and high levels of disturbance precluding establishment and maturation of many species.

Aquatic Vegetation

For aquatic vegetation, the hydrological variables with most influence on spatial patterns and flow–ecology gradients were the discharge required to mobilise the median particle size (Q_D50MOVE) and the logarithm of the frequency of discharge events required to mobilise the median particle size (FD50MOVE). Three significant relationships were identified between selected vegetation metrics and flow metrics based on GLS regression (aquatic vegetation Hypotheses 2–4).

The percentage of days in the 12 months prior to sampling when discharge was above the threshold required to mobilise the median particle size (FD50MOVE) was a significant predictor of total vegetation cover and emergent species cover. The third model included the discharge required to mobilise the median particle size as a significant predictor of total vegetation cover.

The relationships between vegetation metrics and flood frequency metrics (FD50MOVE and HSNum) were negative, indicating that higher flood frequencies reduce vegetation cover. The implication for environmental flow management is that reductions in the frequency of substrate mobilisation will result in increased in-stream vegetation cover, mostly due to increases in emergent vegetation.

A measure of flood frequency related to mobilisation of the median particle size (FD50MOVE) was apparently a more sensitive measure of flood frequency than the number of high spells (HSNum) because it captures a mechanistic process, namely that substrate mobilisation is a key mechanism of plant biomass loss during high flow events.

In this regard, the results support the aquatic macrophyte conceptual model (Appendix 2) and the first hydro-ecological principle of Bunn and Arthington (2002).For fish, the most important flow metrics are number of zero flow days, mean daily flow, CV of daily flows, mean duration of low spells (75th percentile), magnitude of ARI_1yr, and the constancy and predictability of mean monthly flows.

Fish

Eleven flow metrics influenced the structure of fish assemblages in streams of the study area (Table 9.1), with baseflows, gradients of zero and low flows, and flow variability of most importance during the 4-year antecedent flow period. When the 15-year antecedent flow period was considered, high flow metrics and the constancy and predictability of mean monthly flows emerged as more important than low flow metrics.

Two widely distributed fish species (Duboulays rainbowfish and long-finned eels) were tested for their relationships with gradients of variation in flow magnitude (mean daily flow), flow variability (CV of daily flow) and flow regime harshness (percentage of zero flow days) (Hypothesis 3). These flow metrics were selected for analysis because they were shown to have a significant influence on patterns of fish assemblage structure in multivariate analyses (Hypothesis 1). Total species richness was also tested for its relationship to the same metrics of flow magnitude and variability in the study area, given that there were differences in total species richness and native species richness between some regulated and unregulated sites (Hypothesis 2).

Densities of Duboulays rainbowfish (*Melanotaenia duboulayi*) and long-finned eels (*Anguilla reinhardtii*) and total species richness did not show any clear or significant relationship with gradients of variation in flow magnitude, daily flow variability and percentage of zero flow days. It appears probable that the full gradient of flow variability across the SEQ study area was not sufficiently extreme to be reflected in predictable patterns in the abundance of these widespread species, nor in fish species richness, even though the flows tested have been shown to have affected species richness and species occurrence in fish assemblages.

No significant relationships between individual flow variables and fish metrics (total species richness, native species richness and total density of native species) were detected. For environmental flow management the implication is that fish assemblages and species have a more complex dependence on the entire flow regime than can be elucidated from the study of relationships with individual metrics. This is consistent with previous research in the region (e.g. Kennard et al. 2007; Stewart-Koster et al. 2007).

In summary, these findings show that when all biotic components are considered, the important features of flow regimes in SEQ range across the full suite of variables known to influence the biota of streams and rivers, that is the magnitude, frequency, duration, timing and variability/predictability of low to medium and high flows (Poff et al. 1997).

They confirm the importance of considering the entire flow regime across the lowest to the highest flow magnitudes as well as flow variability, spell durations and the overall seasonal pattern of the flow regime (expressed as the constancy and predictability of mean monthly flows). These results indicate that the entire flow regime has played a part in structuring the composition of all three biotic assemblages studied in SEQ streams/rivers. It follows from this that the entire flow regime must be managed, not simply a few flow variables as typically employed in benchmarking studies (Brizga et al. 2002).

Effects of dams and flow regime change

Dams and flow regulation/supplementation have had significant impacts on riparian and aquatic vegetation and fish in SEQ streams and rivers. These impacts varied from one RFC to another, and appeared to reflect the particular changes in flow brought about by each dam. These findings are consistent with the ELOHA framework, which predicts that ecological responses to flow regime change by dams or other factors will vary according to the type of flow regime (i.e flow class) and the particulars of how the flow regime has been altered.

Each dam studied in SEQ altered the Reference (IQQM) flow regime in a different way, that is no two dams produced exactly similar hydrological changes. It is therefore to be expected that ecological responses would differ across the flow regime classes and between some regulated sites and their Reference sites, but not others.

The ELOHA framework predicts that hydrologically altered streams that come to resemble unregulated streams should be similar ecologically if flow is the major driver of biotic response. This prediction was supported for some ecological response metrics and for some HFCs, but not for all metrics and all flow classes. Again this result is a reflection of the particular types and degrees of flow regime change associated with each dam. For example, tree and shrub assemblage structure at regulated sites on the Nerang River below Hinze Dam was significantly different to that at unregulated sites in RFC 5, but when reclassified according to Historic flow data, riparian assemblages were more similar to those of unregulated sites within this class. This type of pattern was repeated for aquatic vegetation composition and for Duboulays rainbowfish (*Melanotaenia duboulayi*) at some regulated sites.

Some of the greatest differences in flow regulation effect on aquatic vegetation and fish occurred at sites downstream of dams where overall flow regime changes have been relatively minor on the Gower scale. For instance, at regulated sites on Six Mile Creek (Gower metric of 0.052) some individual metrics have changed markedly. The mean annual 1-day, 3-day, 7-day and 30-day minima are in the range 50–100% lower than at Reference sites, and the low spell duration is 100% higher below Six Mile Creek Dam.

Such large reductions in low flow levels and the huge increase in duration of low flows could be responsible for the lack of aquatic macrophytes at this site, In addition, Six Mile Creek sites were characterised by harsh conditions for in-stream aquatic vegetation (sandy substrates, encroaching riparian canopy cover). There was a 77% reduction in densities of Pacific blue-eye (*Pseudomugil signifier*) at this site, possibly due to loss of macrophyte habitat or unpredictability of habitat availability.

These findings again support the perception that each dam must be managed individually according to the flow regime class within which it is located, the particulars of flow regime change from the modelled unregulated condition (Reference condition) and the ecological responses to these changes in flow.

Relationships between flow gradients and ecological response

Predictable ecological response to flow regime change were sought by means of two main types of statistical analysis; firstly, in terms of observed ecological responses plotted against the multivariate Gower metric and secondly, by estimating observed versus predicted ecological responses (from PLS models) and plotting these against the Gower metric and/or against individual flow metrics (the latter analysis follows Poff and Zimmerman 2010).

Using the PLS Reference models to predict the metric values for regulation sites showed a significant effect of flow regulation for the density of reeds, rushes and sedges (D_LOMAND). Densities were found to be higher in all regulated sites than predicted from the non-regulated sites.

Species density (D_SPECIES: richness standardised by sampling area) was significantly lower in strongly regulated sites (so excluding sites on Obi Obi and Six Mile Creek), but an effect of flow regulation could not be detected from the PLS regression models as the 95% confidence interval included zero. No effects of regulation could be detected for species richness (RICH), the density of natives (D_NATIVES) or the density of native regeneration (D_REGEN_NATIVE). These results do not support the hypothesis that increasing flow regime change will result in increasing biotic change.

For aquatic vegetation, the impact of flow regime change on TOTCOV was significantly correlated with the Gower metric but, in contrast to the predictions of the ELOHA framework, the effect of flow regulation on TOTCOV decreased with increasing flow regime change from natural. This may be due to the fact that flow regime changes downstream of dams in the study area vary from dam to dam.

Poff et al. (2007) suggested that dams homogenise faunas through changes in flow variability. It is clear from the analyses of this study that dams are not necessarily homogenising aquatic floras but rather, the effects of flow regime changes are dependent upon within-site habitat features such as riparian canopy cover and substrate size. This implies that the effects of environmental flows on aquatic vegetation would vary substantially within a reach.

For fish, an effect of flow regulation was found for the relative density of native fish relative to alien species; this metric was lower than expected at two regulated sites and higher than expected at 10 sites. This implies that a high density of alien individuals was rarely associated with flow regime change in the study area, in contrast to the common observation that the converse occurs (Bunn and Arthington 2002).

Furthermore, when deviations from expected were plotted against the Gower metric, there was no obvious pattern of response to the overall degree of flow regime change. An effect of flow regulation was also found for non-migratory species richness; this metric was lower than expected at six regulated sites and higher than expected at six regulated sites. When deviations were plotted against the Gower metric, there was no obvious pattern of response to the overall degree of flow regime change.

Deviations of these two fish metrics from the expected did not show any consistent relationships with individual flow metrics, such as the CV of daily flows. Collectively, these results do not support the hypothesis that increasing flow regime change will result in increasing biotic change.

However, they do show very clearly that each dam has had a different effect on fish species richness and assemblage structure in terms of native versus alien species. Some of the patterns are consistent with expectations from the literature, but others are not. The common observation that alien fish species can flourish at regulated sites was upheld at only two sites of 12 studied.

In summary, the ELOHA prediction that increasing flow regime change will result in increasing biotic change was not supported for any biotic component. The lack of any gradients of biotic response to overall flow regime change (Gower metric), and to very few individual flow metrics shown to influence each biotic component, could be a function of the relatively low level of overall flow change, the relative recency of dams in the region (constructed or altered 10–50 years ago) and the number of more strongly regulated sites available for testing this relationship (six).

It was not feasible to test for gradient relationships within each flow class for the same reason (lack of replication of regulated sites). Lack of strongly regulated sites is a function of the particular study area (SEQ) and from an ecological perspective is a positive outcome. For the full testing of concepts embedded in the ELOHA framework, it presented difficulties. Tests of the ELOHA framework in other regions of Australia and in other countries may reveal different outcomes and are strongly recommended.

9.2 Influence of land use

Objective 8: *To assess the relative influence of flow regime alteration versus other pressures (e.g. land use extent/type, riparian degradation, water quality impairment, presence of alien species) on habitat condition and ecological condition/health.*

Whilst stream flows are recognised as one of the principal influences on stream ecology, many other catchment characteristics not directly related to stream flows are also important drivers of stream habitat structure and ecological processes. A key question for the ELOHA field trial, and for the ELOHA methodology as a whole, is whether the influences of flow can be extricated from the influences of other environmental gradients and anthropogenic disturbances.

The effects of environmental factors other than flow were apparent for all biota tested for their responses to gradients of environmental variability in the study area. That such gradients exist is expected and unavoidable (Chapter 5). In the SEQ study region there is considerable natural environmental variation in terms of geology and climate. The area exhibits a fairly strong rainfall gradient with a decrease in rainfall in a westerly (inland) direction across the catchments.

Given that catchment size, shape and topography are important influences upon stream number, size, water yield and hydrograph shape, it is unsurprising that differences in morphology and catchment variables were detected amongst flow classes. It is surprising that differences were not detected amongst a greater number of variables.

An issue for any ELOHA trial is whether hydrological Reference sites (based on classification of IQQM flow data) are appropriate references with respect to other key landscape and land use characteristics, and in some instances it appears that they were not. For example, regulated sites on the Nerang River below Hinze Dam differed in their substrate characteristics from Reference sites, which had sandy substrates (Chapter 7). Differences in other environmental variables can obscure the effects of flow regime change, and confound the extraction of predictable responses to flow impacts by dams.

A related issue is the potential effect of land use change. Fortunately for this study, there was little difference in land uses across the hydrological classes, and only weak correlations between flow alteration and different land uses. This finding suggests that the development of generalised relationships across gradients of flow variability and flow regime alteration is unlikely to be compromised by other co-varying land use disturbances within the catchments selected for study. The possible effects of these disturbances must still be considered, but may not directly influence any differential ecological responses among the flow regime classes.

Riparian vegetation responses to environmental gradients and land use disturbance

Environmental variables considered during the analysis of riparian vegetation structure included climatic factors, landscape factors, land use and disturbance regimes, reach scale geomorphology, in-stream habitat structure and flow history. Across the combined dataset from all sampling periods, 49.9% of spatial variation in riparian vegetation structure could be associated with local and catchment characteristics and flow, leaving 50.1% of variation unexplained. A slightly lower proportion of the variation in near-stream vegetation communities was explained (44.05%).

Catchment variables explained 16.33% of variation in bankfull riparian vegetation structure, with Coldest month mean temperature (CMA_TEMP), catchment relief (CAT_RELI), the proportion of felsic igneous geology in the upstream catchment, the proportion of mixed sedimentary and mafic igneous (MIXEDM) and the proportion of mafic igneous (MAFIC) geology the driving variables. Land use variables Production from dryland agriculture and plantations (PDA), and intensive uses (IU) explained 5.6% of variation. Hydrologic metrics alone explained a lower proportion (14.08%) of observed variation in riparian tree and shrub communities in the study area relative to the other environmental variables.

Climate variables and in particular coldest month mean temperature (CMA_TEMP) were clearly important influences on both assemblage structure and individual riparian vegetation metrics.

Overall, this study has found relatively little evidence of catchment disturbance impacts on patterns of riparian vegetation across the study area. This is reassuring, as it should facilitate the study of flow change impacts on riparian vegetation, rather than presenting a confounded matrix of interactions that could obscure any effects of dams and flow regulation – the focus of this study.

Aquatic vegetation responses to environmental gradients and land use disturbance

Environmental variables considered during the analysis of aquatic vegetation structure included climatic factors, landscape factors, land use and disturbance regimes, reach scale geomorphology, in-stream habitat structure, water quality and flow history. Across the combined dataset from four sampling periods, 42.9% of spatial variation in vegetation structure could be associated with local and catchment characteristics, leaving 57.1% of variation unexplained.

Catchment variables explained 23% of variation in vegetation structure, with D50 (median particle size), RipCover, PIA (production from irrigated agriculture and plantations), PDA (production from dryland agriculture and plantations), Depth and BFSHEAR (bankfull shear stress) being the most important environmental parameters describing patterns in vegetation metrics. In particular, median particle size was especially important, being the most important metric in six of the eight random forest models.

Hydrologic metrics explained only 4.1% of variation in species cover data when compared with catchment-scale environmental variables (23%). Local variables (water quality, turbidity, riparian canopy cover and Reynolds number) explained 6.7% of overall variations in vegetation cover at the study sites.

Fish responses to environmental gradients and land use disturbance

Environmental variables considered during the analysis of fish responses included climatic factors, landscape factors, land use and disturbance regimes, reach scale geomorphology, in-stream habitat structure and flow history. Across all three sampling periods, 52.9–57.4% of spatial variation in fish presence–absence patterns could be associated with gradients in climatic factors, catchment characteristics, land use disturbance, geology, channel morphology, and with 4 and 15-year historical flows.

Fish distribution patterns (based on presence–absence data from each site) were strongly associated with climatic gradients in the study area. Rainfall and reach scale air temperature were the most important climatic variables influencing fish distributions. After accounting for the influence of climatic factors, catchment land use, geology and in-stream habitat structure, flow variables alone explained only an additional 5–6.5% of variation in fish presence–absence assemblage patterns.

In contrast, flow variables had more influence on numerical fish assemblage composition in the study area. Specifically, short-term flow variables explained 8.97–20.34% of variation in fish assemblage structure and were most important in the first sampling period (July–August 2009). Long-term flow variables explained 1.24–9.43% and were also most important in the July–August 2009 sampling period.

9.3 Practical advice on healthier rivers

Objective 9: *To provide information and guidelines on the relative influence of flow and other pressures on river ecosystems, and practical advice on how to manage particular combinations of flow alteration and the other pressures so as to achieve healthier rivers*.

Riparian vegetation

Application of the ELOHA framework to riparian vegetation in SEQ has shown that key concepts of the framework (i.e. streams with different flow regimes will have different riparian floras and increasing flow regime divergence from natural will be associated with increasing biotic change) were not fully substantiated.

Hydrologic metrics were considered to be relatively good predictors of riparian assemblage structure. Of the hydrological metrics studies here, the CVDry (May–October)) was particularly important both to the riparian assemblage structure and a number of riparian metrics. It is suggested here that variation in dry season flows may be particularly critical as riparian vegetation may be more reliant on stream flows during the dry season when rainfall is lower.

However, if flow was a primary driver of vegetation patterns one would expect the importance of flow to be greater (relative to other environmental and land use influences) for vegetation situated nearer the stream edge (relative to all vegetation within the bankfull confinds) as these vegetation assemblages are more likely to be impacted by flow disturbance and have greater contact with the riparian water table. Our results indicate that this was not the case as relationships between various hydrological variables and both riparian assemblage structure and metrics for near-stream vegetation, were generally weaker relative to the bankfull relationships.

Catchment characteristics and land use variables explained 16.3% of variation in bankfull vegetation structure. Amongst these, the coldest monthly mean temperature (CMA_TEMP) and underlying geology (FELSIC, MIXEDM and MAFIC) accounted for some of the spatial variation in riparian vegetation assemblages. The importance of climate was evident from both the analysis of the riparian assemblages and the riparian vegetation metrics. A climatic gradient was also evident across the RFCs and HFCs, a result that is not surprising given the importance of climate in driving flow regimes. However, the presence of this gradient confounds the attempts made here to understand whether stream flows are a direct influence on vegetation or simply a correlate for other environmental influences (i.e. climate).

The implications of these findings for stream managers and water planners are fourfold:

- Firstly, regulation may be having an effect on the riparian vegetation of SEQ streams. Given that riparian vegetation are generally long-lived relative to the time dams have been in place for most of the region, it is perhaps not surprising that the effects of regulation have been detected for a group of plants with shorter life histories (reeds, rushes and sedges), and hence sufficient time has elapsed since the construction of the dams to allow a response in these vegetation types to be manifested.
- Secondly, variation of streams flows during the dry season (May–October) are likely to be particularly critical to the health and long-term persistence of riparian vegetation, and hence particular attention should be focused on the effects of regulation on flow variation in this season.
- Thirdly, for riparian vegetation environmental flows need to be estimated in relation to their hydraulic consequences particularly in terms of flood characteristics i.e. bankfull shear stress, bankfull discharge and bankfull duration);
- Fourthly, these results do not rule out a significant impact of regulation on other components of the riparian flora. Insufficient time may have elapsed for an effect from regulation to be detected amongst slower growing vegetation types (even at the regeneration stage as tested here).

This study has demonstrated that riparian vegetation assemblages respond to natural landscape gradients (e.g. climate), catchment land use and components of the flow regime. Given the likelihood of further flow regime change by water resource development and climate change, flow regulation impacts need to be incorporated into the monitoring of stream ecosystem health. Riparian vegetation should be used as indicators of flow regime change, as well as being indicators of broader catchment, riparian and channel health.

The studies conducted here provide an excellent opportunity to set up long-term monitoring, such as every five years at the sites sampled during this study. This would provide the information needed to determine if the trends reported here develop into stronger patterns of riparian vegetation response to flow regime change as well as other changes (e.g. climate change).

Aquatic vegetation

Although aquatic vegetation composition and aquatic vegetation metrics varied across HFCs, it was unlikely that the flow regime itself was the primary driver of these patterns. Patterns in vegetation metrics across HFCs were similar to the gradients in channel morphology (bankfull width:bankfull depth) and latitude that occurred across HFCs. In summary, there was little evidence to show that clear relationships existed between aquatic vegetation structure and HFCs.

The study has shown that hydrologic metrics are, in general, poor direct predictors of vegetation assemblage structure. This should not be construed to imply that the flow regime is unimportant to aquatic vegetation. The flow regime is critical for at least two reasons. Firstly, this study and others (Riis and Biggs 2003; Mackay 2007) have demonstrated the importance of substrate stability as a driver of vegetation assemblage patterns. Substrate stability is a function of several factors, particularly stream bed particle size and shear stress.

While shear stress is not directly determined by hydrology, two of the parameters in the equation (hydraulic radius and slope) are expected to vary with varying discharge (Gordon et al. 2005). Thus the flow regime is an important driver of aquatic vegetation structure through well-established mechanisms of action on hydraulic habitat characteristics and their stability, as predicted in the first hydro-ecological principle of Bunn and Arthington (2002) and the aquatic vegetation conceptual model (Appendix 2). Secondly, the variable presence of water is vital as the medium that supports submerged vegetation, hence the importance of water depth for a range of species and functional groups (e.g. Duivenvoorden 2008).

Catchment characteristics and land use variables explained 23% of variation in aquatic vegetation structure. Amongst these, riparian cover and adjacent land use (irrigated agriculture, dryland agriculture and plantations) accounted for some of the spatial variation in aquatic vegetation assemblages. These factors influence aquatic vegetation through the resource axis of the aquatic vegetation conceptual model (Appendix 2).

The implications of these findings for stream managers and water planners are that:

1. waterway management requires management of the catchment, stream channel and riparian zone as a whole
2. the effects of flow regime change by dams and other factors need to be interpreted in terms of stream hydraulics and habitat structure.

The results of this study also have implications for environmental flow development and the monitoring and management of stream ecosystem health. For in-stream aquatic vegetation, environmental flows need to be estimated in relation to their hydraulic consequences. As an example, for environmental flows to maintain vegetation assemblages they should be based on natural patterns and frequencies of substrate mobilisation. The frequency of high flow occurrences that mobilise stream substrates should be maintained to support aquatic plant diversity and cover. Loss of high flows capable of mobilising substrates could have impacts on in-stream vegetation.

The ecological health of streams is a function of its biodiversity, and aquatic vegetation is an important component of that diversity as habitat for other biota (Pusey et al. 1993), as a food resource and as spawning sites for fish (Pusey et al. 2004). However, aquatic macrophytes were not included as biotic metrics in the SEQ ecosystem health monitoring program (EHMP).

This study has demonstrated relationships between flows and simple aquatic vegetation metrics (i.e. total richness and total cover). These metrics are easily estimated in the field without requiring detailed knowledge of aquatic vegetation taxonomy, since they are based on counts and estimates of substrate coverage.

Aquatic vegetation can therefore provide an easily applied and useful indicator of a wide range of pressures on catchments and streams, including climate change, catchment land use, the condition of the riparian zone, stream channel characteristics and flow regulation. Given the likelihood of further flow regime change by water resource development and climate change, flow regulation impacts need to be incorporated into the monitoring of stream ecosystem health. Aquatic vegetation variables could be used as indicators of flow regime change, as well as providing indications of broader catchment, riparian and channel health.

Fish

Fish assemblages showed several statistically significant impacts of flow regime change associated with dams in SEQ, when effects were tested using three types of data analysis (multivariate ordination, PERMANOVA and gradient analysis).

Each dam has altered the downstream flow regime in a different way, bringing about ecological changes that are unique to individual sites and flow regime classes. Impacts include changes in numerical fish assemblage composition and in the abundance of individual species. These records of ecological impact below dams can be used to develop environmental flow guidelines and water management rules for each dam.

Consistent and predictable fish responses to gradients of flow regime change were not revealed by any of the statistical tests applied, possibly because the overall degree of flow regime change associated with dams in SEQ is low (0.25 on a Gower scale of 0–1). Furthermore, the passage of time since dams were constructed or altered and flow regimes started to change is relatively short (10–50 years).

Results suggest that most fish species may still be on a trajectory of change from Reference (unregulated) condition rather than reaching some form of equilibrium with modified flow regimes. Therefore it can be expected that impacts could worsen with longer periods of flow regulation, or greater degrees of flow alteration, and/or under the influence of climate change in the region.

This study has found relatively little evidence of catchment disturbance impacts on patterns of fish assemblages in the study area. It must be noted that the study sites were deliberately selected to reflect gradients of flow regime change, and not to reflect other disturbance gradients, such as land use change. However, the effects of flow regime alteration on both flow regimes and riverine fish assemblages are likely to vary with increasing intensity of land use, interacting with water abstraction and, potentially, with climate change.

Fish distribution patterns (presence–absence data) were strongly associated with climatic gradients in the study area. Rainfall and reach scale air temperature were the most important climatic variables influencing fish distributions. Given this strong association with climatic variables, it can be expected that fish assemblages will be influenced by changes in climate, such as increasing air temperature and reduced rainfall/runoff or increasing frequency and severity of flooding.

Streams in the study area experienced several high flow events during the study period, and some were flooded again in early 2011. The effects of the 2011 floods on the fish assemblages of SEQ should be studied to determine if flooding has implications for the fauna of the region. A comparison between the observed effects of flow regime alterations by dams and the effects of recent floods could be instructive. Monitoring of fish populations along a gradient of change associated with the January 2011 floods would offer a means to place the impacts of dams and flow regulation into the broader context of extreme events.

9.4 Geographic scope of ELOHA study results

Objective 10: *To show how the findings of this study can be related to rivers and flow regime types beyond the geographic scope of this research project.*

The ELOHA framework explicitly acknowledges that the escalation of water demand and development often outstrips the pace at which thorough scientific assessments can be conducted, and that water-allocation decisions are often made without full and detailed knowledge of the potential consequences of flow regulation (Arthington et al. 2006; Poff et al. 2010).

This framework is designed to derive predictions of altered flows based on clustering similar catchments (using hydrological classification or bioregionalisation) and developing environmental flow rules for each 'cluster' or class of river catchments. This may allow extrapolation of flow–ecology understanding and models developed in one catchment to other catchments within the same hydrologic class, where class is designated by ecologically-meaningful hydrological and geomorphic descriptors.

The framework recommends a number of steps, starting with deriving a classification of catchments within or between regions and using this to develop flow–ecology relationships for each class or type of river. Follow on by bringing in societal values and management needs and using this information in an adaptive management context, adjusting the implementation through iterative monitoring (Figure 1.1).

To show how the findings of the SEQ ELOHA study can be related to rivers and flow regime types beyond the geographic scope of this research project requires collation and classification of flow data for any specified region. As a first step, this study has matched the hydrological classification for SEQ with the Australia-wide hydrological classification of Kennard et al. (2010a), as a basis for proposing an appropriate geographic scope for extending the ELOHA study outcomes (Table 9.2).

Kennard et al. (2010a) completed the first continental-scale classification of Australian riverine flow regimes. This classification, based on 830 minimally disturbed gauges and 120 hydrologic metrics, identified 12 flow regime classes across Australia. Four of these flow regime classes occur within SEQ: Perennial–Stable Baseflow, Perennial–Unpredictable Baseflow, Intermittent–Unpredictable and Highly Intermittent–Unpredictable Summer Dominated.

Table 9.2: Continental flow classes identified by Kennard et al. (2010a) from the classification of Australian flow regimes and their occurrence in the SEQ study area

Flow class	Flow class description	Occurrence in study area
1	Perennial, Stable Baseflow	Teewah Ck, Christmas Ck, Logan R at Round Mountain and Teviot Brook junction
2	Perennial, Stable, Winter Baseflow	-
3	Perennial, Stable, Summer Baseflow	-
4	Perennial, Stable, Unpredictable Baseflow	Logan/Albert catchment and further south into NSW
5	Rarely Intermittent, Unpredictable Winter	-
6	Rarely Intermittent, Predictable Winter	-
7	Intermittent, Unpredictable	Amamoor, Obi Obi, Kin Kin, Tinana, Six Mile, Warrill, Laidley, Canungra, Tallebudgera Cks, Mary, Mooloola, Caboolture, North Pine, South Pine, Albert, Logan, Coomera R.
8	Intermittent, Unpredictable Winter	-
9	Highly Intermittent, Predictable Winter	-
10	Highly Intermittent, Predictable Summer	-
11	Highly Intermittent, Unpredictable Summer	Throughout south-east Qld
12	Extremely Intermittent, Variable Summer	-

The geographic distribution of the classification of Australian flow regimes is described in Kennard et al. (2010a). The following test is based on this publication, with comments on its relevance to the SEQ ELOHA study area.

Continental class 1 streams (called Stable Baseflow streams) were perennial with comparatively high baseflow contribution (mean baseflow index = 0.35), high runoff magnitude and high constancy of monthly mean flows (Colwell's C = 0.37). These streams were highly predictable due to baseflow constancy, but had a relatively weak seasonal signal because discharge magnitude was relatively uniform throughout the year.

Streamflows tended to be very stable within years (i.e. low variability in daily flows) and among years, with low skewness and low rates of rise and fall. High-flow events (e.g. >1st percentile) were comparatively small, frequent and of short duration, and maximum flows generally occurred at a similar time from year to year (e.g. low variability in timing of maximum flows). Stable baseflow streams were generally small (median catchment area = 101 km^2), and were widely distributed geographically.

Continental class 1 streams occurred most frequently along the south-east Australian coast, including the SEQ ELOHA study area. Kennard et al. (2010a) commented that these streams were 'minimally influenced by the prevailing climatic signal due to the high baseflow contribution to runoff (driven by significant groundwater contributions)'.

The most common continental flow class type of relevance to the ELOHA study is the Perennial–Unpredictable Baseflow group (class 4, Table 9.1, Figure 9.1). Streams/rivers in this class were less predictable than those in classes 1–3 and had a relatively weak seasonal signal. Stream flows also tended to be less stable within and among years (i.e. high variability in daily and annual flow) and the timing of maximum flows (i.e. summer flows) was more variable.

The relative magnitude of floods of various annual return intervals was also higher than in other perennial streams. Such streams tended to be located in small (median catchment area = 224 km^2). Streams in this class were located mostly in the Logan–Albert catchments within the SEQ study area. Streams of this type are widely distributed across eastern and southern Australia, extending into northern NSW (Figure 9.1).

Continental flow class 7 (Unpredictable–Intermittent) is also widely distributed across eastern and southern Australia, extending further south than flow class 4 (Figure 9.1). Discharge patterns in Continental flow class 7 were of very low predictability, more variable (higher variability in daily and annual flow) and had relatively high skewness compared with other classes. The timing of maximum flows was also more variable.

Unpredictable–Intermittent streams and rivers (median catchment area = 299 km^2) were widely distributed on the eastern coastal fringe of the continent, especially at the junction of drainage divisions I (north-east coast) and II (south-east coast) where the climate is transitional between subtropical and temperate. These types of streams also extend into the eastern upper headwaters of the Murray–Darling drainage division and occur as well in north-eastern Australia (Figure 9.1).

Continental flow class 11 of the continental classification (Highly–Intermittent, Unpredictable–Summer) occur throughout SEQ. These streams differed from other Highly Intermittent systems in that minimum, and especially maximum, monthly flows were less predictable and exhibited weaker seasonality, and although still summer-dominated, the higher variability in Julian date of maximum flow suggests that high flows could occur at any time during the summer.

Class 11 streams also had much higher flow variability, skewness and rates of rise and fall, as well as higher relative magnitude of floods of various annual return intervals. Such streams were almost exclusively restricted to the north-east drainage division (much of Queensland) and typically consisted of large rivers (median catchment area = 863 km^2).

This comparison of the ELOHA hydrological classification for SEQ with the Australia-wide hydrological classification of Kennard et al (2010a) indicates the potential geographic scope for extending flow–ecology relationships. However, it must be noted that the ELOHA classification provides a finer-scaled discrimination of flow regime classes, therefore distinguishing six RFCs (IQQMs) rather than four documented in the continental classification.

Although the six RFCs can be matched approximately to four of the continental flow classes (Perennial–Stable Baseflow; Perennial–Unpredictable Baseflow; Intermittent–Unpredictable; and Highly Intermittent–Unpredictable Summer Dominated), there are some slightly equivocal matches.

The wide geographic spread of the matching hydrological classes would naturally embrace significant biogeographic differences, and could potentially render the precise application of outcomes from SEQ somewhat difficult at the species level (e.g. Jacobs and Frank 1996 for aquatic vegetation). Nevertheless, some species in streams of SEQ extend to the north and south of the ELOHA study area, therefore potentially allowing flow–ecology relationships to be extended to those regions and catchments.

The continental analysis described the long-term statistical pattern of the hydrologic regime, not the short-term history of hydrological events. This means that any extrapolation of ELOHA results from one study area to another should firstly explore ecological variables that represent long-term adjustments to natural flow variability and altered hydrological regimes (e.g. changes in species richness and ecological trait composition of communities; Poff and Allan 1995), rather than ecological variables that fluctuate directly in response to short-term hydrologic events such as recruitment-driven variations in abundance (Kennard et al. 2007).

Long-term ecological adjustments to altered hydrological regimes represent the outcomes of sustained pressure on individual species and their recruitment processes, and therefore provide a powerful indicator of strong impact. It will be most useful for managers to know that a particular type of flow regime alteration by dams can actually eliminate some species or trait groups and encourage others (e.g. alien species). Thus extrapolations from one biogeographic region to another may be feasible and useful if trait groups rather than species are studied.

Field validation is, of course, recommended to verify significant relationships and hydrological impact thresholds.

9.5 Educational Objectives

Objective 1: *To provide opportunities for young scientists to participate in a large, multi-disciplinary field research program and to gain experience in field studies and the analysis and communication of results.*

The project employed three Research Fellows (Dr Cassandra James, Dr Stephen Mackay and Dr Robert Rolls) and three Research Assistants (David Sternberg, Anna Barnes and Tim Howell). Casual staff also worked on field and laboratory analysis from time to time.

David Sternberg left the project in 2010 to take up a Ph D Scholarship at Griffith University, drawing upon his experience in the NWC project to develop his Ph D research proposal and successfully win a research scholarship.

Anna Barnes left the project in late 2010 to take up a position with the Brisbane City Council.

Tim Howell finalised his Ph D while working part-time on the NWC project and a paper written at that time has been published. Tim is now employed as a consultant ecologist.

All members of the project team have participated fully in meetings and decisions, and will be taking part in the writing of journal publications. The following publications have been completed and 12 journal papers describing this ELOHA trial and its implications for environmental flow assessment and management are in progress and planned.

9.6 Publications to-date

Arthington AH 2009, Ecology of Desert Rivers 2006, Richard Kingsford (ed.) 2006, Cambridge University Press, Australia/ New Zealand, 354 00+xiv, Book Review, *Australian Zoologist* 34 (4):502–504.

Arthington AH 2009, Australian lungfish, *Neoceratodus forsteri*, threatened by a new dam, *Environmental Biology of Fishes* 84:211–221.

Arthington AH and Balcombe SR 2011, Extreme hydrologic variability and the boom and bust ecology of fish in arid-zone floodplain rivers: a case study with implications for environmental flows, conservation and management, *Ecohydrology* 4: 708–720.

Arthington AH, Naiman RJ, McClain ME and Nilsson C (eds) 2010, *Environmental Flows: Science and Management*, Special Issue of *Freshwater Biology* 55 (1): 148 pp.

Arthington AH, Naiman RJ, McClain ME and Nilsson C 2010, Preserving the biodiversity and ecological services of rivers: new challenges and research opportunities, *Freshwater Biology* 55 (1):1–16.

Arthington AH, Olden JD, Balcombe SR and Thoms MC 2010, Multi-scale environmental factors explain fish losses and refuge quality in drying waterholes of Cooper Creek, an Australian arid-zone river, Marine and Freshwater Research 61 (8):842–856.

Leigh C, Sheldon F, Kingsford RT and Arthington AH 2010, Sequential floods drive 'booms' and wetland persistence in dryland rivers: a synthesis, *Marine and Freshwater Research* 61 (8):896–908.

Poff NL, Richter BD, Arthington AH, Bunn SE, Naiman RJ, Kendy E, Acreman M, Apse C, Bledsoe BP, Freeman MC, Henriksen J, Jacobson RB, Kennen JG, Merritt DM, O'Keeffe JH, Olden JD, Rogers K, Tharme RE and Warne A 2010, The ecological limits of hydrologic alteration (ELOHA): a new framework for developing regional environmental flow standards, *Freshwater Biology* 55 (1):147–170.

Sheldon F, Bunn SE, Hughes JM, Arthington AH, Balcombe SR and Fellows CS. 2010, Dryland river waterholes: A review of the ecological roles and threats to aquatic refugia in arid landscapes, *Marine and Freshwater Research* 61 (8):885–895.

Zhang Y, Arthington AH, Bunn SE, Mackay S, Jun Xia and Kennard M (in press 2011), Classification of flow regimes for environmental flow assessment in regulated rivers: the Huai River Basin, China, *River Research and Applications*.

10. Key outcomes and recommendations

10.1 ELOHA concepts tested and verified

Five testable concepts underpin the ELOHA framework. Each has been tested in the SEQ trial and confirmed (with some caveats). Results are summarised here. Implications for management are listed below and discussed through this Executive Summary.

- Rivers of the chosen region can be grouped into distinctive hydrological classes using ecologically relevant hydrologic (flow) metrics (measures of flow magnitude, seasonal timing and frequency of low flows and flooding, and overall flow variability). **YES**
- Within a defined geographic region, the ecological characteristics of rivers in each hydrological class are expected to be relatively similar compared to the ecological characteristics between the classes; therefore, these classes may represent distinct 'management units'. **YES, to some degree for all biotic components.**
- Rivers within each class that are 'regulated' by dams (and weirs, etc.) in the same way will show the same types of ecological response to flow regime change. **YES and NO, depending on biotic component.**
- Increasing degree of overall flow regime change on a scale of 0 (no change) to 1 (total dissimilarity) will have increasing impacts on ecological response variables. **NO, for six good reasons.**
- Increasing degree of change in selected flow metrics will have increasing impacts on ecological response variables. **YES, for some biotic components.**

Through the conduct of this field study, several new elements have been introduced that will strengthen future applications and management implications/benefits of the ELOHA framework. These new developments of the ELOHA framework are asterisked below. The implications of the research findings summarised above can be used to achieve eight management objectives.

- predict the ecological impacts of new dams
- define environmental flows for over-allocated rivers
- quantify the relative importance of flow regime as a driver of stream ecosystem health
- provide ecological 'indicators' for monitoring the outcomes and benefits of environmental flows
- quantify the relative importance of other drivers of stream ecosystem health*
- provide ecological 'indicators' for monitoring stream and catchment health*
- guide river flow management and integrated catchment management*
- provide information to support adaptation to climate change.*

10.2 Key outcomes of field studies

10.2.1 Hydrology

Flow regimes were described in terms of 35 flow 'metrics' selected for their ecological relevance and ease of computation using available software. Flow metrics characterised the five key facets of the flow regime (i.e. *magnitude, frequency, timing* and *duration* of discharge events, and discharge *variability/predictability*) whilst minimising metric redundancy. Hydrologic metrics were calculated using the RAP and the IHA package. Magnitude metrics were standardised by upstream catchment area to downweight the influence of these metrics on flow classifications.

Hydrological regimes vary markedly across the SEQ region in response to gradients of rainfall, proximity to the coast and landscape characteristics. Classification of the flow regimes of unregulated river basins (termed Reference flow regimes) in SEQ was based on modelled pre-development flow data derived from an ''IQQM. This analysis identified six RFCs that separated unimpounded flow regimes along a gradient of discharge magnitude (per unit of catchment area) and discharge variability.

A strong geographic element was present in the Reference classification. For example, RFC 4 included IQQM nodes in the west and north-west of the study area. This region has lower rainfall than the eastern part of the study area and consequently RFC 4 is characterised by long periods of low flow and low discharge magnitude per unit of catchment area. In comparison, RFC 5 included IQQM nodes in the eastern (coastal) part of the study area. This part of the study area is characterised by high rainfall and hence RFC 5 is characterised by relatively high discharge magnitude per unit of catchment area.

The geographic extent of flow regime alteration by dams and weirs (and possibly land use) in the study area is considerable. Comparison of modelled (Reference – IQQM) flow regimes and gauged (Historic) flow regimes demonstrated a gradient of flow regime change across the region reaching 0.25 on a scale of 0–1, where 1 equals complete change from Reference condition. All rivers in the study area showed some departure of the flow regime from their original character, even those not affected by upstream dams or weirs.

In general, the greatest flow regime changes in the study area have occurred in streams/rivers downstream of dams (i.e. Nerang River, Reynolds Creek, Yabba Creek, Lockyer Creek, Brisbane River and Burnett Creek). However, three of the 11 gauges with the greatest flow regime change are not situated downstream of dams. These are gauges on Running Creek, Mudgeeraba Creek and the South Pine River. Land use changes in these systems are extensive. The primary land use change in the Running Creek catchment is agriculture, while urbanisation is the dominant land use for catchments of Mudgeeraba Creek and the South Pine River. Furthermore, the presence of dams does not necessarily imply extensive flow regime change (e.g. Six Mile Creek has a flow change metric of 0.052, indicating relatively minor change from the Reference flow regime).

Individual flow metrics showed substantial increases and decreases compared to the original flow regimes of rivers that are now impounded. Magnitude metrics, especially low spell duration, have undergone the greatest change relative to the Reference value, with most gauges showing an increase in low spell duration under recent conditions (termed the Historic flow regime). These marked changes could represent the effects of dams on downstream flows, or levels of water extraction from impounded and regulated rivers, or effects of increasingly dry conditions over the study period, or all three processes.

Gradients in flow regime alteration described by an overall dissimilarity metric and individual flow metrics provided the opportunity to test how riparian and aquatic vegetation and fish respond to flow regulation (supplementation).

10.2.2 Ecological survey results

The ELOHA field trial surveyed aquatic (in-stream) and riparian vegetation and fish at a range of Reference (largely unregulated) sites in each RFC and at paired sites downstream of six major dams: Moogerah Dam (Reynolds Creek), Maroon Dam (Burnett Creek), Baroon Pocket Dam (Obi Obi Creek), Six Mile Creek Dam (Six Mile Creek), Borumba Dam (Yabba Creek) and Hinze Dam (Nerang River). Field survey methods and approaches to statistical analysis differed for each biological component in relation to hypotheses drawn from the literature reviews and the ecological characteristics of each element.

The riparian vegetation of SEQ is diverse with totals of 191 tree and shrub species and 43 vine species identified from 44 sites. Alien taxa comprised 23% of all riparian plants recorded. Aquatic vegetation surveys recorded 74 plant taxa from 40 sites, the most common plants being submerged species. Alien taxa comprised 27% of in-stream taxa recorded. Fish surveys collected 35 species from 40 sites, including five alien species which contributed 3.7% of the total number of fish collected.

ELOHA survey results document biodiversity and new species records across SEQ streams and rivers for the first time. These biological datasets and associated environmental measurements have numerous applications in environmental flow assessment, river health monitoring, integrated catchment management and, potentially, in adaptation to climate change. The database will be held by Griffith University and made available to other researchers and agencies upon formal request to the research team.

10.2.3 Ecological variation across flow classes

Survey results generally support the ELOHA prediction that distinctive flow regime classes will have different ecological characteristics. There were significant differences in 12 bankfull riparian metrics across the RFCs, but only one significant difference (species density) for the near-stream vegetation. In contrast, there was little evidence of clear differences in aquatic vegetation structure across flow classes defined by their recent flow history. Fish assemblages were significantly different across RFCs in terms of species richness, total fish density, native density and alien density.

These results suggest that there is merit in considering the Reference hydrological classes as distinctive management units with different biotic assemblage characteristics that should be maintained as a component of stream ecosystem health in SEQ. This ELOHA study provides support for monitoring overall stream health in relation to the different flow classes of streams identified here for the SEQ region.

10.2.4 Ecological responses to flow variability

Biotic assemblage patterns were clearly influenced by prevailing flow regimes. When all three biotic components are considered, the important features of flow regimes in SEQ range across the full suite of variables known to influence the biota of streams and rivers.

Riparian vegetation: Variation in dry season flows may be critical as riparian vegetation may be more reliant on stream flows during the dry season when rainfall is lower. Furthermore, variation in flows at this period of the year may result in frequent spells during which the streams cease to flow, resulting in a dropping of the local riparian groundwater table. For riparian vegetation, environmental flows need to be estimated in relation to their hydraulic consequences, particularly in terms of flood characteristics (i.e. bankfull shear stress, bankfull discharge and bankfull duration).

Aquatic vegetation: The frequency of occurrence of high flows that mobilise stream substrates should be maintained to support aquatic plant diversity and cover. Loss of high flows capable of mobilising substrates could have impacts on in-stream vegetation. For aquatic vegetation, environmental flows need to be estimated in relation to their hydraulic consequences, particularly in terms of the high flows that mobilise stream substrates and determine plant diversity and the extent of in-stream cover by aquatic vegetation.

Fish: Species richness in streams of the region is influenced by mean daily flow, the variability (CV) of mean daily flow, the number of floods greater than the median, and the constancy and predictability of monthly flows. The number of alien fish individuals per site increased with increase in the number of zero flow days.

For fish, environmental flows need to focus especially on zero and low flows and daily flow variability, as these flows affect the availability of suitable aquatic habitat. Also, as the number of floods increased, fewer native species and more individuals of some species were present, suggesting fish diversity has declined under the stress associated with more frequent floods.

Results from the field surveys and several forms of data analysis show that many hydrological characteristics of flow regimes have played a part in structuring the diversity and composition of riparian and aquatic vegetation and fish assemblages in SEQ streams and rivers. It follows from this that the entire flow regime must be managed, with emphasis on those variables of most relevance to biotic assemblages and particular species.

10.2.5 Ecological responses to other environmental variables

All biotic components were influenced by climatic gradients, catchment and geomorphic variables, land use and in-stream variables (e.g. habitat structure) as well as flow. In some sites, these variables had more influence on biotic assemblages than flow variables. This implies that environmental flows cannot be expected to achieve improvements in stream ecosystem health when managed in isolation from catchment and local scale processes. The condition of the catchment surrounding each stream reach must also be assessed and managed to achieve healthy stream and river ecosystems.

The measurement of catchment variables and types/degrees of land use during this field study adds a new dimension to the ELOHA framework. This expanded framework can assess the impacts of land use or any other stressor on stream ecosystem health and rate the relative importance of all stressors. Such information is essential to make decisions about the most effective stream restoration strategies. For instance, restoring riparian vegetation may be more effective than providing an environmental flow.

10.2.6 Ecological responses to flow regime change

It has been shown that by comparing flow regime changes for individual flow metrics (i.e. comparing values for individual metrics under the Reference and Historic flow regimes) two broad categories and four types of flow regulation by these dams can be identified (Chapter 3).

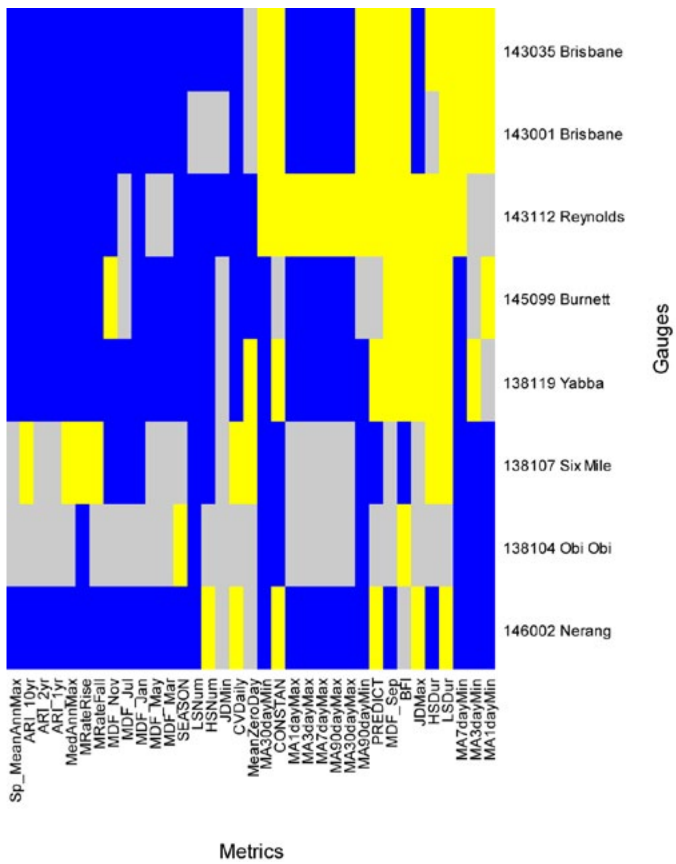
The first broad category of flow regulation has resulted in substantial increases (mostly >100% but also in the range 50–100%) in some metrics (i.e. Historic values are larger than Reference values). This group includes the Brisbane River downstream of Wivenhoe Dam (but upstream of the Lockyer Creek junction), Brisbane River at Savages Crossing (downstream of the Lockyer Creek junction) and Reynolds Creek.

The flow metrics that have increased in this group are related to discharge magnitude, spell duration and variability (Figure 9.4; Figure 3.15). This group has also undergone large decreases (50–100%) in the average recurrence intervals (ARIs) of one, two and 10 year floods (note that Reference values are still greater than Historic values in this case).

The second broad category of flow regulation has resulted in (mostly) small increases in some flow metrics (10–50%), but predominantly decreases of vary magnitude (mostly 50–100%) in flow metrics. This group includes Baroon Pocket Dam, Six Mile Creek Dam, Borumba Dam and Hinze Dam. There is some variation in the magnitude of the change in individual flow metrics within this group. The dams that have caused the least extent of change in downstream flow regimes are Baroon Pocket Dam and Six Mile Creek Dam (Figure 10.1).

Figure 10.1: Heat map showing the percentage difference between Reference and Historic flow metrics, expressed as (Historic–Reference)/Reference

MeanZeroDay is expressed as the difference between Reference and Historic values due to division by zero. Negative values indicate that the Historic metric value is higher than the Reference metric value, and positive values indicate that the Reference metric value is greater than the Historic metric value (see legend). Gray cells indicate a change of ± 10%. Yellow cells indicate an increase in value of greater than 10% and blue cells indicate a decrease in value of greater than 10%.



The implication for the development of environmental flow regimes is that each dam has to be managed individually rather that according to any general rules or guidelines for streams/rivers in a particular hydrologic class (a tenet of ELOHA). Furthermore, each flow component of importance to biotic assemblages in regulated reaches must be considered individually.

Flow regulation had significant impacts on riparian and aquatic vegetation and fish at some locations. Ecological responses to flow regime change varied in accord with the particulars of Historic (regulated/supplemented) flow regimes and other site characteristics.

Sites on the Nerang River were significantly different in their tree and shrub assemblage structure, and an effect of flow regulation was detected for the metric D_LOMAND (densities of reeds, rushes and sedges). Sites with regulation had higher densities of these species than predicted from non-regulated sites.

Sites on the Nerang River also differed significantly in aquatic vegetation assemblage structure. However, these differences could be due in part to the far coarser substrates of the Nerang River sites compared to other Reference sites in the same RFC, which have sandy substrates.

For fish, total species richness and native species richness were significantly greater (21% and 29%, respectively) in regulated sites on the Logan and Nerang Rivers relative to unregulated sites. Duboulays rainblow fish was significantly more abundant at some regulated sites and the Pacific blue-eye (*Pseudomugil signifier*) was 77% less abundant at other sites.

Although flow regime change had significant influences on riparian and aquatic vegetation and fish in the study area, predictable responses of biotic assemblages and species to flow regime change were rarely apparent for any of the biotic components studied here. The relatively low degree of flow regime change across the study area may account for this result (maximum change of 0.25 on a multimetric Gower scale of 0–1). Furthermore, the dams studied here are relatively recent (10–50 years) and biotic assemblages may not have adjusted fully to flow regime change. Some response variables (e.g. various fish metrics) appear to be moving along a trajectory of change but have not reached equilibrium with the new flow conditions identified from analysis of Historic (gauged) flow data.

It was not feasible to test for gradient relationships within each flow class for the same reason (lack of replication of regulated sites). Lack of strongly regulated sites is a function of the particular study area (SEQ) and from an ecological perspective is a positive outcome. For the testing of the ELOHA framework, it presented difficulties. Tests of the ELOHA framework in other regions of Australia and in other countries may reveal different outcomes and are strongly recommended.

10.2.7 Ecological response to flow change gradient

The ELOHA prediction that increasing flow regime change will result in increasing biotic change was not supported for any biotic component when selected biotic metrics were tested against the **overall gradient of flow regime change** across the study area. There were no statistically significant gradients of biotic response to this overall gradient of increasing flow regime change, expressed on a scale of 0 (no change) to 1 (complete dissimilarity).

The lack of any consistent and statistically significant ecological responses to the overall gradient of flow regime change could be a function of six interacting factors:

1. There was a relatively low level of maximum flow regime change (0.25 on a scale of 0–1), that is a relatively gentle gradient of change for testing.
2. This overall gradient incorporated six different flow regime classes with known ecological differences.
3. Each dam is known to have affected the downstream flow regime in a different way, so there was a mixture of changes in individual flow metrics embedded in the overall measure of flow regime change.
4. Dams in the region were constructed or altered 10–50 years ago, so ecological adjustments to altered flow regimes may still be occurring, especially for longer-lived riparian species.
5. The number of more strongly regulated sites available for testing the effects of the overall flow change gradient was low (six). Many more strongly regulated sites for testing might produce significant relationships.
6. Other environmental gradients in the study area have affected biotic assemblages and their influence confounds the ecological response patterns along the ‘pure’ flow regime gradient.

These factors would challenge most applications of the ELOHA framework in complex landscapes and set the scene for the innovative methods of data analysis discussed below.

10.2.8 Ecological response to changes in flow metrics

When the effects of environmental variables not related to flow were removed through a predictive modelling process, the impacts of change in flow metrics were more apparent, as described below.

Riparian vegetation: Using PLS regression models to predict the expected riparian metric values for regulated sites revealed a significant effect of flow regulation on the density of reeds, rushes and sedges. Densities of these stream-side plants were higher in all regulated/supplemented sites than predicted from their geomorphic and catchment characteristics. Species richness was significantly lower in strongly regulated/supplemented sites but the effect was not statistically significant. No gradients of response to flow regulation could be detected for species richness, the density of native species or the density of regeneration of native species.

Aquatic vegetation: Using PLS models to predict the expected total in-stream cover at stream sites did not reveal any significant gradients of plant response to change in eight flow metrics. However, plots of the ecological changes associated with change in each flow metric provide quantitative information about the impacts of individual dams.

Fish: Similar models were developed to predict the expected values of fish metrics at regulated sites. Deviations from expected of two fish metrics (density of native fish relative to alien species and non-migratory species richness) did not show any significant relationships with individual flow metrics. However, plots of fish responses to two flow metrics (CV of daily flows and the mean number of days without flow) show clearly that each dam has had a different effect on fish species richness and assemblage structure, as predicted by ELOHA.

Information from the PLS predictive modelling approach and the plots of ecological deviation from expected can be used to guide river flow management and restoration of over-allocated rivers, as illustrated in the management section of this document.

10.2.9 Influence of climate

Strong associations between climatic variables (temperature and rainfall), stream flows and all three biotic components suggest that streams of the region could be vulnerable to climate change (e.g. rising temperatures, low flows, longer dry spells and extreme floods). If land use and water abstraction patterns also change over time, it can be expected that stream ecosystem health will deteriorate as a consequence of the interactions and synergies between climate, land use and flow alteration.

Results of this ELOHA study can be used to provide guidance on possible ecological responses to climate change (e.g. by building predictive models of the anticipated response of species and trait groups to climate change scenarios). The ecological knowledge provided by this ELOHA study can also be applied in the development of strategies to combat climate change (e.g. studies to increase the resilience of riparian and freshwater species and ecosystems).

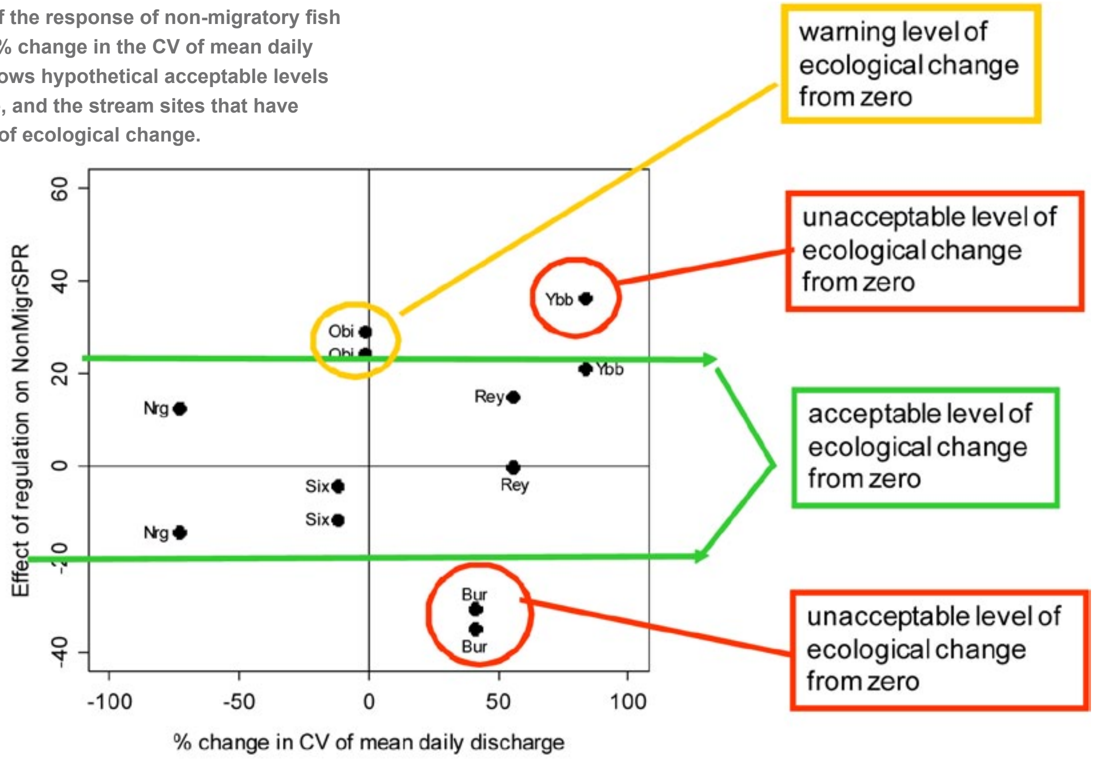
Flow regimes of the SEQ region have recently swung between severe drought and flood conditions, and several sites were markedly affected by one or other extreme state. Streams in the study area experienced several high flow events during the study period, and some were flooded again in early 2011. Biota of the region was presumably both resistant to flow variability and resilient to dramatic changes in flow ranging between extreme low and extreme high flows.

The strong association between climatic variables (temperature and rainfall) and all three biotic components suggests that streams of the region will be vulnerable to climate change (e.g. rising temperatures, low flows and longer dry spells). If land use and water abstraction patterns also change over time, it can be expected that stream ecosystem health will deteriorate as a consequence of the interactions and synergies between climate, land use and flow alteration.

10.3 Key recommendations for management

- ELOHA survey results document biodiversity and new species records across SEQ streams and rivers for the first time. These biological datasets and associated environmental measurements have numerous applications in environmental flow assessment, river health monitoring, integrated catchment management and, potentially, in adaptation to climate change. The database will be held by Griffith University and made available to other researchers and agencies upon formal request to the research team.
- Results from the field surveys and several forms of data analysis show that many hydrological characteristics of flow regimes have played a part in structuring the diversity and composition of riparian and aquatic vegetation and fish assemblages in SEQ streams and rivers. It follows from this that the entire flow regime must be managed, with emphasis on those variables of most relevance to biotic assemblages and particular species.
- This study has revealed widespread alterations to flow regimes by dams (and other stress factors such as land use) in the SEQ region, and documented the associated ecological impacts of flow regulation (supplementation) for the first time. The findings of this study can be used to review the management of environmental flows in SEQ as part of the WRP and ROP for the Moreton Region.
- Each dam has altered the downstream flow regime in a different way and the ecological changes detected are therefore unique to individual sites and hydrologic classes. Nevertheless, the records of ecological responses to change in each important hydrologic metric provided by this project can be used to provide guidance on achieving sustainable of water regimes.
- Ecological responses to changes in important hydrologic metrics can be presented as graphs that summarise all of the unique ecological responses to each type of hydrologic alteration downstream of individual dams and weirs. In the graph given in Figure 7.1, for example, hypothetical levels of acceptable change in this biological metric could be set at –20% to +20% change. Sites that exceed these hypothetical 'benchmark' levels of change can then be delineated as 'unacceptable level of ecological change from zero' and earmarked for review of the level of CV of mean daily discharge. Sites on Burnett Creek and Yabba Creek, for instance, have increased in CV mean daily discharge by 40–80%, and each change is associated with different ecological responses.
- This example highlights the fact that even the same type of hydrologic metric change can have different ecological implications and signals the need to look further into the other environmental variables that may be involved. While the modelling techniques applied in this study aimed to exclude the effects of other environmental variables, different in-stream characteristics may not have been fully eliminated by modelling. The individuality of ecological responses to dams is illustrated by this example.

Figure 10.1: Graph of the response of non-migratory fish species richness to % change in the CV of mean daily discharge. Graph shows hypothetical acceptable levels of ecological change, and the stream sites that have unacceptable levels of ecological change.



- Many examples of ecological responses to flow metric alterations presented as graphs of this type will provide guidelines to support decisions about environmental water allocations. The great utility of the ELOHA framework is that it is designed to provide this critical information for many waterways across a defined geographic region.
- Given the evidence of widespread alterations to flow regimes by dams and other factors across SEQ and the likelihood of further flow regime change by water resource development, increasing demand for water and climate change, it is recommended that flow variability and flow regulation impacts need to be incorporated into the ongoing monitoring of stream ecosystem health via the SEQ EHMP.
- The SEQ EHMP does not consider flow regulation as a probable risk to stream ecosystem health. From the findings of this study, it appears that the risks may be greater than presently presumed. Consideration of the utility of incorporating riparian and aquatic vegetation and fish metrics into the SEQ EHMP as indicators of flow regime change, land use change and, potentially, climate change, is strongly recommended.
- The findings of this ELOHA study show that riparian and aquatic vegetation and fish provide strong signals of catchment condition and land use, as well as flow regulation impacts.The field survey methods developed for this ELOHA study provide a sound basis for development of monitoring protocols to track ecological responses to flow regulation (supplementation), land use change and future climate change
- Monitoring programs should seek to collect biotic response data along gradients of flow regime change, environmental flow allocations and land use (following a common geographically referenced framework derived from this ELOHA study and the SEQ EHMP), to enable description of the relationship between catchment attributes and disturbances, flow characteristics and ecological outcomes.
- Streams in the study area experienced several high flow events during the study period and some were flooded again in early 2011. Fish showed the impact of more floods on species richness at study sites. The effects of the 2011 floods on the biotic assemblages of SEQ streams should be studied to determine if extreme flooding has implications for the health of the riparian and aquatic flora and fauna of the region. The ELOHA database provides the 'before flood' records that are often missing when such studies are contemplated. A comparison between the observed effects of flow regime alterations brought about by dams and the effects of recent floods could be instructive.
- A comparison of the ELOHA hydrological classification for SEQ with the Australia-wide (continental) hydrological classification of Kennard et al. (2010a) indicates the potential for extending flow–ecology relationships to the north and south of the study area.
- The wide geographic spread of the matching hydrological classes would naturally embrace significant biogeographic differences, and could potentially render the precise application of outcomes from SEQ somewhat difficult at the species level. Nevertheless, some species in streams of SEQ extend to the north and south of the ELOHA study area, therefore potentially allowing the flow–ecology relationships reported here to be extended to those regions and catchments.

- Long-term ecological adjustments to altered hydrological regimes represent the outcomes of sustained pressure on individual species and their recruitment processes, and therefore provide a powerful indicator of strong impact. It will be particularly useful for managers to know that a particular type of flow regime alteration by dams can actually eliminate some species or trait groups and encourage others (e.g. alien species). Thus extrapolations from one biogeographic region to another may be feasible and useful if trait groups rather than species are studied.
- Field validation is recommended to verify significant relationships and hydrological impact thresholds in other biogeographic regions.
- This ELOHA study provides a substantial dataset to formulate river restoration strategies for over-allocated rivers in the region, integrated with improved land use patterns to increase the resilience of riparian and aquatic ecosystems.
- Validation of the flow alteration – ecological response relationships revealed by this ELOHA study can be validated by restoring desirable flows to impounded streams and rivers, and monitoring the ecological outcomes. An adaptive monitoring and management framework integrated within the SEQ EHMP and Healthy Waterways Partnership is recommended.

10.4 Key recommendations for policy development

- Hydrologic alteration should be considered as a probable present and future risk to stream ecosystem health in south-east Queensland based on the findings of this ELOHA trial. The ecological effects of hydrologic alteration across south-east Queensland are likely to continue, particularly for longer lived riparian vegetation and fish that may still be responding to past as well as present changes in hydrologic regime..
- Hydrologic patterns are part of a complex interaction between a range of climatic and catchment factors that influence aquatic biota. This ELOHA trial has demonstrated that it is possible to tease out the relative effects of these factors using the statistical methods applied here.
- Metrics describing the condition of riparian and aquatic vegetation, fish communities and fish species could be used as indicators of hydrologic alteration in monitoring programs such as the Ecosystem Health Monitoring Program.They provide strong signals of hydrologic alteration, catchment condition and climate variability across the study area, and may therefore be useful indicators of land-use and climate change as well as hydrologic alteration..
- Ongoing validation of the relationships between hydrologic alteration and ecological responses detected by this study could be addressed by restoring desirable flows to impounded streams and rivers, and monitoring ecological outcomes. Restoration experiments provide the ultimate test of flow-ecology relationships.
- Priority for revisions of environmental flow arrangements should be given to dams that have had relatively strong impacts on flow regimes in the region, since the greatest ecological impacts of hydrologic alteration generally occur in association with moderate to strong flow regulation downstream of dams, particularly on the Nerang River downstream of Hinze Dam.

10.5 Key recommendations for research

- Further analyses of the ecological datasets produced by this project are recommended. Considerable knowledge with significant implications for management and policy could be extracted from this ELOHA database through further analyses addressing different questions from those asked in this project. Some key analyses that might be conducted include:
  - the identification of threatened or refugial habitats and biotic assemblages in rivers and streams and their riparian zones across south-east Queensland, and setting targets for restoration
  - development of population models for key riparian, aquatic plant and fish species to gain improved understanding of the importance of flow (and other factors) for life history stages, strategies and recruitment processes
  - modelling of climate change impacts under a range of scenarios for riparian and aquatic vegetation and fish assemblages of south-east Queensland
  - synthesis of the outcomes of further data analysis to support improvements to the ELOHA framework by suggesting ecological metrics that provide deeper insight into the ecological impacts of hydrologic alteration.
- Continued long-term monitoring of the sites and ecological components studied here is strongly recommended to capture more of the spatial and temporal variability of aquatic ecosystems in this region. This would contribute to validation of the patterns detected in this study and would also enable:
  - assessment of resilience and recovery from specific disturbances (e.g. the 2011 floods)
  - investigation of responses to further hydrologic alteration, land use and climage change.
- Key requirements to conduct such a trial include the availability of a good hydrologic monitoring network and either existing hydrologic models of pre-regulation discharge patterns or project team members skilled in hydrologic modelling. Ecological components considered in future trials may differ from those examined here, dpending on skills, knowledge and values placed on ecological assets in the selected regin. Considerable skills in statistical analyses of ecological datasets are essential as are strong collaborative relationships between researchers, managers and other stakeholders such as land owners and community groups.

The utility of the ELOHA framework has been demonstrated here; however, this is the first field trial in Australia. Further trials of the ELOHA framework are required across different types of aquatic ecosystems and along stronger gradients of hydrologic alteration than those present in the south-east Queensland study area.

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